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Fire in Eastern Oak Forests: Delivering Science to Land Managers

Proceedings of a Conference
November 15-17, 2005
Fawcett Center
The Ohio State University, Columbus, Ohio

Abstract

Contains 20 papers and 36 poster abstracts presented at a conference on fire in oak forests of the Eastern United States that was held at the Ohio State University, Columbus, Ohio, on November 15-17, 2005.

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FIRE IN EASTERN OAK FORESTS: DELIVERING SCIENCE TO LAND MANAGERS

Proceedings of a Conference
November 15-17, 2005
Fawcett Center, The Ohio State University, Columbus, Ohio

Edited by

Matthew B. Dickinson
USDA Forest Service, Northern Research Station
Forestry Sciences Laboratory, Delaware, Ohio 43015

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Historical Perspectives on Eastern Oak Forests

THE PALEOECOLOGY OF FIRE AND OAKS IN EASTERN FORESTS

William A. Patterson III[1]

Abstract.—Oaks (*Quercus* spp.) currently dominate eastern deciduous forests, but are widely perceived as declining, with regeneration inadequate to perpetuate many stands. Most stands regenerated following fire in the 19th and early 20th centuries, and a lack of recent fire is viewed as contributing to the shortage of sapling and pole-size stands. But paleoecological studies provide conflicting evidence for the role of fire in the long-term maintenance of oak forests. Here I describe the methods used in reconstructing past vegetation and fire regimes, review the results of previous studies, and present new results for sites from Virginia to Maine. Oaks have dominated mixed mesophytic forests in western Virginia for more than 6,000 years, with sedimentary charcoal levels suggesting a fire regime dominated by infrequent, light surface fires. The arrival of European settlers and a presumed increase in fire activity had little effect on oak abundance. At a higher elevation on more xeric soils, increased fire with settlement caused a shift from oak to pine dominance. On Long Island, NY, oaks have dominated xeric soils for thousands of years, but with more fire than in western Virginia. However, a dramatic increase in fire activity with settlement increased the importance of pines relative to oaks in southeastern Massachusetts' outwash plains. On the Maine coast, where oaks have been minor component of the vegetation for thousands of years, fires appear to have caused slight increases in oak importance both before and especially since European settlement. These results suggest that, depending on the landscape context, fire can favor or select against oaks, and that managers should carefully consider how fires will interact with climate, topography, and other factors before prescribing fire as a solution to the current lack of oak regeneration. It is likely that burn severity and fire return intervals, as they impact both oaks and their potential competitors, will determine whether or not individual oak stands will benefit from the reintroduction of fire.

INTRODUCTION

With at least 36 species, one or more of which occur in every state east of the Great Plains (Samuelson and Hogan 2006), oaks (genus *Quercus*) are the preeminent trees of the eastern forest. Twenty-one species are at least locally important as timber species (Burns and Honkala 1990). All, and especially those of the white oak (Leucobalanus) group, provide mast for game and nongame species of wildlife. Species not important for timber (e.g., the scrub oaks as a group) provide essential food and cover for several species of Lepidoptera (moth) larvae that are rare in at least portions of their ranges (Wagner et al. 2003). Oaks dominate in a variety of habitats, from rich cove forests of the southern Appalachians to dry ridgetops and sand plains of New England and the Atlantic Coastal Plain.

Throughout the range of oaks there is a widely held belief that regeneration is inadequate to perpetuate existing stands. Abundant deer, especially in urban interface areas, browse and kill seedlings and saplings. Late 20th-century fire suppression is blamed for the lack of regeneration in many stands. Tirmenstein (1991) citing several studies, observed:

> Fire has played an important role in deciduous forests of the eastern United States. Evidence suggests that most oaks are favored by a regime of relatively frequent fire. Many present-day oak forests may have developed in response to recurrent fire. Declines of oak forests have been noted throughout much of the East and are often attributed to reduced fire frequency.

Detailed information on the effects of fire on many oak species is available through the Fire Effects Information System. Most are viewed as being favored, at least in the regeneration stage, by fire. Tirmenstein (1991) noted, in discussing the fire ecology of white oak: "it is unable to regenerate beneath the shade of parent trees and relies on periodic fires for its perpetuation. The exclusion of fire has inhibited white oak regeneration through much of its range." Crow (1988) argued that frequent fires are required to maintain northern red oak (*Quercus rubra*),

[1]Forestry Program, University of Massachusetts, 214 Holdsworth Natural Resources Center, Box 34210, Amherst, MA 01003, 413-545-1970, email: wap@forwild.umass.edu.

and that increased fire frequency has caused, in some cases, the replacement of mesic hardwoods (e.g. sugar maple, *Acer saccharum*) by red oak. Conversely, reduced fire frequency is resulting in the replacement of northern red oak by mesic hardwoods in many areas of the Lake States and Northeast (Crow 1988).

The important role of fire in the development of oak stands during the historic period, i.e., roughly since the arrival of European settlers on the North American continent, is well documented (see Abrams et al. and Ruffner, this volume). Although this period spans many human generations, it represents no more than several for most oak species, which typically live 150 to 250 years with some (e.g., white oak) living more than 600 years (Tirmenstine 1991). Given the longevity of oaks and the time that it takes forest stands to develop, mature, and be replaced by new growth, managers should consider long-term interactions between oaks and environmental factors, including fire. The goal of this paper is to examine the paleoecology of oak as a genus, and to examine the interaction between oaks and fire over hundreds and even thousands of years. I start from the premise that it is through understanding the long-term history of the interaction of fire and oaks that we will gain the wisdom required to manage modern oak ecosystems and to perpetuate them far into the future.

PALEOECOLOGICAL METHODS

It is widely accepted that Native Americans more than lightning were the primary source of ignitions that burned through eastern forests prior to the arrival of Europeans (Bromley 1935; Day 1953; Cronon 1983). Although opinions differ on the extent of Native American burning (Russell 1983), there can be little argument that human-caused fires affected vegetation development, at least near Native American population centers (Patterson and Sassaman 1988). The first successful European settlements were in the early 17th century in Virginia and Massachusetts, but Mann (2002) argued that Europeans influenced Native populations, and by inference their effects on vegetation through burning, long before that. As early as the late 15th century with Spanish exploration in the South, and perhaps earlier (by Vikings to the north), Europeans brought diseases that decimated Native American populations throughout the East. To the extent that human populations were reduced, the influence of fire on vegetation probably was similarly reduced.

There are several methods for exploring fire and vegetation history: written accounts by explorers and naturalists, dendroecology and fire scar analysis, evaluation of 19th century photos in comparison with modern photographs (repeat photography), and, in the 20th century in particular, fire records and quantitative descriptions of fire effects. All of these methods have been applied effectively to describe vegetation and fire interactions before or at the time that Europeans settled the West. But few are useful in the East, where an understanding of fire regimes and fire-vegetation interactions unaffected by Europeans depends on a knowledge of how fire, vegetation, and humans interacted 500 or more years ago. Trees older than about 400 years are rare in the East, especially outside of swamps. Historical accounts described the land only at the time of observation, and extrapolation to before 1600 AD is tenuous (Mann 2002). Photographs, written records and ecological data from the 19th and 20th centuries tell us little about what might have occurred 300 to 400 years earlier.

Thus, paleoecologists have only fossil pollen and charcoal analyses of lake and bog sediments with which to investigate fire-vegetation interactions for the prehistoric period. Fossil pollen analysis, first developed by von Post (1916, cited in Davis 1963) in Scandinavia, has been used in North America since the 1930’s to reconstruct vegetation-climate interactions. Iversen (1941) first used charcoal analysis to investigate the effects of Stone Age people on vegetation of Denmark. But the technique has only recently (since about 1950 initially, and, widely, since the 1970’s) been employed in North America (Patterson et al. 1987). Both pollen and charcoal analyses involve examination of indirect evidence of vegetation and fire and are fraught with a series of assumptions and constraints. But the data they provide about periods in the past for which no other information is available is of great value.

Fossil Pollen Analysis

Paleoecologists obtain sediments from small ponds, lakes, and bogs using coring devices that preserve the stratigraphy of the core, with the youngest sediments at the top and older ones below. Cores are extracted in 0.5- to 1-m lengths and samples are withdrawn at intervals that depend on the rate at which sediments are estimated to have accumulated and the objectives of the particular study. A study of vegetation-climate interactions over thousands of years might involve sampling a core at intervals of 10 or more cm (100- to 200-year precision). Fire-vegetation interactions, are best described by data from cores sampled at one to several centimeters (with perhaps 5- to 20-year precision).

The theory of pollen analysis is well established. Von Post (1916), and, more recently, several North American studies observed that pollen is generally well preserved in cold, anaerobic, acidic, lake and bog sediments; that pollen of different species or genera can be distinguished one from another; and that for most species the proportion of pollen grains of different types in sediment samples approximates the proportion of plants of those types on the landscape. Sediments from cores drawn from small ponds or wetlands are more likely to reflect local conditions (Jacobson and Bradshaw 1981), with cores from bogs often containing information about vegetation of both the wetland and the surrounding upland.

Not all species can be distinguished based on their pollen; this is a particular problem for oaks. None of the 36 species native to eastern North America can be distinguished one from another based on pollen characteristics. Oaks as a group can be distinguished from other members of the Fagaceae family, e.g. beech and chestnut, and they can be distinguished from the hickories, birches, hemlock and pines, so it is largely at the level of genera that we can make inferences about fire and oak interactions. As with the other wind-pollinated species mentioned, oaks produce abundant pollen compared to maples and other largely insect-pollinated species.

Pollen data usually are expressed as a percentage, with the amount of oak pollen, as an example, a percentage of all fossil pollen identified on microscope slides after sediments are processed to remove inorganic and nonpollen organic constituents (Faegri and Iversen 1989). Percentage data suffer from the inherent limitation that if the absolute abundance of one constituent (e.g., oak pollen) increases (or decreases) dramatically, then the other constituents (e.g., pine, hemlock, hickory, birch and whatever other pollen types are identified as being present in the sample) must decline (or increase), on a precentage basis, even when the absolute number of pollen grains remains constant. The problem is illustrated with hypothetical data (Table 1A), where an absolute decline in hemlock pollen is reflected in an apparent increase in pine, oak, and other types even though they have the same absolute amounts in all samples. This problem can be circumvented if a known quantity (say 40,000 grains per cubic centimeter of sediment being processed) of a marker grain (pollen of a type not present in the local flora, e.g., *Eucalyptus* pollen for the Eastern United States) is added during processing of the sediment and counted along with the fossil grains. Where the amount of fossil pollen is compared with that of the marker grains, fossil pollen can be expressed as an absolute amount of pollen (number of grains per unit volume of sediment, see Table 1B).

If sediment accumulates at the same rate in a basin over time, the absolute pollen frequency (APF) allows species/genera represented in the data set to be evaluated independently of other types (Davis 1965). However, sediment accumulation rates are not always the same, so APF (number of grains/cm^3) is divided by sediment accumulation rate (cm/year) to yield a pollen influx rate (in number of grains/cm^2/year) (Davis and Deevey 1964). This measure of pollen abundance represents an ideal that is not always attainable because sediment accumulation rates cannot always be estimated accurately.

Cores are dated and sedimentation rates established by a variety of methods. Carbon-14 dating can establish absolute dates to 30,000 or more years before the present. But the accuracy of C-14 dates is on the order of decades to 100 years or more (Bradley 1999) and declines rapidly in sediments less than about 500 years

Table 1.—Hypothetical pollen data showing the effects a large change in the absolute amount of one pollen type (hemlock) on the percentages of other types. The number of hemlock grains (A) decreases from 150 to 5 from the deepest to the shallowest sample, whereas the number of all other types does not change. The decrease in hemlock grains is properly reflected in a sharp decline in hemlock pollen as a percentage of the total even though the total number of grains identified declines. But the percentages of the total represented by the other three types—pine, oak and all other types combined (other)—increase by more than 50 percent. Expressing the pollen as number of grains per cubic centimeter (absolute pollen frequency) yields a more appropriate interpretation (B). The addition of 40,000 grains of an exotic pollen grain per cc of sediment sample processed, and counting these grains along with the fossil grains, is assumed. Expressing data as APF still assumes that each sample represents the same number of years of accumulated sediment, an assumption that must be verified by careful dating of many sediment samples and the calculation of sediment accumulation rates for different portions of the core being analyzed.

A

depth	Pine #	Pine %	Oak #	Oak %	Hemlock #	Hemlock %	Other #	Other %	Total #	EXOTIC #
500	100	39.2	25	9.8	5	2.0	125	49.0	255	400
503	100	39.2	25	9.8	5	2.0	125	49.0	255	375
506	100	39.2	25	9.8	5	2.0	125	49.0	255	350
509	100	38.5	25	9.6	10	3.8	125	48.1	260	400
512	100	33.3	25	8.3	50	16.7	125	41.7	300	375
515	100	25.0	25	6.3	150	37.5	125	31.3	400	425

B

	#/cc	#/cc	#/cc	#/cc	#/cc
500	10000	2500	500	12500	25500
503	10667	2667	533	13333	27200
506	11429	2857	571	14286	29143
509	10000	2500	1000	12500	26000
512	10667	2667	5333	13333	32000
515	9412	2353	14118	11765	37647

old. Other isotopes, including Lead-210 (Pb-210), are useful for dating younger sediments. Lead-210 is a decay product of radon gas, which occurs naturally in soils. Concentrations of Lead-210 in sediments are used to establish sediment accumulation rates and to date sediments that generally are less than 150 to 200 years old. Cesium-137, a product of atomic bomb testing in the 1940's, allows dating of younger sediments.

Regionwide changes in the abundance of several pollen types have been dated independently. Examples include the decline in hemlock (*Tsuga canadensis*) in eastern North America approximately 5,000 years ago (Webb 1982) and the recent (1900-1920 AD) decline in chestnut (*Castanea dentata*) due to the chestnut blight (Anderson 1974). Increases in the abundance of pollen of ragweed (*Ambrosia* spp.) and plantain (*Plantago* spp.) mark the local advent of European agriculture from 300 to 350 years ago on the East Coast to 100 to 150 years ago in Minnesota.

The most precise estimates of sediment accumulation rates can be obtained from varved lake sediments, i.e. those that have alternating layers of light and dark sediment that correspond to changes in seasons within a year. A pair of one light and one dark layer forms a couplet that indicates a full year of sediment accumulation (Fig. 1). Several processes can form banded (or laminated) sediments (O'Sullivan 1983), and precise chronologies can be established from them. However, they are rare, having been identified in no more than a dozen or so lakes in Eastern North America.

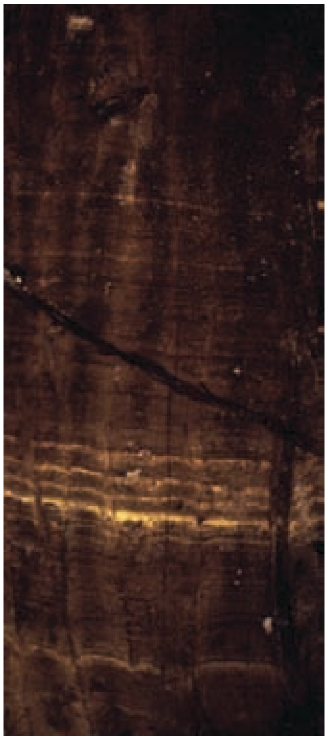

Figure 1.—Photomicrograph of varved sediments from a lake in northwestern Minnesota. Thin, horizontal bands of light and dark sediments throughout the core are thought to represent changes in the chemical composition of sediments related to seasonal variations in oxygen content of the bottom waters, with each pair of bands representing one year (Foster 1976). Broad bands in the middle of the section may have been formed by erosion events, and pairs may or may not represent one year's sediment accumulation (Patterson 1978). The core section is approximately 10 cm wide by 25 cm long, with the youngest sediments at the top of the section.

In summary, although imprecise by the standards applied to modern ecological studies, fossil pollen analysis techniques are largely standardized among research labs and limitations are well understood. Pollen analysis provides information about time periods for which no other information is available and is a useful tool for exploring hypotheses about past vegetation development and vegetation-environment interactions.

Sedimentary Charcoal Analysis

Sedimentary charcoal analysis is less widely used than pollen analysis, and there are a variety of competing techniques for quantifying past charcoal production i.e., fire activity. The half dozen or so U.S. labs routinely performing charcoal analysis tend to use a variety of methods that may or may not yield comparable results when applied to the same sediments (Patterson et al. 1987). Recent advances in the theory and methods of sedimentary charcoal analysis (Clark 1988a, b) are most useful when accurate and precise information on sediment accumulation rates is available.

Unlike pollen, which is produced annually in roughly the same amounts from similar vegetation types, charcoal is produced intermittently from fires that occur at different return intervals and that burn with different intensities in different fuels across varying potions of a landscape. Abundant charcoal is carried long distances in towering convection columns from large, intense crown fires burning in conifer forests. Conversely, charcoal produced by lower intensity surface fires burning in deciduous forest (e.g., oak and other hardwoods in the Eastern United States) is less likely to be transported long distances by air, even when large areas burn in a single fire. Short-distance transport by air and hydrologic transport in runoff probably characterizes these fires. Analysis of pollen data tends to focus on running averages that emphasize the year-to-year continuity in pollen production and gradual changes in vegetation, whereas fires can elicit dramatic changes in charcoal influx from one year to the next, and can dramatically affect pollen production within the area contributing pollen to a basin.

Patterson et al. (1987) discussed how charcoal is produced, transported to deposition sites, and eventually preserved in sediments. Larger charcoal particles signal locally occurring fires (Clark and Patterson 1997; Clark and Royall 1995a), so quantifying charcoal by particle size rather than by number of fragments has been emphasized. Charcoal often is quantified on microscope slides prepared for pollen analysis, but Clark (1988b) argued that it is the abundance of macroscopic charcoal (i.e., fragments longer than 150 to 200 microns) that provides the most information about local fire history. This is likely true for intense fires burning in heavy fuels in dry conifer forests. It is less clear if the generalization applies to fires burning through predominantly leaf (nonwoody) litter in deciduous forests.

Charcoal area (the sum of the sizes of all fragments in a sample) often is represented relative to fossil pollen content (i.e., square microns of charcoal:number of fossil pollen grains, or μ^2Ch:P) based on the idea that local fires produce large charcoal particles while at the same time reducing pollen production by killing plants. Because fires, and thus charcoal production, are transient on the landscape, reconstructing fire histories requires sampling sediments at close intervals and the analysis of contiguous (or nearly so) analysis of samples—what Green and Dolman (1988) refered to as fine resolution analyses. It is for this type of analysis that varved sediments can be most useful (Swain 1973; Clark 1988a). However, they are not required, as shown by Motzkin et al.'s (1993) detailed reconstruction of the fire and vegetation history of a Cape Cod cedar swamp. It is perhaps most important to sample small basins with well-defined watersheds relative to the area burned if one hopes to identify the response of vegetation to individual fires (Patterson and Backman 1988).

THE PALEOECOLOGY OF OAKS

Hundreds of sites have been cored and evaluated for fossil pollen content in North America. The National Oceanographic and Atmospheric Administration's (NOAA) Fossil and Surface Pollen Data website provides access to data for many of these. A map of the distribution of sites for which data are archived can be viewed at: http://www.ncdc.noaa.gov/paleo/pollen.html by clicking on the North American portion of the Web Mapper box. The cursor can be used to outline the eastern portion of the resulting view of North America to locate individual sites. Clicking on individual red stars allows the viewer to see a pollen diagram for that site, or access the original data used to generate the diagram. As an example, clicking on the one star in the state of Kentucky and then clicking on Diagram (rather than Data) provides view of the pollen diagram for Jackson Pond (Wilkins et al. 1991). The site has sediments that date to at least 20,000 years ago, and the pollen data show that oaks have dominated on the surrounding upland for approximately 10,000 years.

There are more than 100 sites for Eastern North America in the North American Pollen Data Base, and diagrams and data are available for each. Data from these sites have been merged to provide maps of North America which show the abundance of individual pollen types at discrete times (generally in increments of 500 or 1,000 years) in the past. By clicking on the Pollen Viewer box, you can access animated reconstructions of the spatial and temporal variation in the abundance of individual types. Using the scroll box to locate Quercus (5,20 and 40%) brings up color-coded range and abundance maps for oak for the past 21,000 years. Comparisons across many sites of oak pollen abundance in modern sediments with the occurrence of oak species on the surrounding landscape (as from forest inventory data) allow us to make inferences about the importance of oak as a component of past vegetation. When oak pollen exceeds approximately 5 percent, oak generally is at least present within the area contributing pollen to the basin. At 20 percent, oak is abundant; at 40 percent it dominates the overstory in the surrounding forests (Webb 1978; Webb et al. 1978). By clicking on the Play button, you can see changes in oak pollen percentages over the past 21,000 years and interpret the extent of oak trees on the landscape.

Animating the pollen viewer shows that oaks reached their current range limit by approximately 10,000 years ago and were most important during the Holocene, from 9,000 to 10,000 BP. They were the first hardwoods to establish and dominate following the retreat of boreal spruce-fir forests to the north, and their lower pollen percentages after about 9,000 BP is partly a reflection of the arrival and establishment of other species of the eastern deciduous forest (e.g., beech, the hickories and maples and, finally, chestnut). The pollen viewer shows a contraction of the area dominated by oaks (> 40 percent) during the past 500 years, especially in the Central States west of the Appalachians. This probably reflects the loss of forest land to agriculture in the Midwest. Similar declines (but more recent recoveries) in the Northeastern States have been attributed to overcutting of oaks for fuelwood in the 18th and early 19th centuries (Brugam 1978; Backman 1984). Cordwood shortages during this period are documented in the historic records for south-coastal New England (Stevens 1996).

THE PALEOECOLOGY OF FIRE

The NOAA web site referenced for pollen also provides access to fire history data: http://www.ncdc.noaa.gov/paleo/impd/paleofire.html. This site also links to a Web Mapper for North American sites, but the map shows only one "charcoal sediment" site east of the Mississippi River (at Indiana Dunes National Lakeshore). Pollen diagrams with accompanying charcoal data have been produced for dozens of sites in the Eastern United States by researchers from, among others, the Universities of Minnesota, Maine, and Massachusetts; Rutgers, Brown and Duke Universities; and the Harvard Forest (Harvard University). Several of these sites are included in the North American Pollen Data Base, but with no charcoal data.

The reasons for the paucity of data for charcoal for the NOAA sites are many, but perhaps the most important is the lack of a standardized way of reporting charcoal results. Summaries of data for some sites have been published (Patterson and Backman 1988; Patterson and Sassaman 1988; Clark and Royall 1996; Parshall and Foster 2002); From these summaries we can generalize that just before European settlement, the amount of charcoal in sediments decreased from west to east, with the lowest values in the mountainous regions of New England (Patterson and Backman 1988). For the most part, sedimentary charcoal content decreased with the arrival of European settlement west of the Appalachians, but increased in New England. This pattern generally is consistent with perceptions of the role of fire in eastern North America before and after European settlement (Pyne 1982).

THE PALEOECOLOGY OF OAK AND FIRE

Access to charcoal data in the same NOAA formats available for pollen would greatly facilitate an evaluation of the prehistoric role of fire in maintaining oak forests. For now, researchers are left with the evaluation of fire-oak interactions on a site-by-site basis. Reviews are available but the results are ambiguous. Clark and Royal (1995) documented a shift from northern hardwoods to white pine/oak forests with the establishment of a local Indian settlement about 1450 AD in southern Ontario, at Crawford Lake, and attribute the shift to Indian-ignited fires. In support of the role of Native American burning in the maintenance of oak forests, Delcourt and Delcourt (1997) used pollen, charcoal, and archeological data to conclude that "Native Americans played an important role in determining the composition of southern Appalachian vegetation through use of fire." They argued that fires would have stimulated sprouting "thereby increasing the abundance of chestnut and oaks growing in open groves."

Campbell and McAndrews (1995) disagreed with Clark and Royall's (1995b) Crawford Lake interpretation despite an apparent sharp increase in sedimentary charcoal at about the time that oak and pine percentages increased at the expense of elm, sugar maple, and beech. They attributed the change in forest composition to climate anomalies associated with the Little Ice Age. Clark (1995) defended the fire hypothesis but careful examination of the pollen and charcoal data (Figure 3 in Clark and Royall 1995b) apparently shows that oak pollen percentages began to increase just before (ca. 1400 AD) a major fire, as indicated by the highest charcoal influx values (centered on ca. 1450) in the core. Oaks producing pollen that contributed to the increase in percentages ca. 1400 AD probably established as early as 1350, given the several decades that it takes oaks to produce pollen after establishing from seed. Peak pollen values for oak (30 to 40 percent) are in sediments that postdate the ca. 1450 fire (approximately from 1500 to 1800). Clark (1995) notes that the Crawford lake site appears to show a different set of transitions (with a different cause—Native American burning vs. climate change?) compared to other sites in the region. At Devel's Bathtub in west-central New York, Clark and Royal (1996) found abundant oak pollen throughout the Holocene, but surprisingly little charcoal in the sediments. On the basis of data from this site, they questioned the dependence of oak on fire "at Devil's Bathtub and other sites in the northeast" (Abrams 2002).

Following the publication of their discussion with Campbell and McAndrews regarding the interpretation of the Crawford Lake site, Clark and Royall (1996) reviewed local and regional sediment charcoal for evidence of fire in presettelment northeastern North America. Thye reported that "We did not find evidence for fire in mixed oak forests, where it has been speculated that fire might be necessary for oak recruitment." Abrams (2002), reviewed several published pollen and charcoal diagrams, including those for Crawford Lake and Devil's Bathtub. Despite acknowledging the reservations of Clark and Royall (1996), he concluded that "many paleoecological papers show an intrasite increase in regional or local charcoal with increases in oak abundance."

My own review of published and unpublished work for this paper supports the somewhat ambiguous interpretation suggested by Clark and Royall. It is clear that at many sites charcoal is abundant when oak is dominant, or that across sites, samples from sites with more oak tend to have more charcoal than those from sites surrounded by more mesic, less fire-prone forest types. Crawford Lake notwithstanding, what appears to be lacking is evidence that an increase in fire has <u>caused</u> oak to become more abundant where it was less so. Most fire managers would acknowledge that litter fuels in oak forests are more prone to burn than those in mesic forests. Indeed, the northern hardwood forests of western New England are widely referred to by foresters as "asbestos forests" (Borman and Likens 1979), an acknowledgement of the low flammability of beech, maple, and birch litter. But questions remain: Is there more oak pollen in sediment samples with abundant charcoal because fire favors oak-dominated vegetation? Is there more charcoal in sediments with abundant charcoal because oak fuels are more likely to burn?

I believe that insights can be gained by examining individual sites. I next use data from sites in Virginia, southern New England/Long Island, and on the coast of Maine to illustrate some of what we have learned about the interaction between fire and oaks along the Atlantic Seaboard. Following the example of Jacobson (1979), I have chosen pairs of sites in each of the three regions to illustrate how oaks growing on different sites respond differently to fire.

Western Virginia

We reconstructed fire and vegetation histories for two different sites near Staunton in western Virginia (Patterson and Stevens 1995). Brown's Pond, at an elevation of 2,000 feet in Bath County (38°08′50″ N, 79°36′23″ W), lies in a protected cove and is surrounded by mixed mesophytic forests dominated by red and white oaks and a variety of associated hardwood species. Pines occur only occasionally on the surrounding uplands. Soils are deep (greater than 60 inches to bedrock) loams. Pollen (but not charcoal) data for sediments dating to 17,350 BP are available in the North American Pollen Data Base (Kneller and Peteet 1993). We obtained a 90-cm-long core from the site in 1994 and analyzed the sediments for pollen and charcoal content (Fig. 2). Increases in pollen of agricultural indicators (chiefly ragweed) mark the advent of cultivation by Europeans 300 years ago. Oaks have been the dominant species in the Brown's Pond watershed, with pollen ranging from 40 to 60 percent (of all pollen grains identified) in all samples. Charcoal to pollen ratios (Ch:P) range from 0 to more than 400. Our work on Mount Desert Island, Maine suggests that Ch:P values in excess of 150 to 200 indicate fires occurring in the watershed of small ponds, whereas lower values probably represent charcoal blown in from afar (Patterson and Backman 1988; Clark and Patterson 1997). Although individual peaks in charcoal abundance are evident, the sampling interval was broad and identifying individual fires and estimating the interval between them are not possible. Still, we concluded that fires have been part of the local landscape throughout the 6,000-year period represented by the core. Charcoal values were somewhat higher before Europeans arrived, but the overall abundance of fire on the landscape did not change with their arrival.

Green Pond is approximately 30 miles southeast of Brown's Pond at an elevation of 3,200 feet in Augusta County (37°56′23″ N, 79°02′51″ W). It lies on an exposed ridge known locally as Big Levels and is surrounded by chestnut (*Quercus prinus*) and black (*Q. velutina*) oaks, and pitch pine (*Pinus rigida*) over dense thickets of ericaceous shrubs. Soils are very stony loamy sands with depths to bedrock of 40 to 48 inches. A

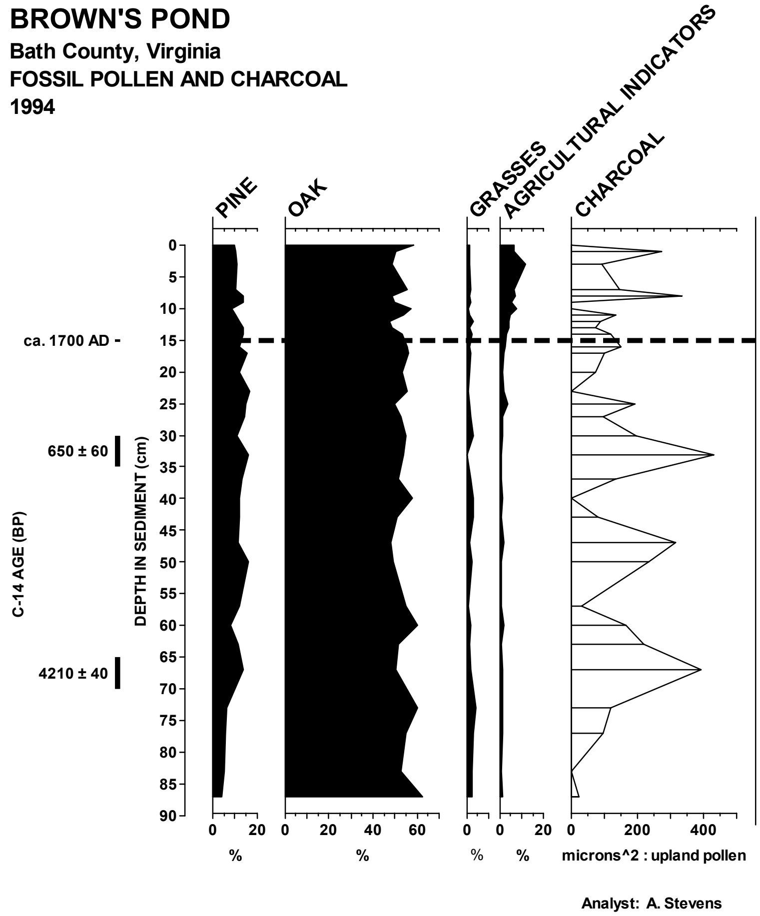

Figure 2.—Pollen and charcoal profiles for Brown's Pond, Bath County western Virginia. The date of local settlement by Europeans (dashed line) is from historical accounts.

35-cm-long sediment core obtained in 1994 represents approximately the last 900 years. Oaks (40 percent of total pollen) dominated the uplands prior to European settlement about 250 years ago. Pines, which like oaks, produce abundant pollen, were less important, and charcoal values were somewhat lower (Ch:P = 0-300) than observed at Brown's Pond. One or more major fires apparently burned near the pond about 150 to 250 years ago, with charcoal abundance increasing to 400 to 600 before declining in recent sediments (Fig. 3). The increase in fire activity was followed by an increase in pine and a decline in oak. Although oaks remain an important component of the modern vegetation, recent pine pollen percentages are more than double those prior to settlement.

Results from Brown's and Green ponds suggest that on mesic soils in protected locations of the southern Appalachians, light surface fires may contribute to maintaining oaks as a dominant component of mixed mesophytic forests—perhaps by selecting against more shade tolerant but less fire tolerant mesophytic hardwood species. On exposed sites with poorer soils where shade-tolerant (mesic-site) species compete less well (e.g., at Green Pond), oaks may persist with less fire. More frequent fires, at least some of which are likely to be more severe on shallow soils, may favor the establishment of pines and increase their importance on the landscape. However, longer lived oaks still persist, even in the face of competition from dense shrub understories, which may prevent pines from regenerating without fire (Patterson and Stevens 1995).

South Coastal New England and Long Island

Deep Pond lies at an elevation of 28 feet on the Ronkonkama Moraine in northeastern Suffolk County, Long Island, New York (40° 56′ 11″N, 72° 49′ 52″W). The uplands surrounding the pond are dominated by black and white oak and pitch pine. Soils are deep, loamy sands. A sediment core spanning the past 2,200

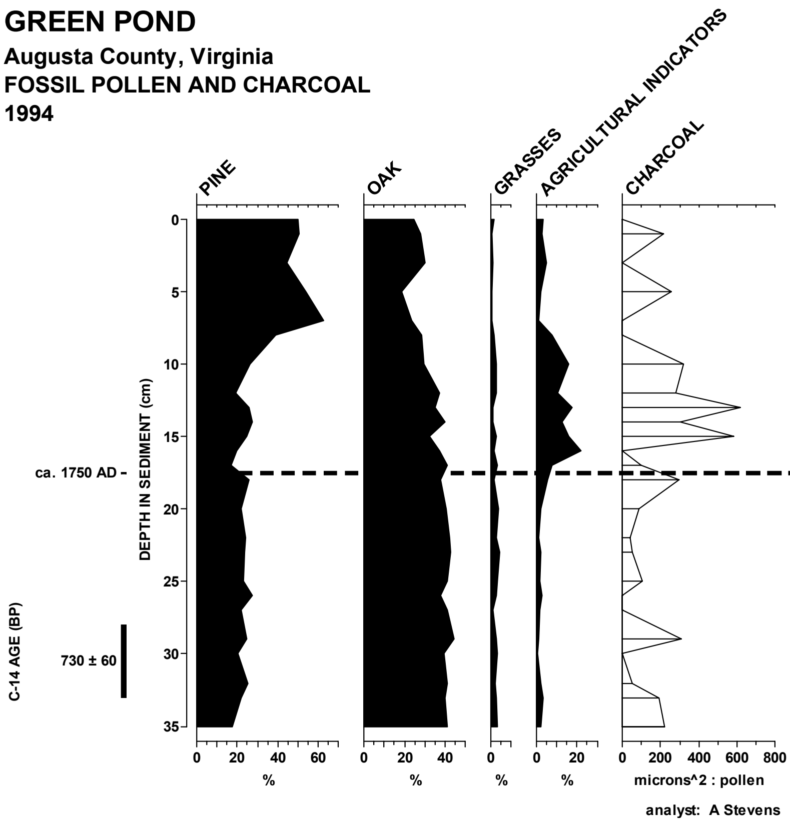

Figure 3.—Pollen and charcoal profiles for Green Pond, Augusta County, western Virginia. The date of local settlement by Europeans (dashed line) is from historical accounts.

years was recovered in 1981 (Backman 1984). Oaks dominated throughout the sample period, though percentages have declined somewhat, with pitch pine increasing in importance since European settlement about 300 years ago (Fig. 4). Charcoal values are high, averaging 400 to 600 before settlement and increasing to 500 to more than 1,000 in recent centuries. As at Green Pond in Virginia, recent increases in fire activity appear to favor pitch pine.

Charge Pond, at an elevation of 66 feet in Plymouth County, Massachusetts (41° 48′ 58″N, 70° 40′ 31″W), occupies a kettle hole in outwash of the Wisconsin glacial stage. Course, sandy soils currently support a mixture of pitch pine and scrub oaks. The area around the pond burned most recently in a catastrophic fire in 1957. Backman (1985) described the fire and vegetation history for the past several hundred years based on his analysis of a core obtained in 1981 (Fig. 5). Fire activity has been high throughout the last several hundred years, as evidenced by charcoal values which rise from 500 to 1,000 before European settlement to 4,000 to 5,000 during the post-settlement period. Oaks and white pine (*Pinus strobus*) were dominant before settlement. Because tree oaks are more likely to occur with white pine, we assume that white and black oak were more important than scrub oaks (Patterson and Backman 1988). Today, the vegetation on the upland is referred to as pitch pine-scrub oak barrens, but our data suggest that these species—especially pitch pine—have been dominant only since fire activity increased following European settlement.

Data from Deep and Charge ponds suggest that on course-textured soils, where pines are better able to compete with oaks, oaks survive at moderately high fire activity, but composition has shifted towards an increasing importance of pitch pine with the high fire activity during the post-settlement period. A more detailed examination of the pollen data suggests that with increasing fire activity, more mesic hardwood species, including hickory (*Carya* spp.), maple and beech, decline in importance.

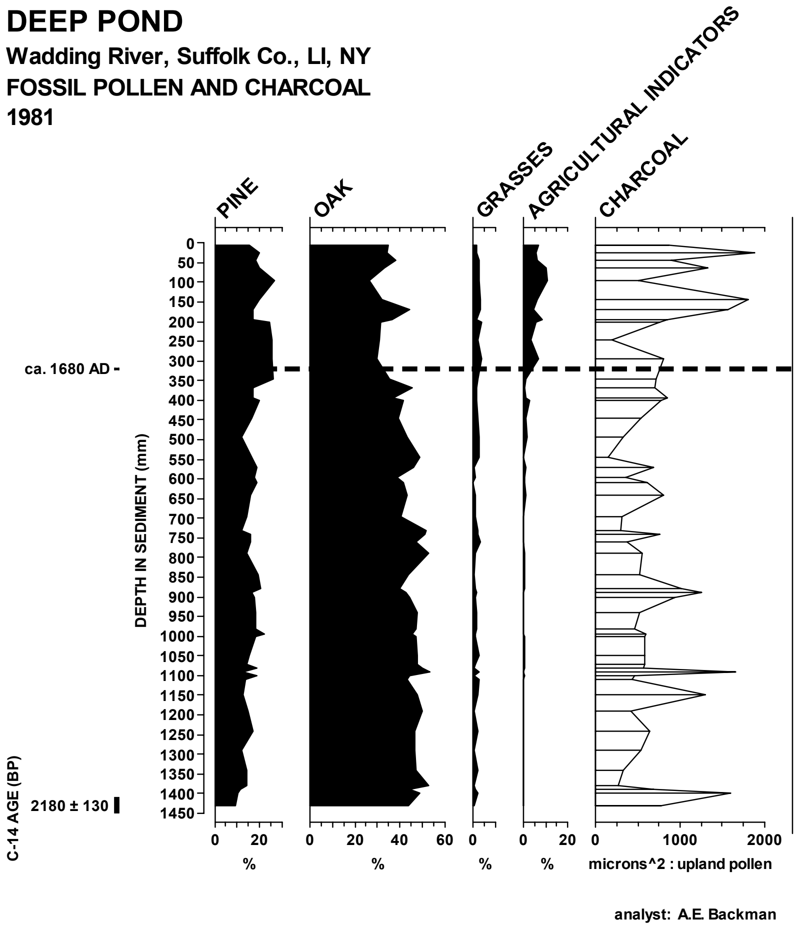

Figure 4.—Pollen and charcoal profiles for Deep Pond, Suffolk County, Long Island, New York. The date of local settlement by Europeans (dashed line) is from historical accounts.

Coastal Maine

Mount Desert Island in Hancock County, Maine, is a 120,000-acre mountainous, ocean island 40 miles south of Bangor. Diagrams for two small ponds provide insights about the interaction between fire and oaks near their northern range limit. Only red and bear oak (*Quercus ilicifolia*) occur naturally on the island today.

Sargent Mountain Pond is at an elevation of 1,145 feet (44° 20′ 04″N, 68° 16′ 10″W). Mature spruce (*Picea* spp.) forests on shallow-to-bedrock, granitic soils surround the small mountain pond, which lies just outside the western boundary of the great Bar Harbor Fire, which burned nearly 20,000 acres on the Island in October 1947. Oaks are not present in the pond's small watershed but there is a stand of red oak regenerated by the 1947 fire about 1 mile to the north. Pollen and charcoal analysis of a 30-cm-long sediment core provides information on the fire and vegetation history of the watershed for the past 400 years (Fig. 6). At least two fires burned the surrounding upland, one about 1700 AD, 60 years before the first permanent settlement on the Island at Somesville 2 miles to the west, and a second in the mid-19th century. The second fire probably occurred in conjunction with the logging of the mountain's virgin stands of spruce. Charcoal at background levels of less than 200 μ^2 for the rest of the core shows that no other fires have burned within the watershed during the period of record. The two discrete fires separated by 150 years allow interpretation of post-fire succession—alder (*Alnus* spp.), followed by birch, and finally spruce—that is consistent with what has been observed following early and mid-20th century fires (Patterson et al. 1983; Patterson and Backman 1988). Oak pollen occurs at generally less than 5 percent and rises only slightly following fires, perhaps reflecting the increased importance of regional pollen following stand-replacement fires on the watershed.

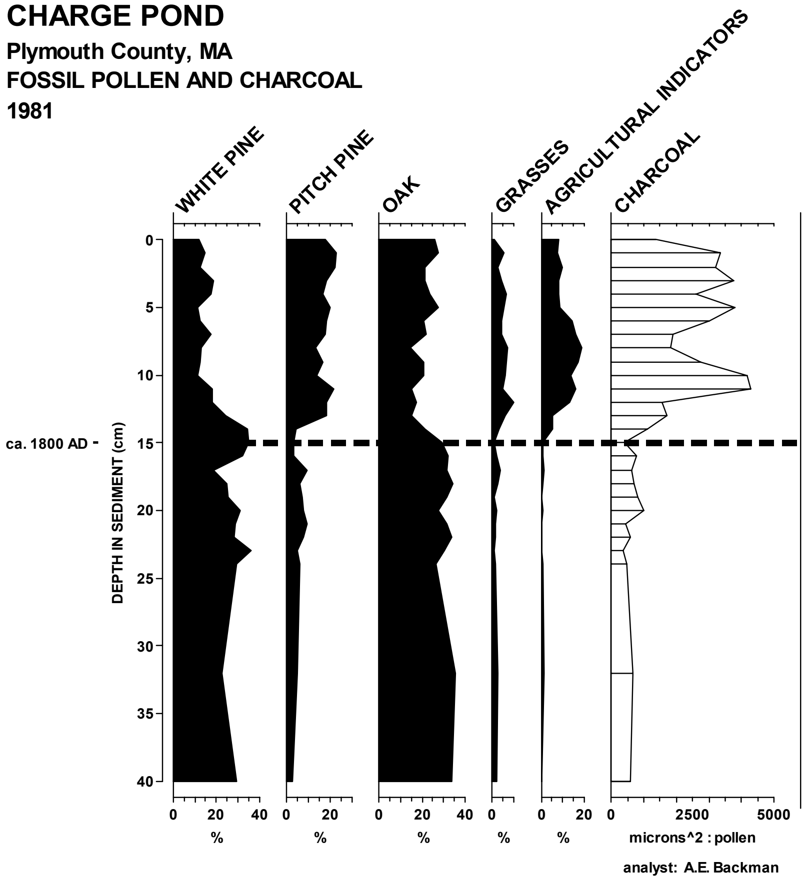

Figure 5.—Pollen and charcoal profiles for Charge Pond, Plymouth County, Massachusetts. The date of local settlement by Europeans (dashed line) is from historical accounts.

Lake Wood, at a lower elevation (35 feet) and 6 miles northeast of Sargent Mountain Pond (44° 24′ 27″N, 68° 16′ 06″W), lies in a glacially scoured valley with soils derived from course glacial debris overlain in places by marine clays. The entire watershed of Lake Wood burned in the 1947 fire. The vegetation today is dominated by paper (*Betula papyrifera*) and gray birch (*B. populifolia*), red (*Pinus resinosa*) and white pine, and occasional red oaks. A small stand of hemlock in the southeastern part of the watershed regenerated following the 1947 fire. Hemlock is uncommon in the forests of Mount Desert Island.

Like Brown's Pond in western Virginia, we have a long record of fire-vegetation interactions from Lake Wood. Unlike the Virginia core, the Lake Wood core was sampled and analyzed with very fine resolution; sample-to-sample precision is on the order of 20 to 40 years. The pollen data suggest oak has been present as at least a minor component of the local vegetation for much of the past 6,000 years despite substantial changes in species dominance. Hemlock, white pine, birch, sugar maple, and beech (not shown in Fig. 7) dominated until about 2,000 years ago when spruce (plus cedar and fir) replaced the hemlock and northern hardwoods. The charcoal record clearly documents the local occurrence of the 1947 Bar Harbor fire at the top of the profile. This fire occurred during the driest month in more than 150 years of recorded weather history in Maine (Baron et al. 1980), and may have been the most severe fire to occur on the watershed during the entire 6,000 year period represented by the core. However, different fire regimes appear to dominate at different times, and the abundance of oak varies with them. Approximately 3,000 to 6,200 BP as many as five or six charcoal peaks occurred, usually in conjunction with changes in the abundance of hemlock. We are investigating whether these fires followed declines in hemlock caused by insects, disease, or blowdown, or themselves were the cause of the declines (unpublished data), but oaks are

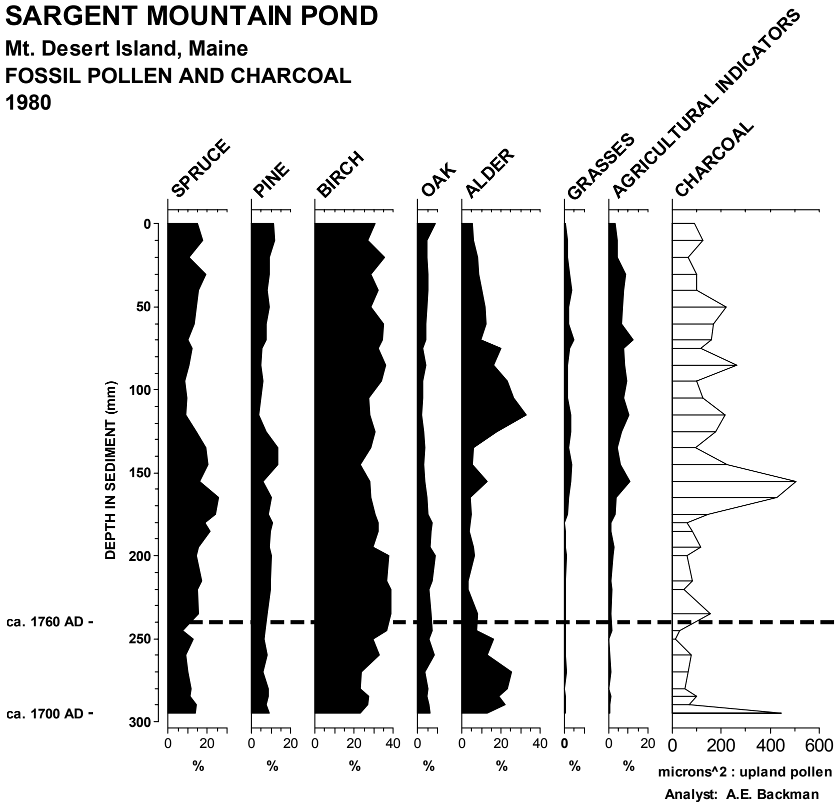

Figure 6.—Pollen and charcoal profiles for Sargent Mountain Pond, Hancock County, Maine. The date of local settlement by Europeans (dashed line) is from historical accounts and Lead-210 analysis.

clearly more abundant during this period than during the subsequent 1,000-year period (2,000 to 3,000 BP) with no fires. The shift from hemlock to spruce in the region at 2,000 BP is accompanied by an overall increase in fire activity, but oak does not increase again until well after this change in vegetation (and fire regime). Oak pollen percentages at the surface of the core are higher (at 11 percent) than at any time since about 3,700 BP.

In conjunction with a companion site (The Bowl) on the Island for which we have comparable detail for an even longer period of time (unpublished data), the detailed sampling of the Lake Wood core over a long period provides us, with the opportunity to seek answers to the questions posed earlier: Do changes in fire regimes cause changes in oak abundance? Do the increased flammability of oak fuels cause more frequent fires on the landscape? Our analyses to answer these questions are incomplete but the evidence for this one site where fire has historically been infrequent on the landscape suggests at least a correlation between fire activity and oak abundance.

SUMMARY AND CONCLUSIONS

Paleoecological evidence, such as the fossil pollen and charcoal data presented here, is incomplete, fraught with uncertainties, and dependent on assumptions that often are difficult to evaluate. But they are the only data available that allow us to make inferences about fire and vegetation before the time of recorded history. There is value in considering the long-term interactions between fire and vegetation, especially for long-lived forest trees like the eastern oaks, because processes like fire, which influence stand initiation and development, have secondary effects that are realized over decades or even centuries. Changes in fire return intervals, the interaction between fire occurrence (and effects) and climate (which itself changes), and the rare occurrence (in some systems) of catastrophic fires, must be studied over periods far longer than fire scientists are accustomed to contemplating.

In this paper I have examined the value of using paleoecological data to investigate the response of oaks as a group to fire. Chief among the difficulties is that

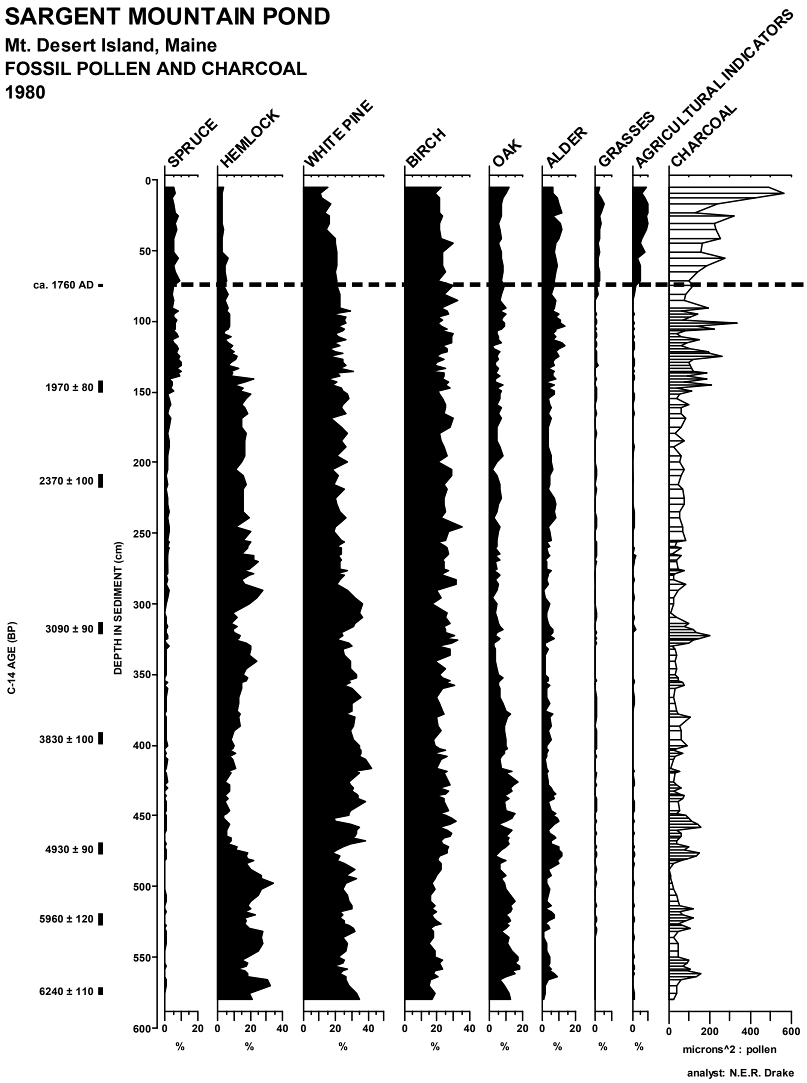

Figure 7.—Pollen and charcoal profiles for Lake Wood, Hancock County, Maine. The date of local settlement by Europeans (dashed line) is from historical accounts.

we cannot distinguish among any of the dozens of oak species based on their pollen morphology. Similarly, we often can only guess at how changes in the abundance of charcoal in sediments relate to the intensity, severity, or extent of individual fires. And it is only under unusual circumstances that we can identify individual fires and their effects on vegetation using fossil data. The detailed analyses for the Sargent Mountain Pond and Lake Wood sediments are unique in this regard.

Yet fossil data do provide information that may be useful in the future management of oak forests. Chief among the lessons we have learned is that it is difficult to generalize about the interaction between fire and oaks on the centuries-long time scales that are relevant to forest stand development and change. Oaks have persisted for thousands of years in a fire regime apparently characterized by relatively infrequent, low-intensity fires at Brown's Pond in western Virginia. But oaks also have persisted for millennia under frequent, probably higher intensity surface fires at Deep Pond on Long Island. At Green Pond, surrounded by more xeric soils and vegetation at higher elevations in Virginia, one or more severe fires at the time that Europeans arrived changed the balance between oaks and pines. The same is true for Charge Pond in southeastern Massachusetts. In neither case were oaks reduced to minor components of the vegetation, but the balance between oak and pine was clearly shifted toward the later. On the Maine coast, near the northern limits of the range of oaks, fire at times has been rare, and oaks have been a minor component of the vegetation as a whole. Yet when fire return intervals are reduced (i.e. fires occur more often), oaks seem to compete better with more shade-tolerant conifers and hardwoods.

In the Upper Midwest, Jacobson (1979) found that white pine migrated into oak forests, perhaps in response to a “slight decrease in fire frequency” at various times during the Holocene. This finding is counter to those for the Green and Charge ponds in the East. Jacobson did not provide charcoal data to support his hypothesis, but it is likely that fires were so frequent in the oak savannah stands he postulates that pines would not have been able to survive. White (1983) found that annual burning maintains oak savannas with grassy understories in prairie-forest transition stands in east-central Minnesota. Jacobson assumed that white pine cannot tolerate return intervals of less than 20 years.

There is little evidence from sites on the East Coast that fires were frequent enough to give rise to savannas like those of the Upper Midwest. At none of the sites I have discussed has an increase in charcoal, either before or after European settlement, coincided with a substantial increase in grass pollen. Stevens (1996) suggested that frequent Native American burning on the island of Martha’s Vineyard, Massachusetts, may have been responsible for the coincidence of high charcoal values and high oak and grass pollen percentages. She also found abundant charcoal with abundant oak and grass pollen in post-settlement sediments, but it may have been grazing and cultivation as much as fire that was responsible for this correlation, which was largely restricted to coastal sites regionally (Foster and Motzkin 2003).

Recent work at the Harvard Forest emphasized the variety of opinions on the importance of interactions between fire and oaks in the Northeast. Parshall and Foster (2002) generally discountrd the importance of fire relative to climate and soil factors as influencing regional variation in the vegetation of the oak region of southern New England. But Foster et al. (2002) postulated that fire is important, albeit at lower levels than along the coast, in maintaining oak and chestnut forests in central Massachusetts. Their work shows that detailed reconstructions of fire and vegetation histories at selected sites hold the promise of further advances in our understanding of the long-term relationship between fire and oaks. However, our interpretation of historical data must be informed by a knowledge of fire-vegetation interactions from contemporary studies. Long-term prescribed burning experiments, like those conducted for nearly half a century at the Tall Timbers Research Station in Florida (Stoddard 1962), will increasingly provide information that will allow us to better interpret the paleoecological record.

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FIRE SCARS REVEAL VARIABILITY AND DYNAMICS OF EASTERN FIRE REGIMES

Richard P. Guyette[1†], Daniel C. Dey[2], Michael C. Stambaugh[1], and Rose-Marie Muzika[1]

Abstract.—Fire scar evidence in eastern North America is sparse and complex but shows promise in defining the dynamics of these fire regimes and their influence on ecosystems. We review fire scar data, methods, and limitations, and use this information to identify and examine the factors influencing fire regimes. Fire scar data from studies at more than 40 sites in Eastern North America document fire regimes in forests with oak. Fire frequency was highly variable in both time and space even at regional scales (less than 500,000 ha). Many sites burned frequently (2- to 3-year mean fire intervals) while nearby sites (less than 40 km distant) burned infrequently (mean fire intervals more than 20 years). The fire scar record shows that major factors controlling temporal differences in fire regimes are changes in human population density, culture, and annual drought. Spatial differences in fire regimes are influenced by regional temperature, human population density, and topographic resistance to the spread of fire. Severe fire years (more than 10 percent of trees scarred at the sites) were associated with strong regional droughts that covered most of the Eastern United States and southern Ontario, Canada. Major fire years in Eastern North America occurred about 3.6 times per century before suppression efforts in forests with an oak component. Fire regimes with numerous human ignitions were more influenced by droughts. We synthesize mean fire intervals during the pre-European settlement period using an empirically derived regression model. The model was developed using two variables to predict broad scale spatial differences in fire frequency based on fire interval data derived from dendrochronologically dated fire scarred trees. Sixty-three percent of the variance in mean fire intervals was explained by mean maximum temperature and 12 percent by mapped human population density and historical documentation. The model is used to map coarse scale fire intervals in forested regions of the Eastern United States.

INTRODUCTION

Wildland fire is an important disturbance influencing oak forests (Johnson et al. 2002; Wade et al. 2000). Changing forest composition and fire risk have given the history of wildland fire regimes new relevance. Owing to the longevity of trees, forest composition often changes slowly in response to fire and at time scales well beyond the lifespan of single human observers. To understand how and why forests are changing, we often can benefit from a longer-term perspective. Dendrochronology can provide this perspective with high resolution spatio-temporal data for periods of more than 300 years (several generations of trees) in Eastern North America. This period spans changes in human societies from subsistence to fire suppression. Tree-ring dating of fire scars and climate reconstructions from the width of tree rings provide long records of fire dates, fire locations, and drought. We examine fire history in forests of Eastern North America during the last three centuries, emphasizing forests that have an oak (*Quercus* spp.) component. We document and analyze fire regimes using dated fire scars and examine variables that are hypothesized to influence these fire regimes. Fire scar data from more than 60 sites in 16 states and the province of Ontario (Table 1) are used to document fire regimes and examine their temporal and spatial variability. To present a more balanced regional analysis and overview, we excluded a considerable amount of data from Missouri (more than 20 fire history sites).

Dendrochronological analysis of fire scars and the reconstruction of fire history from scar data have many positive and precise features as well as limitations. Fire scars on oaks provide a detailed record of fire events, such as exact locations and dates (Guyette and Stambaugh 2004; Smith and Sutherland 1999). Each fire scar potentially has a spatial resolution of less than 1 m and a temporal resolution of less than one year. These characteristics generally set this method apart from other

[1]School of Natural Resources, University of Missouri, Columbia, MO 65211; [2]USDA Forest Service, Northern Research Station, Columbia, MO 65211; [†]Corresponding author, 573-882-7741, email: guyetter@missouri.edu.

Table 1.—Mean fire intervals from fire scar studies in eastern North America (NH = northern hardwoods, CH = Central Hardwood, SP = Southern Pine-Hardwoods, PT = Prairie-Forest Transition; na = not available)

Site name	State	Forest Type	Trees and forest type	Pre-Euro-fire interval	Post- Euro-fire interval	Site data reference
Green Mts.	VT	NH	red pine forest	na	18	Engstrom and Mann 1991
Basin Lake	ON	NH	pine-oak woodland	21	na	Guyette and Dey 1995a
Papineau L.	ON	NH	oak-pine forest	29	7	Dey and Guyette 2000
Opeongo	ON	NH	pine-hardwoods	29	25	Guyette and Dey 1995b
Bracebridge	ON	NH	red oak forest	11	na	Guyette and Dey 1995c
Joko	ON	NH	pine hardwoods	13	na	Dey and Guyette 2000
Seagan	ON	NH	pine hardwoods	17	na	Dey and Guyette 2000
Sault Ste. Marie	ON	NH	pine hardwoods	26	73	Alexander et al. 1979
Big Bay	MI	NH	pine- hardwoods	25	na	Torretti 2003
Itasca SP	MN	NH	conifer hardwood	25	36	Spurr 1954
Itasca SP	MN	NH	conifer-hardwood	29	13	Clark 1990
Itasca SP	MN	NH	conifer-hardwoods	30	9	Frissell 1973
Costal sand	MI	NH	costal pine forests	18	88	Loope and Anderton 1998
Upland sand	MI	NH	upland pine forests	23	29	Loope and Anderton 1998
Savage Mt.	MD	CH	oak forest	8	8	Shumway et al. 2001
Oreton	OH	CH	white oak forest	na	4	Sutherland 1997
Pike Knob	WV	CH	white oak forest	na	14	Schuler and McClain 2003
Lemm Swamp	TN	CH	oak woodlands	5	8	Guyette and Stambaugh 2005
Saltwell	TN	CH	oak woodlands	6	8	Guyette and Stambaugh 2005
Richland Creek	TN	CH	oak woodlands	5	4	Guyette and Stambaugh 2005
Huckleberry	TN	CH	oak woodlands	6	4	Guyette and Stambaugh 2005
Boone Barrens	IN	CH	oak savanna	19	5	Guyette et al. 2003
Brush Mt.	VA	CH	pine – hardwood	na	8	Sutherland et al. 1992
Mill Hollow	MO	CH	oak-pine forest	6	2	Guyette and Cutter 1997
MOFEP 4	MO	CH	oak –pine forest	3	na	Guyette and Stambaugh 2004
MOFEP 5	MO	CH	oak –pine forest	4	na	Guyette and Stambaugh 2004
Mahans Crk.	MO	CH	oak –pine forest	3	4	Guyette and Cutter 1997
Mill Creek	MO	CH	oak-pine forest	4	4	Guyette and Cutter 1997
Hartshorn	MO	CH	oak-pine woodland	3	2	Guyette and Cutter 1997
Shannondale	MO	CH	oak forest	13	4	Guyette and Cutter 1997
Gee Creek	AR	CH	oak-pine forest	15	2	Guyette and Spetich 2003
Gobblers Knob	AR	CH	oak-pine forest	16	2	Guyette and Spetich 2003
Vermillion	MO	CH	pine-oak forest	3	na	Guyette 1997
Rd1645	MO	PT	oak woodland	4	8	Cutter and Guyette 1994
Pleasant Prairie	WI	PT	oak savanna	na	7	Wolf 2004
Ava Glades	MO	PT	redcedar-oak glades	6	11	Guyette and McGinnes 1982
White Ran.	MO	PT	oak woodland	4	8	Dey et al. 2004
Caney Mt	MO	PT	oak savanna	5	7	Guyette andCutter 1991
Loess Hills	MO	PT	bur oak woodland	8	4	Guyette and Stambaugh 2005
Granny Gap	AR	SP	pine–oak forest	5	4	Guyette and Spetich 2003
Chigger Rd	AR	SP	pine-oak forest	13	2	Guyette and Spetich 2003
GSMNP	TN	SP	pine – oak forests	na	13	Harmon 1982

Table 2.—Methods of documenting fire history, their data acquisition and characteristics

Method	Sources	Resolution	Time depth	Limitations and advantages
Dendrochronology	Crossdated fire scars from trees, remnant wood. Also ring widths, stand age	Seasonal to annual with high precision	Commonly two to four centuries, rarely more	Stand replacement fires not well documented, high resolution in time and space
GLO notes	Historical archives	None	About 150 to 200 years ago	Inferential, spatially explicit information
Charcoal sediment	Lakes, peat	Variable, decadal to centuries	Centuries to 12,000+ years	Local sources and low resolution, long-term data
Written accounts	Historical literature and travel accounts	Low to daily in journals	About 500 years	Nonquantitative, often biased, includes cultural information
Alluvial charcoal	Alluvial sediments	About 500 years or the limits of carbon dating	1000 to 20,000+ years	Low resolution, record of large, intense, sediment-producing fires

fire history methods (Table 2). However, a number of problems may arise when developing and interpreting the fire scar record. A woody growth increment without a scar is only weak evidence of "no fire" because many trees are resistant to scarring. In Eastern North America, challenges often arise when constructing fire scar histories, such as the limited availability of preserved fire scars, limited abundance of closely spaced recorder trees, differences in record length, and sampling methods that are defined by the limits of the resource. However, even studies with the most severe sampling problems rise above the alternative: an absence in our knowledge of fire history. Problems in developing fire history from fire scars (and how we address them in this paper) include:

1. *Problem:* Many trees have thick bark and are not scarred in fires. Thus, the lack of a scar does not mean the absence of fire. *Solution:* Minimize the use of studies based only on a single or several trees.

2. *Problem:* Stand replacement fires are not included in fire scar studies because all fire scars occur on survivor trees. *Solution:* Infer from the number of trees scarred when larger and more severe fires occurred.

3. *Problem:* Tree species differ with respect to recording fire history. This can result in errors when comparing histories. *Solution:* Sample a sufficient number of trees to develop a composite fire interval that reflects the actual frequency of fire in an area.

4. *Problem:* Spatial extent of a study site influences the inferred fire frequency. *Solution:* Confine analysis to sites of several hundred hectares (less than 4 km^2)

5. *Problem:* The number of sample trees in any given year may affect the number of fires detected. *Solution:* Truncate data from early periods when there were few recorder trees in the record.

6. *Problem:* Many fire scar chronologies have long, open-ended intervals that cannot be interpreted as true intervals. *Solution:* Estimate mean fire intervals as years per fire.

The goals of this review and synthesis are to examine the long-term and large-scale factors that affect the spatial and temporal variability of fire regimes in Eastern North America. Specific objectives are to examine climate,

Table 3.—Number of fire scar history studies and sites by Ecoregion

Ecoregion	Forest type	Studies	Sites
Warm continental	Northern hardwoods	13	17
Hot continental	Central hardwoods	25	60
Subtropical	Southern pine hardwoods	1	1
Prairie	Forest prairie transition	6	6

human, and landscape influences on fire regimes using data from existing fire scar histories in the eastern oak forests. We present a preliminary synthesis via regression modeling, using the effects of temperature and human population on fire regimes.

REVIEW

Numerous studies (Table 1) have used fire scar data to reconstruct fire history from the four major oak ecoregions and forest types of Eastern North America (Bailey 1997; Johnson et al. 2002). The number of quantitative fire scar histories varies widely among ecoregions (Table 3). Both a lack of scientific inquiry and the scarce availability of datable fire scarred materials contribute to this disparity. Northern Hardwood studies are located in Minnesota (Frissell 1993; Clark 1990; Heinselman 1973; Spurr 1954), Michigan (Torretti 2003), Ontario (Dey and Guyette 2000; Cwynar 1977), and Vermont (Engstrom and Mann 1991). Central Hardwood studies are located in Missouri (Guyette and Kabrick 2003; Guyette et al. 2002; Batek et al. 1999; Guyette and Cutter 1997), Indiana (Guyette et al. 2003), Tennessee (Guyette and Stambaugh 2005; Harmon 1982; Armbrister 2002), Ohio (Sutherland 1997), West Virginia (Schuler and McClain 2003), Virginia (Sutherland et al. 1992), and Maryland (Shumway et al. 2001). Although there are studies for the southern pine-oak region (DeVivo 1991, Van Lear and Waldrop1989), few are fire history studies from fire scars. There are three fire scar sites on the edge of the southern pine-oak region in the Lower Boston Mountains of Arkansas (Guyette and Spetich 2003). Forest-prairie transition sites are on the edge of the Great Plains (Guyette and Stambaugh 2005), the oak savannas of Wisconsin (Wolf 2004), the oak woodlands and savannas of south-central Missouri (Cutter and Guyette 1994; Guyette and Cutter 1991; Dey et al. 2004), and redcedar glades of southwestern Missouri (Guyette and McGinnes 1982).

The Warm Continental-Northern Hardwoods region has several robust fire histories, though most are in forest types with moderate to low oak abundance. Several of these were not based on fire scars or derived from precisely crossdated tree rings (Whitney 1986; Heinselman 1973). Nine studies in this region had an average mean fire interval (MFI) of 23 years, with a range from 11 to 33 or more years before European settlement (ca. 1650 to 1850).

The Hot Continental-Central Hardwoods region is fairly well represented, especially in its western half. Here, many studies indicate a high frequency of burning in oak and oak-pine forests. Fifteen studies had an average MFI of 8 years, with a range from 3 to 21or more years before European settlement. However, in some areas of the Ozarks, high variability in fire frequency can be found within small extents (e.g., 3 km) (Guyette et al. 2002).

Unfortunately, little quantitative fire scar data are available for the Subtropical-Southern Pine-Hardwoods (Table 3). For this review we used sites (Guyette and Spetich 2003) on the edge of this ecoregion in Arkansas. Three upland forest sites in Arkansas had an average MFI of 11 years, with a range from 5 to 16 or more years before European settlement. The topographic roughness and low population density of these sites during this period probably do not represent the region as a whole. However, the southern pine-oak region has a number of species and ecosystems with potential fire scar data, and represents excellent future opportunities for documenting fire history. “Yellow pine” species often have resinous wood that preserves well for at least a

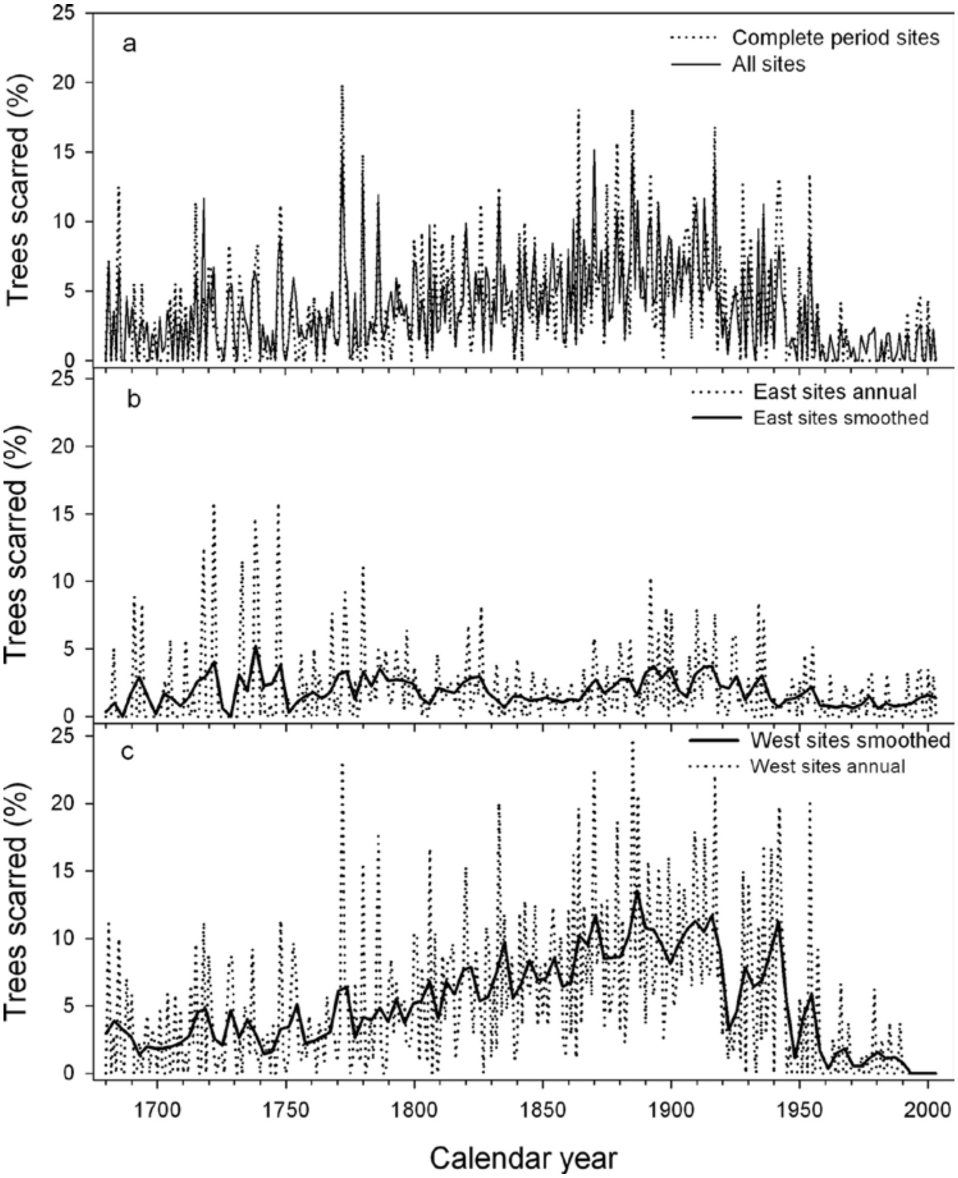

Figure 1.—Time series of percent trees scarred at sites in or near forests with an oak component in Eastern North America. The "complete period site" chronology (a, doted line) is compiled from fire scar chronologies that cover the whole or nearly the whole time period. The "all sites" chronology (a, solid line) is compiled from all fire scar chronologies regardless of length. The eastern chronology (b) is from sites in Ontario, Maryland, Indiana, Tennessee, West Virginia, Virginia, and Ohio. The western chronology (c) is from Arkansas, Missouri, Wisconsin, Minnesota, and Michigan. See Table 1 for Eastern North American site references.

century after the death of the tree. Live and recently dead oaks of the white oak group also can provide excellent fire histories.

The forest-prairie transition region is represented by several studies in Missouri and Wisconsin, that focus on conditions from oak woodlands and savannas to cedar glades. Five sites in the forest prairie transition had an average MFI of 5 years, with a range from 3 to 8 years or more years before European settlement.

The fire scar record (Fig. 1a-c) as indicated by the studies cited, provides the timing and magnitude of fire events that can be linked to climate. However, the timing of annual fire dates is less than exact within a given year due to the nature of tree growth and fire scars. Fires

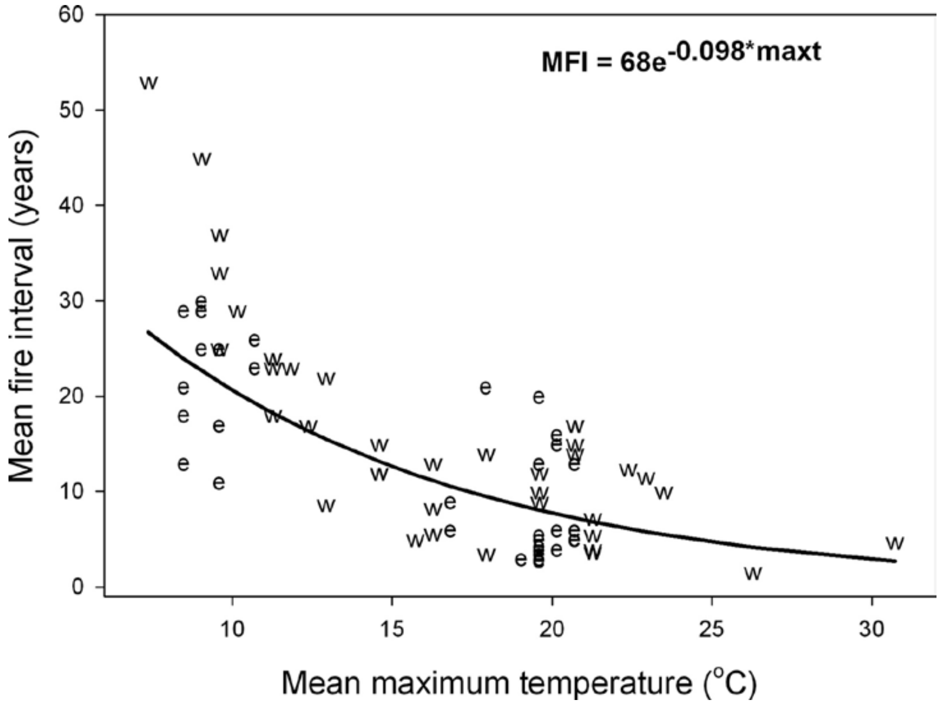

Figure 2.—Mean fire intervals (MFI) predicted from proxy mean maximum temperature (maxt). Mean fire intervals are for the pre-Euro American period (ca.1650 to 1850). Temperature is the average annual mean maximum temperature for 1971 to 2000 minus 0.4 ºC for North American warming. Eastern North American sites are denoted by "e:" while western North American sites are denoted by "w"; fire scar data are from 79 sites with more than 1,500 trees and 10,000 fire scars. See Table 1 for Eastern North American (e) site references and the Model Development section of this paper for western North American (w) site references. The equation in the upper right describes the regression line.

can be dated to the early, mid, and late parts of the growing season or to the dormant season (generally October through April). Fire scar evidence from eastern forests indicates that fires in oak ecosystems occur most frequently during the dormant season of tree growth. Of the 60 or more study sites in eastern North America in which the authors have been involved, about 95 percent of the fire scars occurred between annual rings during the dormant season. This probably results from surface fuel characteristics such as the quantity of leaf litter, its density and arrangement, and its potential moisture content all of which increased during leafoff and can lead to more pyrogenic conditions. The season of burning is greatly restricted by primary fine fuels as the controller of fire propagation. Surface fuels in deciduous forests have more and potentially drier litter when the leaves have fallen. Full summer canopies mitigate the loss of litter moisture and fuel temperature by decreasing solar exposure.

Factors influencing fire intervals

Spatial and Temporal Climate Differences

Climate is a pervasive factor in all fire regimes and has complex effects at multiple spatial and temporal scales. Spatially, we address regional climate differences as well as the effects of year-to-year climate variability on fire history sites. The duration of fuel conditioning, the duration of snow cover, the length and intensity of the warm season, and primary productivity are tied to temperature and precipitation. The correlation between annual mean maximum temperature (proxy period: 1971 to 2000) and pre-European settlement mean fire intervals at 38 sites (Table 1) is significant ($p < 0.001$) and strong ($r = -0.73$, Fig. 2), indicating that temperature likely influences many biotic and abiotic factors that control fire regimes. Precipitation also was significantly correlated ($r = -0.56$, $p < 0.01$) with fire intervals. The negative correlation value reflects the association of shorter intervals between fires occurring at sites with higher amounts of precipitation. In some ways this is counterintuitive since wetter conditions can create shorter fire windows and higher fuel moisture; however, increased precipitation generally leads to higher primary productivity and potential more rapid fuel accumulation. Thus, regional climate parameters, particularly mean maximum temperature and precipitation, may have a major influence on fire regimes.

Temporally, the effects of yearly to monthly differences in climate, particularly drought, are obvious and important to fire occurrence and fire regimes. Equally important in eastern deciduous forest ecosystems are

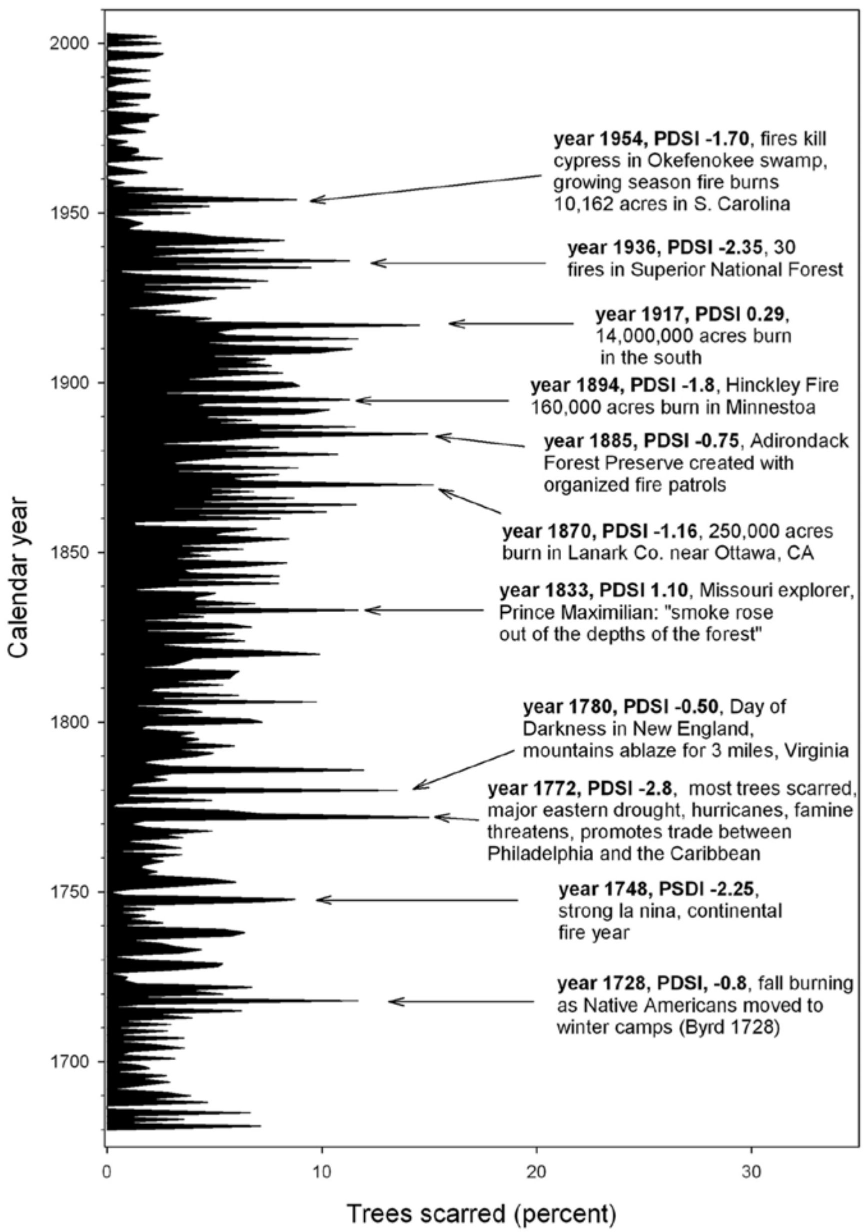

Figure 3.—Percentage of trees scarred at sites with fire scar chronologies in Eastern North American, illustrating the connection between the fire scar record and historical documentations of fire history. PDSI is the reconstructed Palmer drought severity index (Cook et al. 1999). Negative values denote increasing drought.

conditions during leafoff, particularly spring and fall season drought. Our preliminary correlation analysis of drought and fire extent and severity (percent trees scarred) shows that annual reconstructed drought indices (Cook et al. 1999) are weakly ($r < 0.2$) but significantly ($p < 0.05$) associated with the percentage of trees scarred at various sites in Eastern North America, particularly those in the western portion. Fire occurrence and percent trees scarred in eastern oak forests were only weakly related to reconstructed annual drought over the past three centuries. This probably results from limitations of drought reconstructions and their resolution, and asynchrony between fire occurrence (dormant season) and the bases of reconstructed drought (growing-season ring width). However, in years when many sites and trees were scarred in Eastern North America, there is evidence for widespread drought that continues through growing and dormant seasons (Fig. 3). Eight of ten of the largest fire years occurred during severe droughts. We judge by the percentage of trees scarred (years with twice the average percent trees scarred) that major fire years in Eastern North America occurred about 3.6 times per century before fire suppression efforts in forests with an oak component. These large fire years often are associated with historical documentation (Fig. 3) of smoke, wildfire, and fire prevention.

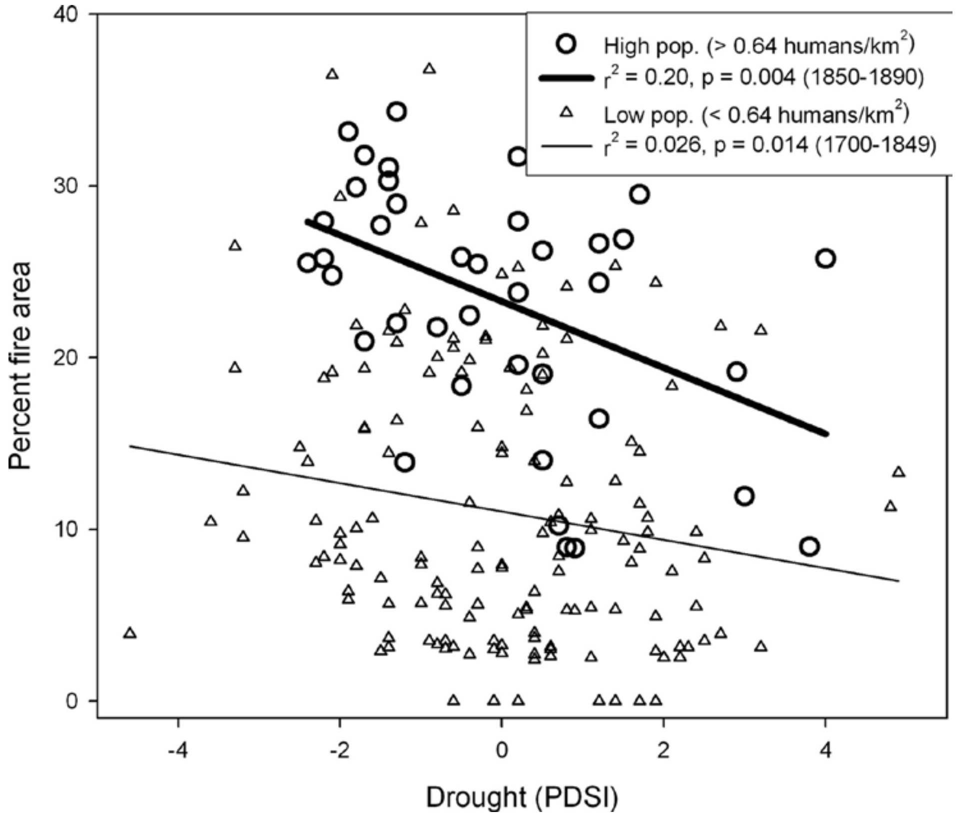

Figure 4.—Regression lines indicate the increased strength of the drought-fire relationship during periods of high versus low human population density. Fire data are from more than 2500 dated fire scars at 27 sites in the Current River region of the Ozark Highlands between 1700 and 1890. Fire area is based on the product of the percent of trees and the percent of sites burned annually. PDSI is the reconstructed Palmer drought severity index (Cook et al. 1999).

The fire scar record illustrates that drought-fire associations are not fully realized in the absence of numerous ignitions, as many severe droughts result in no fire events. Human population density influences the relationship between drought and wildfire by increasing ignitions. In fire regimes with few lightning ignitions, the effects of drought on fire occurrence often are unrealized until human ignitions become abundant. For example, in the Ozark Highlands, the extent of wildfire is moderately correlated ($r = -0.44$, $p = 0.004$) with drought during periods with abundant anthropogenic ignitions (1850 to 1890) but weakly correlated ($r = -0.15$, $p = 0.014$) during periods with low human population density (1700 to 1849) (Fig. 4). This supports the hypothesis that humans may have played a significant role in past fire regimes.

When interpreting fire scar records, a common assumption is that years with a higher percentage of trees scarred result solely from the effects of drought on fuels and scarring. However, it is possible that many past wildfires were the result of climate-conditioned fuels *and* numerous drought-inspired human ignitions. While one might expect an increase in fires owing to accidental ignitions during drought periods, there is no necessary causation between drought and the number of intentional fires. This theory is supported by consistent correlations between drought and arson documented by the modern fire record. For example, fire season drought is significantly correlated with the number of arson fires in the Eminence Fire Protection District (1970 to 1989) in the Missouri Ozarks ($r = -0.76$). On a larger scale, drought and the number of arson fires summarized by

Table 4.—Relationship between number of arson fires and drought by state (Drought data are the annual Palmer Drought Severity Index; the lower the value of this index, the more severe the drought; thus, the negative correlation between drought and arson; correlation coefficients in bold are significant at ($p<0.05$); fire data: U.S. Dep. Agric. 1940-1997)

State	Correlation (No. arson fires and drought)	Mean no. arson fires	Mean no. lightning fires	No. of years of data
Missouri	-0.39	1069	20	48
Arkansas	-0.61	1638	85	48
Illinois	-0.49	46	0.72	48
Indiana	-0.29	36	1.7	48
Tennessee	0.04	1148	12	48
S. Carolina	-0.36	1918	58	48
Florida	-0.42	2769	580	48

Table 5.—Effects of topographic roughness on mean fire intervals at increasing human population densities (topo-resistance refers to the resistance of a landscape to the propagation of fire); details on the calculation of topographic roughness indices found in Guyette et al. 2002; Peak mean fire intervals are for the first decades of most frequent burning

Site Group	Number sites	Topo-resistance	Roughness index	MFI at peak	Population at peak no./km^2	Year at peak
Boston Mts., AR	6	high	1.08	2.7	5.5	1875
Current River, MO	10	moderate	1.020	3.5	0.64	1850
Highland Rim, TN	4	low	1.0004	2.3	0.50	1770

year and state are significantly correlated (Table 4) (U.S. Dep. Agric. 1940-1997). Thus, during much of our history, and even during the recent period of societal pressure and legal penalties for arson, humans are still purposefully burning landscapes during droughts.

Topographic Resistance to Fire Propagation

The frequency of fire is a function of two types of ignitions: local and neighboring. For example, fires at a site 1 to 3 km^2 in area result from ignitions produced locally (new ignitions within the site) or from the propagation of ignitions into the site by a spreading fire. Thus, fire frequency at a site is both a product of local ignition and the resistance of the surrounding landscape to the propagation of fire. Part of this resistance is a function of topographic features such as hills, valleys, and bodies of water that disrupt the continuity of fuels (Guyette and Stambaugh, in press). For example, the highly dissected topography of the Current River watershed in Missouri has been shown to be related to the frequency of fires during periods of low human population density, when anthropogenic ignitions were limited in number (Guyette et al. 2002; Guyette and Dey 2000). In topographically rough terrain, low-intensity surface fires are impeded by steep slopes, discontinuous fuels, and variability in fuel moisture and fuel loading. In addition, topographic roughness likely mitigates the effect of increasing anthropogenic ignitions as many more human ignitions are required to maintain the same mean fire interval compared to less topographically rough regions (Table 5). Thus, at the time of Euro-American settlement, short fire intervals (1 to 3 years) in topographically rough landscapes occur at later dates as human populations increased. Before

1875, about 10 times as many humans were required to maintain a fire frequency of less than 3 years in the topographically rough Boston Mountains, Arkansas, as were required in the Oak Barrens of the Highland Rim in middle Tennessee (Table 5).

Fire and Culture

Of all of the cultural aspects affecting wildland fire, none have been as profound and effective as suppression efforts enabled by the modern technologies of ground and air transportation. Although grazing, landscape fuel fragmentation, intentional burning, and other cultural influences have been detected in fire scar records, these effects are not as pronounced as fire suppression in modern industrial societies. All but a handful of fire scar chronologies show a significant decline in fire frequency in the mid to late 20th century. The few fire scar chronologies that do not have a decline in fire frequency often are derived from private lands with owners who have a tradition of burning, or are in remote regions with long response times in fire protection (Jenkins 1997).

Before fire suppression, cultural values concerning wildland burning are not evident in the fire scar record and may be secondary to increasing anthropogenic ignitions as a function of human population density. In many environments, humans are a fire obligate species. Accidental ignitions, purposeful ignitions, machine fires, and debris burning are a few of the hundreds of human activities that can result in wildland fires. The sheer number of humans in an ecosystem may be more important than their cultural attitudes about the use of fire. Unlike fire frequency, which is tempered by fire suppression, fuel fragmentation, and reduction, the number of ignitions continues to rise with increasing human population.

The fire scar record suggests that *Homo sapiens* is a keystone species in many fire regimes. A keystone species influences the “environmental balance of an area or habitat” through a chain of events (e.g., trophic, reproductive, or abiotic modifications). Keystone species have a disproportionate influence on community structure in excess of their abundance. The fire scar record indicates that even at low population densities, humans had a profound effect on fire regimes and vegetation (Batek et al. 1999; Guyette and Kabrick 2003). The influence of a keystone species is derived from the ability to control the most competitive species (i.e., tree species). Temporally and spatially, fire scar frequency has been associated with the presence of humans. The absence of a keystone species releases highly competitive plants (trees) from the limiting effects of fire and allows them to exclude other plants and their herbivores. In forested regions, fire scar frequency increases rapidly as humans culture the forest for both wild and domestic large herbivores. The theory that humans are a keystone species in fire regimes may be especially relevant in Eastern North America, where climate windows suitable for burning often are short and the probability of natural ignitions is low. The fire scar record in Eastern North America is only weakly related to the abundance of lightning fires as documented later in this paper. Unlike lightning, humans can target their ignitions with respect to fuel conditions and location.

Population Density and Anthropogenic Ignition Rates

Since the entry of *Homo sapiens* into North America more than 12,000 years ago, humans have been the dominant ignition factor in many ecosystems. Although some have argued that there were not enough humans to impact flora and fauna at regional levels, the fire scar evidence suggests otherwise. Few humans are necessary to add a significant level of fire disturbance. The topographically rough Missouri Ozarks reach ignition saturation at about 0.64 human per km^2 or less (Guyette and Dey 2000). The rougher Boston Mountains of Arkansas reach ignition saturation at about four or five humans per km^2. The smooth oak flats of the Highland Rim in middle Tennessee reach ignition saturation at about 0.50 human per km^2. Even at relatively low human population densities in a moderately dissected landscape (e.g., Missouri Ozarks) mean fire intervals of 10 years were supported by populations of less than 0.10 human per km^2 during a “depopulated” period (1680 to 1800).

A theoretical analysis of scenarios of human population density, ignition type, and culture illustrates the potential influence of anthropogenic ignitions in past fire regimes and consistency with the fire scar record. The population of North America (north of Mexico) has been estimated to be from 1 million to more than 10 million at the time of first contact and before population reductions by a variety of agents, e.g., introduced diseases. We chose the conservative estimate of about 1 million humans for this analysis. We used 500,000 humans for the estimate of population in the Eastern United States and southeastern Canada. The study area in eastern North America is about 3,024,000 km^2. We then subtracted the area of the Great Lakes (245,000 km^2) to obtain an area of 2,779,000 km^2. Although rough, this estimate is more precise than population estimates. Thus, population density roughly averaged 1 person for every 5.5 km^2, or 0.18 human per km^2 (500,000 humans/2,779,000 km^2). Given this population density, we generate quantitative scenarios of the rates of purposeful and accidental ignitions. Present-day rates of human-caused fires in forested areas of the Missouri Ozarks average one wildfire per 350 humans per year (0.0029 fire/human/yr). This rate occurred during an era of fire suppression (1991 to 2003), laws and societal pressures against burning, relatively few open fires (e.g., camp fires) per person, and limits on fire propagation imposed by land fragmentation. This rate applies to all arson (31 percent) and accidental fires (66 percent). Prior to European settlement, if the people of a culture viewed fire neutrally, the rate of ignitions by humans might be 10 times this amount (0.029 fires/human/yr). Thus, based on a pre-European settlement population and a neutral fire culture, the rate for human ignitions would be 0.0052 human-caused fires per km^2 per year (i.e., 0.029 fire/human/yr multiplied by 0.18 human per/km^2). This translates to a single human ignition for every 190 km^2 per year. If the people of a culture purposely used wildland fire as a tool (for many possible reasons) the rate of human ignitions might be 100 times greater, or 0.052 human ignition per km^2/year (i.e., 0.29 fire/human/yr multiplied by 0.18 human per/km^2). This translates to a single human ignition for every 19 km^2 per year. This rate (0.052 human-caused fire/km^2/yr) is about 200 times the rate of lightning ignition in the Central Hardwoods region (0.00025/ km^2/year) (Schroeder and Buck 1970). For example, if the average fire size was 1.9 km^2, this rate of ignition would maintain a fire rotation interval of about 10 years. This scenario and fire interval is not unlike the rate of burning (13 percent sites scarred annually) that occurred in 1790 in the moderately rough Current River landscape at population levels of about 0.017 human/km^2 (Guyette et al. 2002). The anthropogenic ignition rate is potentially much greater if intentional burning occurs (Williams 2000, Williams 2001). We conclude that even low levels of human population density combined with a culture of landscape burning were sufficient to provide levels of ignitions needed to maintain the fire frequencies we document with fire scars in Eastern North America.

SYNTHESIS AND MODELING OF PRE-EUROPEAN FIRE INTERVALS

Quantitative and empirically derived models of fire regimes can be used in many ways. Models allow past fire regimes to be estimated for ecosystems that have no on-site fire history information. Models can be used in conjunction with soil, geology, and species data to reconstruct past and potential flora and fauna. Thus, land managers interested in returning ecosystems to pre-European settlement conditions (including fuels) using prescribed fire can utilize models to estimate components of fire regimes including fire frequency. Researchers can use equations of this type to create a continuous landscape overlay of past fire frequency. Models without vegetation input also can be useful for independently estimating fire frequency at research vegetation plots, and for making inferences about possible changes in future fire regimes due to changes in population and climate. The preliminary results of the modeling that follows should be used with caution as they are temporally and spatially coarse scale, requiring additional validation and data from many regions of Eastern North America. Perhaps the most important aspect of fire interval models and fire history may not be the information provided on fire regimes but the perspective provided on the long-term interactions between humans and their environment.

Model Development

A preliminary model that predicts mean fire intervals for eastern oak forest ecosystems before European settlement was developed from fire scar data. The first iteration of the model was empirically derived from available fire scar data at 38 sites in Eastern North America (Table 1, col. 5). Although this data set showed predictive power, the eastern fire history data alone was inadequate in the range and distribution of the variables and did not meet the assumption of statistical normality. Thus, we used pre-European fire scar interval data from an additional 41 sites in the Western United States (Brown and Sieg 1996; Brown et al. 1997; Caprio 1998; Caprio and Swetnam 1995; Donnegan et al. 2001; Finny and Martin 1989; Fulé et al. 2003; Heyerdahl et al. 2001; Kipfmueller 2003; Miller and Rose 1999; Moir 1982; Swetnam et al. 1989 ; 1991).

These additional data extended the range of temperature and fire interval observations, increased the normality of the distribution of temperature means, more than doubled the degrees of freedom, and served to examine the hypothesis and model under a much broader set of conditions. We used published mean fire intervals, though in several cases we divided the period of record by the number of fires to estimate mean fire intervals. This was necessary when data were not expressed as mean fire intervals, data were not presented in FHX2 format (Grissino-Mayer 2001), fire dates were not published, long fire intervals were open ended, or the fire scar dates were not crossdated. Since the variables we used in this analysis are being refined and additional fire data are needed and forthcoming in many regions, we present this model as a work in progress with updated versions expected in the future. The two predictor variables of mean fire intervals are mean maximum temperature (*maxt*) and human population density (*pop*). Multicollinearity among predictor variables was negligible. Of the possible intercorrelations among variables, none was significant. The largest variance inflation factor (VIF) was 1.19, well below a VIF of 10, which would indicate that multicollinearity was influencing least squares estimates. The model is described by the regression equation:

$$MFI = -12.7 + 64.5e^{-0.098*maxt} - (6.3*\ln[pop]), \quad \text{(Equation 1)}$$

where MFI is the mean fire interval, *maxt* is the proxy mean maximum temperature (30 year average in degrees Celsius),
ln[*pop*] is the natural log of human population density (humans per km^2),
df = 78, $r^2 = 0.75$, all variables are significant ($p < 0.01$).

We tested the model's stability and predictive power by sampling half data with replacement 30 times. Variance explained (r^2) ranged from 0.63 to 0.83, the standard deviation of the coefficient of determination was 0.05; the r^2 mode was 0.75. The model and predictor variables tested significant ($p < 0.01$) in all model runs. The model predictions did not rely on extreme or rare sets of observations and did not vary greatly by the set of observations tested. On the basis of these results, the predictive power of the model as measured by the coefficient of determination proved stable.

The model predicted the mean fire intervals (Fig. 5) for the presence of fire in a 1- to 3-km^2 area during the 150 years before Euro-American settlement. Due to a westward progression of European settlement, this period varies by settlement date and the length of the fire scar record of each site, but generally begins from about 1650 to 1750 and ends between 1780 and 1850. The prediction period for this model also was restricted because the relationship between humans and fire scar frequency was positive only when ignitions were limited (Guyette et al. 2002). During later periods when human population density is high, this relationship becomes more complex and may even be negative. Also, the regression model does not predict or include stand replacement fires (the data are based entirely on survivor trees) or extend to prairies or regions where fire intervals are longer than the lives of recorder trees.

The useful components (predictor variables) of this preliminary model and map also are its caveats. One of the advantages of this model is that it is based on variables that change and can be used to examine different scenarios, for example, landscapes without humans or possible effects of climatic warming or cooling. One disadvantage is that spatial predictions probably are more static than actual (and often unknown) conditions due to the difficulty of variable quantification. This is especially true for the effects

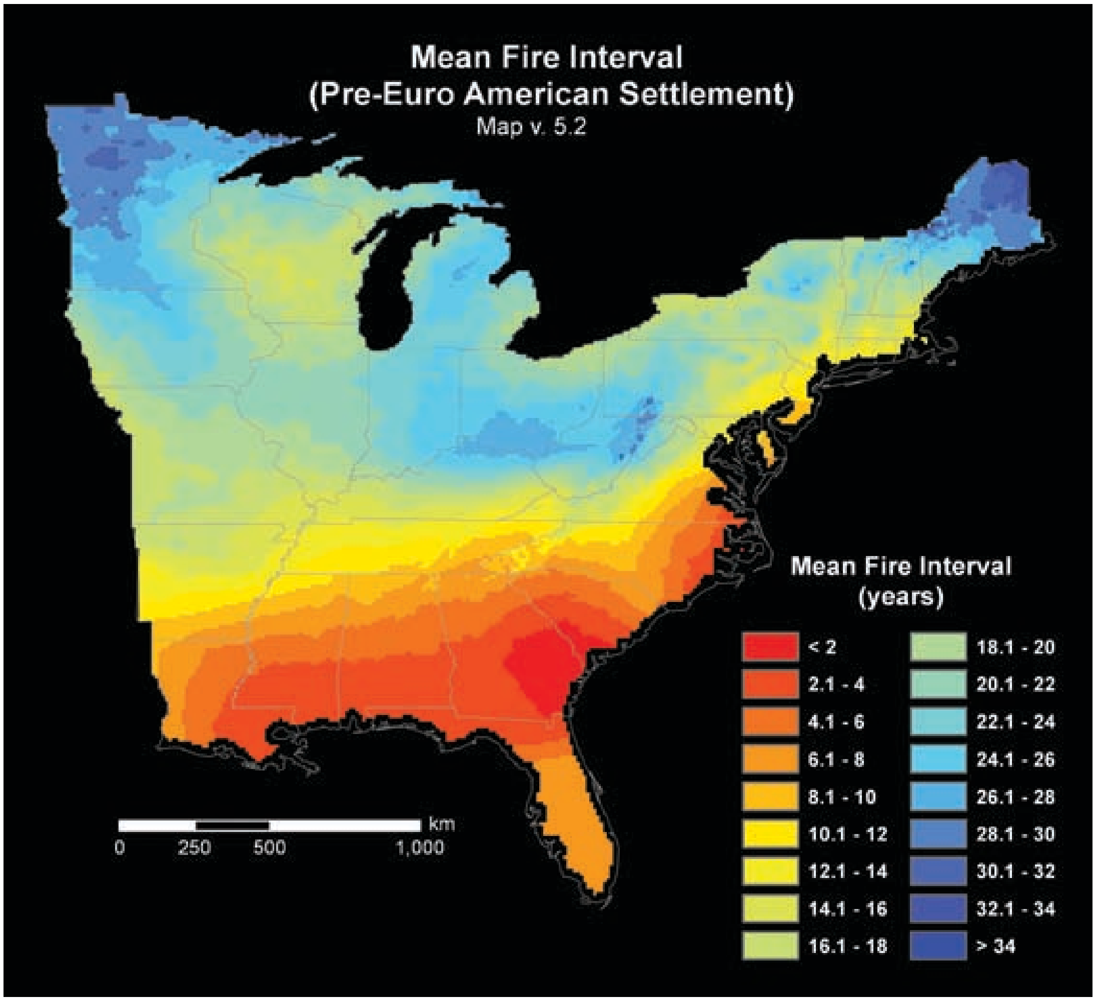

Figure 5.—First approximation of coarse scale mean fire intervals in the Eastern United States as predicted by mean maximum temperature and human population density (Equation 1). The mapped mean fire intervals are based on fire scar histories and model results generated from forested landscapes and nonstand replacement fire events. Mapped estimates are not valid for wetlands, grasslands, and other vegetation types where trees do not grow. The pre-Euro American period is based on fire scar intervals between 1650 and 1850. Estimates near ocean edges are unclassified (black). Mapped mean fire interval estimates are only as stable as human populations (e.g., several centuries earlier, the concentrated populations of the Mississippian cultural phase would have shortened fire intervals in the Mississippi and Ohio River Valleys).

of human-caused ignitions, which can be highly and abruptly variable through time and space.

Mean Maximum Temperature Proxy

We used annual mean maximum temperature (Daly et al. 2004), a precisely measured and modeled quantity compared to fire intervals, as a predictor of mean fire interval in Eastern North America. Because our period of interest (pre-Euro American settlement) for fire history and mean maximum temperature (1971 to 2000) are not temporally matched, we subtracted 0.4°C from the mean maximum temperature values to compensate for the effects of recent warming (Mann et al. 1998). Error caused by the non-temporal overlap of the temperature record with fire scar history will be minimal because the scale of temperature change between 1750 and 1985 (0.4°C) is small (about 35 times less) compared to differences among sites (about 14°C). Mean maximum temperature probably influences and controls many biotic and abiotic components that influence fire regimes and fire intervals such as fuel moisture, fuel production, and combustion. For example, the length of the fire season, as determined by the number of months in which 90 percent of the acres are burned in the four oak ecoregions (National Fire Occurrence Database and GIS Coverage 2005), is strongly correlated with maximum temperature ($r = 0.77$, $p = 0.01$). We speculate that mean maximum temperature is related to fire intervals and regimes in many ways, including:

1. The length of the fire season as influenced by snow cover at northern latitudes and high-elevation sites.

2. The length of the fire season as determined by the drying rate of fuels.

3. The direct effect of temperature in combustion reactions.

4. Mean *maximum* temperatures most often occur during the most fire-prone time of day.

5. The amount, types, and decay rates of fuels in an ecosystem are determined in part by temperature.

Other basic climate variables were considered as predictors. Precipitation, mean minimum temperature, and mean average temperature were included in the analysis but did not enter the stepwise multiple regression equation as significant. The fact that precipitation did not enter the model in stepwise regression probably reflects the potential complexity of the response of fire regimes to precipitation in hot versus cool climates. For instance, fire can be frequent in both hot-wet (Florida) and hot-dry (southern Arizona) climates. Also, we found that lightning ignition rates based on generalized maps of lighting ignitions of forest fires (Schroeder and Buck 1970) did not enter the stepwise regression, and that correlations between mean fire intervals and lightning ignitions were not significant ($r = 0.16$, $p = 0.16$). However, separation of the data into eastern and western sites yielded a significant correlation ($r = 0.40$, $p < 0.01$) with lightning fires in Western but not Eastern North America ($r = 0.12$, $p = 0.46$). This result partially reflects the lack of fire history sites in the in the lightning and fire-prone Southeastern United States.

Population Density and Ignition Proxy

We used human population density as a proxy for the number of anthropogenic ignitions. Although the estimation of Native American populations is fraught with problems (Henige 1998), progress has been made in assessing estimates and observations made by early workers (Denevan 1992; Mooney1928). We used mapped population density classes of Native Americans in North America (Driver and Massey 1957) and more detailed population estimates in the Ozarks (Guyette et al. 2002). We used population class variables to facilitate the quantification of pre-census estimates of human population density, which have a large degree of potential error and may never be known precisely. These estimates are consistent with conservative estimates of Native American population (Waldman 1985). However, temporal continuity in population density estimates is variable. Driver and Massey's mapping of human population is not unlike the *relative* spatial distribution of human population density in 2000 (U.S. Census Bur. 2000), illustrating that suitable human habitat (and spatially relative population densities) is somewhat consistent through time. For example, population densities are high during both periods along coastal areas, in the Northeast, near the Great Lakes, in localized areas of the Southwest, and near Appalachia. Also, spatial error in population estimates may be no greater than the variance in population density owing to the migration, immigration, and decline of populations of humans (Thornton 1987) during the two centuries of the pre-Euro American settlement fire scar record. Although these factors make precise calibration between fire frequency and population density difficult over large temporal and spatial scales, their significant ($p < 0.01$) correlation ($r = - 0.45$) supports the theory that humans had large-scale effects on fire regimes.

Mapping Fire Intervals

The model results were mapped using ESRI®ArcGIS™ software (Environ. Syst. Res. Inst. 2005). Gridded mean maximum temperature data (Daly et al. 2004) and coverage of human population density (Driver and Massey 1957) were applied to Equation 1 to produce estimates of mean fire intervals for a pre-Euro American settlement period (about 1650 to 1850).

Prior to mapping, population data were smoothed using a circular (25-km radius) neighborhood mean to more closely reflect the mobility of humans. Mapped coarse scale fire intervals have error based on the model (Equation 1). The model error indicates that the 95-percent model confidence interval is about ±3 years while the 95-percent prediction interval is about ±11 years (less for shorter intervals). These error estimates include all the variability in vegetation, topography, and local human population not evident in the predictor variables that can affect fire frequency at finer spatial scales. During periods that precede model calibration, the model may work within limits but the mapped estimates of mean fire intervals must be based on temporally appropriate predictor variable data.

CONCLUSION

The quantitative history of wildland fire derived from fire scar studies in Eastern North America provides several insights into past and future fire regimes. Seemingly important variables such as precipitation did not explain variance in mean fire intervals beyond what was explained by temperature. Variables relevant to fire regimes and fire intervals in eastern oak forests were identified and include mean maximum temperature, human population density, extreme drought events, and topographic resistance to the spread of fire. Some of these variables are dynamic, such as human population and temperature, and their potential change in the future may influence fire regimes. Understanding how these variables interact to influence fire regimes will aid in assessing future fire risk with changes in fuel continuity, temperature, extreme climate events, human culture, and population density.

The interaction between drought and intentional human ignitions will make future fire regimes potentially unstable and difficult to predict. Human populations are increasing and will provide more ignitions, making fire regimes more responsive to drought. This will be countered to some extent by improved suppression response time and new technologies. The stability of fire regimes in changing climates may be low, particularly in landscapes with a low resistance to fire propagation and changes in the number and distribution of human ignitions. Drought has inspired purposeful human burning both in the past and present. It is doubtful that this relationship will cease to exist in the future.

The effects of topography on the propagation of fire have been understood by humans for a long time. Fire scar data quantified this effect at a general landscape level. The mitigation of fire by topography has been relatively static over geologic time scales compared to the ephemeral changes caused by the frequency of human ignitions. Thus, in terms of temporal scale, topographic roughness is by far one of the most consistent and important variables affecting the frequency of fire.

Fire history modeling and mapping from fire scars can provide estimates of fire intervals for restoration and reference conditions (Maclean and Cleland 2003; McKenzie et al. 2000; Morgan et al. 2001). However, the quality of any model or map is dependent on the quality of the data from which it is derived. In this study, we mapped large regions of the Southern and Eastern United States for which pre-European settlement fire scar histories were unavailable. Therefore, estimates for these regions are uncalibrated and approximate but might be improved by future studies and data collection.

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UNDERSTANDING THE EVIDENCE FOR HISTORICAL FIRE ACROSS EASTERN FORESTS

Charles M. Ruffner[1]

Abstract.—Evidence for historical fire across the eastern deciduous biome spans several fields, including paleoecology, fire scar analysis, witness tree studies, historical documents and ethnographic sources. In this paper I provide an overview of many of these methods as well as the limitations and examples of each. While the use of any single approach has its cautions and pitfalls, the interpretation of the considerable material available allows a view into the history and effects of fire on eastern oak forests.

INTRODUCTION

As foresters, ecologists, and land managers, each of us is charged with understanding our ecosystems and the processes that have acted to form those systems through time. Learning as much as possible about the original vegetation and structure of our systems is vital to identify and manage these natural areas for the optimal biodiversity inherent in original conditions (Loucks 1970; Ladd 1991; Kay 1995; Olson 1996). Across the East, the historical role of fire has been implicated in driving diversity and maintaining keystone species in many ecosystems and the body of literature investigating these processes continues to grow. Mutch (1991) suggested that fire regime information is "absolutely essential background data for the appropriate design and implementation of resource management projects at the ecosystem and landscape level of organization." In southern Illinois, I work with landowners and agency managers in finding information concerning the pre- and post-European settlement conditions of their land holdings. These endeavors often focus on gaining a better understanding of the historic range of variability represented in some systems, or understanding the spatial distribution of hill prairies and barrens. No matter the objective, much of the data sources available for these studies are found across several disciplines and thus are sometimes difficult to access or, quite frankly, understand.

The objectives of this paper are to review many of the basic approaches used to interpret or infer historical conditions for fire so that land managers can begin to assemble and interpret evidence for their respective holdings, no matter where they occur across the eastern forests. I hope that many engaged land managers will embrace the historical ecology approach in researching and managing their properties. Because so many excellent reviews have been published I reference them often so that attendees here can find those key resources and use the many citations included.

[1]Department of Forestry, Southern Illinois University, Carbondale, IL 62901-4411, 618-453-7469, email: Ruffner@siu.edu

Investigating the role of fire in eastern oak forests

To a large extent, historical fire has been implicated in the wide distribution of oak forest and woodlands, prairies, savannas, and glades that are found within nearly every state in the eastern deciduous biome (see reviews by Lorimer 1985; Abrams 1992; 2002, Van Lear and Watt 1993; Whitney 1994; Dey 2002). The wide range of evidence concerning historical fire across many parts of the East ranges from paleoecology studies (see Patterson, this volume), fire histories and chronologies (Guyette, this volume), archaeological investigations (Chapman et al. 1982; Delcourt et al. 1986, 1998), native American sources (Parker 1968' Adovasio et al. 1985' Doolittle 1992; Kimmerer and Lake 2001), and witness tree analysis and surveyor records (Bourdo 1956; Hutchison 1988; Whitney 1994; Black and Abrams 2001; Abrams 2003), to historical documents such as deed descriptions and tax records (Foster 1992; Olson 1996), plat maps, travel documents and journals (Schoolcraft 1820; Burges 1965; Ladd 1991,;Whitney 1994; Olson 1996; Ruffner and Arabas 1999), and stand structure analysis and dendroecological studies (Henry and Swan 1977; Dorney and Dorney 1989; Nowacki and Abrams 1992; Ruffner and Abrams 2002).

Specific limitations and/or assumptions about many of these data formats must be considered (Forman and Russell 1983, also see review in Whitney 1994). For many of the primary documents, the historical context during its writing or surveyor bias against some landscape feature or misidentification of a species and least of all the observers' training and skill level may shade the interpretation of these historic documents. Nonetheless, as many authors note, these data sources often are the only representation of the pre-European landscape. Despite their variable training levels, many primary authors had " the unsurpassable advantage of actually having seen presettlement landscapes and aboriginal land management practices" (Ladd 1991). Thus, while many methods have their critics, when interpreted together or by applying an historical ecological approach they provide the land manager with much needed information to place her/his land parcel in its historical context and begin to model the types of vegetation structure and function appropriate for that area (Myers and Peroni 1983).

Paleoecology

Across the eastern deciduous biome, many authors have used pollen and charcoal analysis to reconstruct and interpret vegetational development at different scales (Watts 1979; Webb 1981; Whitney 1994). According to Patterson (this volume) some areas of the region have fairly well documented paleoecological reconstructions based on the work of key researchers, for example, Watts' work in the Appalachians (Watts 1979), Davis' work in the northeast and Lake States (Davis 1983), Clark's work in the Midwest and Northeast, and work in New England, primarily from Patterson and associates at the University of Massachusetts and from Foster and others at Harvard Forest.

In comparing sedimentary charcoal and archaeological site distributions in New England, Patterson and Sassaman (1988) found that fires were common on coastal sites where native populations were greatest and their land-use practices most intensive. Across the region, others have noted archaeological site distributions corresponding well with areas characterized by high oak pollen percentages and usually elevated amounts of charcoal fragments (Dincauze and Mulholland 1977; Patterson and Sassaman 1988; Delcourt et al. 1998; Fuller et al. 1998). More recently, Clark and Royall (1995) contributed greatly to this discussion reporting a transition from northern hardwood to white pine-oak forests at Crawford Lake, Ontario, following a period of Iroquois occupation and agricultural clearing. Moreover, Ceci (1979) and Loeb (1998) discussed native corn and hickory planting episodes from the pollen record in the coastal regions of New York. Delcourt et al. (1998) reported that Archaic and Woodland period natives cleared forest gaps to cultivate native plants, and that human-caused fires increased populations of fire tolerant oaks, chestnut, and pine in upland forests of the Cumberland Plateau of central Kentucky. Still, Clark and others have provided evidence that some oak communities can be maintained in oak-chestnut with litle charcoal present in their sedimentary records (Clark and Royall 1995; Foster et al. 2002). While several regional studies suggest the correlation of fire occurrence and native occupation with oak forest distribution, more research at the local level must be completed to better understand the pre-European landscape across the Northeast (Black and Abrams 2001; Parshall and Foster 2002).

Fire Scars

Another quantitative approach to understanding historical fire is using dating fire scars or tree-ring analysis to reconstruct disturbance regimes at native sites (Batek et al. 1999; Shumway et al. 2001; Ruffner and Abrams 2002). Few presettlement fire papers have been published for the Northeast; most represent fire evidence from the Midwest and Central plains (Abrams 1985; Guyette and Cutter 1991; Cutter and Guyette1994; Batek et al. 1999). Buell et al. (1954) reported six fires scars from a single tree dating from 1641 to 1711 in Mettler's Woods, New Jersey, indicated a mean fire interval (time between fires) of 14 years. In western Maryland, Shumway et al. (2001) reported a mean fire interval (for consistency) of 7.6 years in an old-growth upland oak-hickory forest. Across the region, estimates of mean fire interval within oak forests are surprisingly similar ranging from 2 to 24 years (Shumway et al. 2001). At many sites, fire frequency increased or decreased during the initial period following European settlement depending on the stand and region. Fire histories for the Missouri Ozarks indicate that during periods of late Native American settlement (1701 to 1820), fire return intervals were longer (11.96 years ± 2.4, mean ± SE) than those during European settlement (3.64 years ± 0.35) (adapted from Guyette and Cutter 1991). In southern Illinois, little is known of the historical fire regime because most of the needed fire scars were lost after logging of the primary forests and deterioration of cut stumps (Robertson and Heikens 1994; Olson 1996). Robertson and Heikens (1994) reported high fire ignitions during European settlement due to farmers clearing underbrush from the forest, closely following Guyette and Dey's model of frequent European ignitions immediately following settlement.

Moving eastward, a fire history study of mixed oak forests originating after 1850 in southeastern Ohio revealed that fire return interval averaged 7.5 years (Sutherland 1997). In northwestern Pennsylvania, Ruffner and Abrams (2002) reported dendrochronological evidence at a native Seneca village site revealed a frequent, low-intensity fire regime with a return interval of 11 to 26 years during native habitation. However, following European settlement (post-1800) the fire free period is significantly longer with a disturbance-free interval of 28.5 ± 2.8 years. Where prehistoric fire scars are available, these studies have provided valuable insight into native disturbances and their localized impacts. However, they are largely restricted to the area immediately surrounding the study sites and cannot characterize native influences at the landscape level (Batek et al. 1999).

Witness Trees

During the period of early European settlement in the Eastern United States, land surveyors recorded and marked witness trees (warrant, bearing, and/or corner) to identify property corners and boundaries (Lutz 1930; Spurr 1951; Bourdo 1956). When compiled from surveyors' notes, these trees provide a landscape level snapshot of forest composition at the time of European settlement and have been used to reconstruct regional vegetation composition throughout the Eastern and Midwestern United States (Lutz 1930; Siccama 1971; Whitney 1986; 1990, Loeb 1987; Seischab 1990, 1992; Marks and Gardescu 1992; Nowacki and Abrams 1992; Abrams and Ruffner 1995; Black and Abrams 2001). Such vegetation reconstructions are valuable in areas where most original vegetation has been destroyed or altered, making it difficult to assess natural vegetation types or disturbance regimes. In many studies, reconstructed patterns of vegetation have been attributed to edaphic conditions such as topography, soils, slope class or drainage classes (Gordon 1940; Abrams and Ruffner 1995). In more recent years studies have taken a more integrative approach and recognized that native populations also may have affected vegetation patterns. A study by Dorney and Dorney (1989) documented a native maintained oak savanna in northeastern Wisconsin using General Land Office survey information.

In documenting pre-European vegetation of central New York State, Marks and Gardescu (1992) identified old Native American clearings and reported oak-hickory-pine growing in areas previously inhabited by peoples of the Iroquois Confederacy. In western New York, on lands surveyed for the Holland Land Company, including Gordon's (1940) area, Seischab (1990) reported that most oak communities occurred in southern areas along the Allegheny River and its tributaries, dominating a landscape formerly inhabited by Seneca people of the Iroquois Confederacy. Within the Lower Susquehanna Valley of Pennsylvania, Black and Abrams (2001) reported significant differences in witness tree distributions between areas adjacent to Native American village sites and those in areas with low native activity. Whitney and Decant (2003) suggested that Iroquois habitation may be associated with oak-chestnut forests along the French Creek area in their witness tree study of northwestern Pennsylvania.

Most recently, Black et al. (unpublished) integrated witness tree distributions, Native American archaeology, and geologic and topographic variables to investigate the relationships between Native American populations and pre-European settlement forest types on the Allegheny Plateau of northwestern Pennsylvania. Logistic regression of natural and cultural variables demonstrated that Native American habitation and travelways were the most significant predictor of oak, hickory, and chestnut trees in the pre-settlement forest. Although cause and effect cannot be tested at this time (the paleoecology data are still being analyzed), the authors suggest that long-term Native American activities such as agricultural clearing and burning, as well as possible wood and mast resource extraction selected for the disturbance, adapted oak-hickory and chestnut. Overall, an increasing number of studies suggest that Native Americans had at least some association with forest composition, though the degree of the relationship often is not quantified.

Historical documents

Much of the literature surrounding Native American effects on vegetation have been developed from some type of historical document. By the middle of the 20th century, several key historical geographers had laid the framework for investigating native land uses e.g., Maxwell (1910), Bromley (1935), Day (1953), Stewart (1963), and Sauer (1975). These authors reviewed numerous primary and secondary sources to build their cases that native groups had influenced vegetation to varying degrees across the continent (Whitney 1994). While some authors suggested the widespread use of fire by natives, others cautioned against the idea of natives wantonly burning the landscape. Still, the preponderance of early forests described as "open and park like" led Bromley (1935) to conclude that it was "due to the universal factor of fire, fostered by the original inhabitants to facilitate travel and hunting" (Nowacki 2002). Indeed, natives affected ecosystem development in a variety of ways: hunting, fishing, agricultural clearing, and associated activities (including the introduction and dissemination of new cultigens), wood gathering, village and trail construction, and habitat manipulation (Maxwell 1910; Day 1953; Mellars 1976; Nowacki 2002). It is widely accepted that natives intentionally used fire for more than 70 reasons (Lewis 1993), particularly in the alteration and maintenance of surrounding environments for their benefit (Day 1953; Cronon 1983; Mellars 1976; Doolittle 2004).

Examples from early travel documents include William Strachey's (1620) description of the Hampton, Virginia, area as "the seat sometime of a thowsand Indians and three hundred houses…which is the reason that so much ground is there cliered and opened, enough already prepared to receive corne and make viniards of two or three thowsand acres." Another published in 1672, <u>The Discoveries of John Lederer: In three several marches from Virginia to the west of Carolina</u>, relates open valleys and active burning of pine forests near Indian settlements throughout the Piedmont and Ridge and Valley provinces. In 1794, William Bartram traveled through the Cherokee, Creek, and Choctaw nations, characterizing their farms, orchards, and woodlands. In 1810, Francois Michaux published memoirs of his and his father's travels through the southern Appalachians with their astute depictions of native village life and culture. Farther west, Henry Schoolcraft's 1821 <u>Ozark Journals</u> documents his travels through the Ozark region where his training allowed him to accurately identify plant assemblages and report on the character and conditions of many unique ecological sites and phenomenon encountered on his trips through Indian territory. Most authors recorded an anthropogenic landscape heavily influenced by native activities and populations.

Researchers using surveyor accounts, treaties, deeds, wills, sheriff sales, tax records, and other collected papers and manuscripts have characterized and explained vegetation change from pre-European settlement to the present (Raup 1966; Cronon 1983; Foster 1992; Orwig and Abrams 1994; Ruffner and Abrams 1998; Foster 2002). While native effects differ across these studies, there are general patterns. For instance, many early documents reflect a managed landscape with tended agricultural fields interspersed through open woodlands and numerous abandoned habitation areas. This clearly suggests a cultural landscape. In a search of historical documents of Cape Cod, one of the earliest places written about by Europeans, Patterson and Ruffner found evidence for large "open fields" and "upland meadows" surrounding native settlements (Plymouth Colony records, vol. 3, p. 85). A 1648 deed records "the fields over the pond have been cleared and improved by the Indians as long as the oldest can remember" (Massachusetts Archives. 33: 245-247). As in other places across the East, these "planting fields" were the most commonly sought after real estate by Freemen during the ensuing years of European expansion (Williams 1989, Whitney 1994).

Another common observation is the usefulness and purposeful nature of many native burning regimes, which usually were adopted by incoming European settlers (Whitney 1994). Whether the reason was reducing rattlesnake and insect populations, improving browse or fruit production, or easing travel through the forest, authors have reported the widespread adoption of the native practices. Plymouth Colony records revealed evidence for early prescribed burning of pine lands by selected officials because they recognized the need to reduce potential fire hazards around settlements and improvements (Lovell 1984).

CONCLUSION

The use of historical evidence to reconstruct past vegetation distribution and fire regimes spans paleoecological, ethnobotanical, and archaeological fields. While the field of historical ecology has grown over the last 50 years to include many disciplines and journal outlets, I hope that this introduction contains sufficient examples of outstanding research and review articles to allow newcomers to find new sources of information. It is testament to the usefulness of these historical analyses that so many authors rely on them to corroborate other sources of evidence pertaining to past landscapes and events.

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OAK COMPOSITION AND STRUCTURE IN THE EASTERN UNITED STATES

W. Keith Moser[1†], Mark Hansen[2], Will McWilliams[3], and Ray Sheffield[4]

Abstract.—Although oak species currently occupy a dominant position in most eastern deciduous forests, particularly on upland sites, many scientists and managers have expressed concern about the future of this genus in the absence of the disturbance patterns that facilitated its establishment up to now. Reductions in timber harvesting and fire in particular may give the advantage to competitors such as maples. Using data from the Forest Inventory and Analysis program of the U. S. Forest Service, we looked at current data and temporal trends to gauge the trajectory of oak forests in the Eastern United States. The area of the two upland oak groups—oak-hickory and oak-pine—covered 160.3 million acres or 43 percent of eastern timberland. The oak volume per acre of timberland has increased over the last four to five decades. Yet, we are seeing a decline in the proportion of total timberland with at least 20 ft^2ac^{-1} of select red or white oaks (the "select oak" stands). While the select oak basal-area component within these stands increased slightly, it represents a decreasing proportion of the total basal area in the stand, suggesting that associated species are increasing in their share of the overstory. While the total number of seedlings/saplings in the understory of stands with select red or white oak[5] basal area greater than 20 ft^2 ac^{-1} has been increasing, the proportion of all seedlings/saplings that are select white oak seedlings/saplings has been declining over the last 20 or so years. The declining proportion of regeneration represented by oak species suggests a future eastern U. S. forest with substantially reduced proportions of oaks in the overstory. Reintroducing disturbances such as fire is essential to maintain oaks' overstory presence and associated biological and economic benefits.

INTRODUCTION AND METHODOLOGY

Oaks have been in eastern U. S. forests for at least 6,000 years Lorimer (1993). While current oak forests evolved through a combination of ecological and human-influenced factors (McWilliams et al. 2002), changes in disturbance patterns are altering stand development trajectories to the detriment of oak (Larson and Johnson 1998; Smith 2005). Other authors at this conference will present their interpretation of oak regeneration patterns that lead to eventual canopy occupancy (see Abrams, this volume); in this paper we examine trends, status, and implications of the structure and composition of oak forests in the Eastern United States.

We used data from the national forest inventory and analysis program (FIA) of the USDA Forest Service (Frayer and Furnival 1999). The FIA program conducts comprehensive forest inventories to estimate the area, volume, growth, and removal of forest resources in the United States, and measures the health and condition of these resources. The program's sampling design has a base intensity of one plot per approximately 6,000 acres and is assumed to produce a random, equal probability sample. The national FIA program consists of five regional programs[6] that provide estimates of forest area, volume, change, and forest health throughout the United States (McRoberts 1999). We used data from three of these regional FIA programs—North Central, Northeastern and Southern—to depict forest conditions for the Eastern United States. For historical data we used data generated from past Forest Inventory reports for

[1]USDA Forest Service, Northern Research Station, Forest Inventory and Analysis Program, 1992 Folwell Ave., St. Paul, MN 55108; [2]USDA Forest Service, Northern Research Station, 1992 Folwell Ave., St. Paul, MN 55108; [3]USDA Forest Service, Northern Research Station, Forest Inventory and Analysis Program, 11 Campus Blvd., Suite 200, Newtown Square, PA 19073; Southern Research Station, Asheville, NC; [†]Corresponding author, 651-649-5155, email: wkmoser@fs.fed.us.

[5]The categories "select red oaks" and "select white oaks" have historically referred to those species preferred by mill owners for their uniform characteristics, quality and yield of higher grades of lumber. In examining trends over the last 25 or so years, we categorized stands as select red or white oak acreage as in those stands with at least 20 ft^2ac^{-1} of the total basal area in select red white oak species and limited our analyses to those categories. These trends reflect management and utilization of those species that are most identified with the name "oak" and that comprise a significant proportion of the genus' volume.

[6]Soon to be four. The North Central and Northeastern FIA programs will merge into the Northern FIA program by 1 October, 2006.

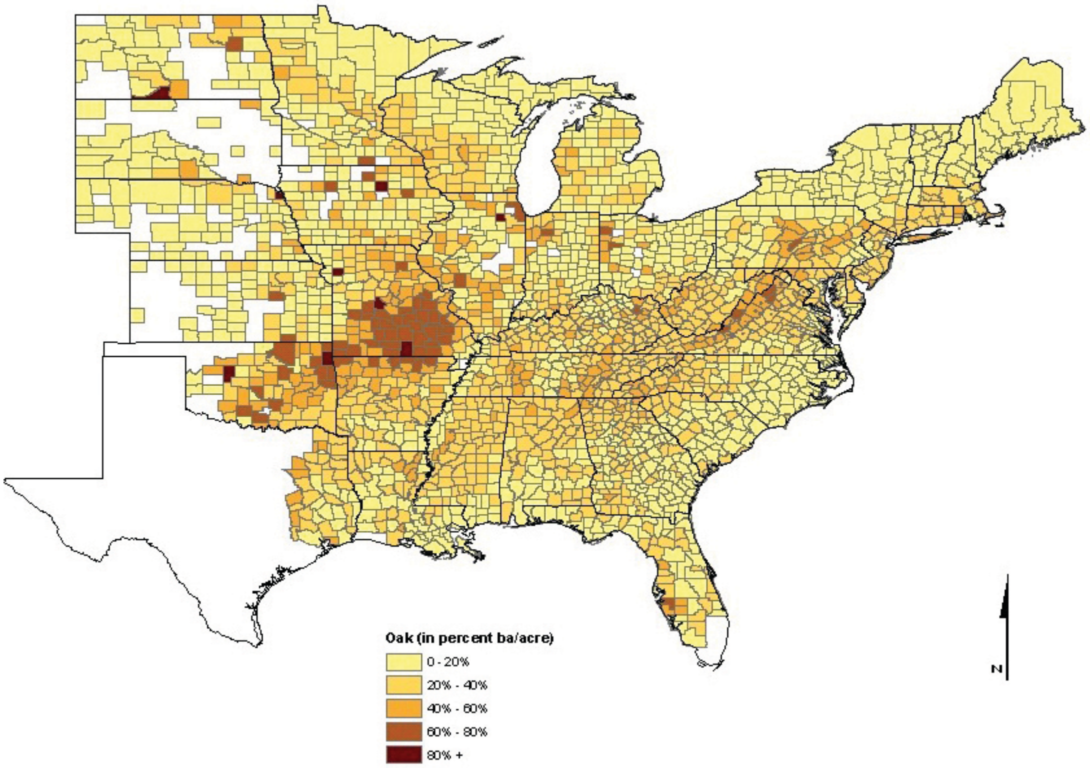

Figure 1.—Oak species' percentage of total timberland basal area, based on the most recent FIA inventories from each state.

states in the Eastern United States and data generated by the FIA Mapmaker program (Miles 2005). For current structure and regeneration, we used data generated by the FIA database.

Of the major deciduous forest-type groups in the Eastern United States, oaks are common associates of all but the northernmost groups of spruce-fir and aspen birch. Oak is most prevalent in the two upland oak groups: oak-hickory and oak-pine. Oaks also are members of other upland deciduous forest types. As defined by the FIA, upland oak forest type groups included eight detailed types within the oak-pine group and 11 oak types among the 17 types within the oak-hickory group. These groups are defined by the proportion of total stocking represented by oaks. Oak-hickory group includes stands where half or more of the stocking is contributed by oak or oak-dominated stands. For the oak-pine group, stocking of oaks and other deciduous species is from 25 to 50 percent (McWilliams et al. 2002).

RESULTS

Current Distribution of Mixed-oak Stands in Eastern United States

We used inventory data from the FIA database to identify the presence of oaks in the Eastern United States. Figure 1 is a map that displays the proportion of the total basal area occupied by all oak species. The occurrence of oak throughout forest stands in the east is readily apparent. This map shows that while eastern oaks are present from Maine to Louisiana and Minnesota to Florida, they comprise the most dominant portion of the canopy in the Ozarks and in portions of the Appalachian Mountains. Other areas with a high proportion of oaks include the Central Lowlands of Minnesota and Wisconsin, and the Tennessee River valley.

The area of the two upland oak groups covered 160.3 million acres or 43 percent of eastern timberland in the most recent inventories (Table 1). Upland oak forests compose at least 48 percent of the timberland in the

Table 1.—Area of timberland in the Eastern United States, in millions of acres and from the most recent inventory, by region and broad forest type, with percent of region total[1]

	Total	Upland oak[a]		Other upland deciduous		Lowland oak[b]		Other lowland deciduous		Conifer		Other	
		Area	Percent	Area	Percent	Area	Percent	Area	Percent	Area	Percent	Area	Percent
Lake States[c]	49.08	8.30	17	24.90	51	-	0	3.97	8%	11.39	23	0.52	1
Central States[d]	25.21	19.06	76	2.02	8	0.09	0	2.96	12	0.85	3	0.23	1
New England States[e]	31.31	4.52	14	16.58	53	-	0	0.81	3	9.30	30	0.10	0
Mid-Atlantic States[f]	56.03	26.93	48	22.50	40	0.11	0	2.21	4	3.82	7	0.46	1
Atlantic states[g]	96.01	47.84	50	1.04	1	3.91	4	9.60	10	32.99	34	0.62	1
Gulf states[h]	105.85	52.05	49	0.10	0	8.03	8	10.13	10	35.00	33	0.53	1
Plains states[i]	5.33	1.56	29	0.55	10	0.01	0	1.32	25	1.67	31	0.23	4
Total	368.82	160.27	43	67.7	18	12.15	3	30.98	8	95.03	26	2.69	1

[1]Source: Forest Inventory Database (Miles 2005).
[a]Includes oak-hickory and oak-pine forest-type groups.
[b]Includes swamp chestnut oak - cherrybark oak, sweetgum-Nuttall oak-willow oak, and overcup oak-water hickory forest types.
[c]Michigan, Minnesota, and Wisconsin.
[d]Illinois, Indiana, Iowa, Missouri.
[e]Connecticut, Massachusetts, Maine, New Hampshire, Rhode Island, and Vermont.
[f]Delaware, Maryland, New Jersey, New York, Ohio, Pennsylvania, and West Virginia.
[g]Florida, Georgia, Kentucky, North Carolina, South Carolina, and Virginia.
[h]Alabama, Arkansas, Louisiana, Mississippi, Oklahoma, Tennessee, and Texas.
[i]Kansas, Nebraska, North Dakota, and South Dakota.

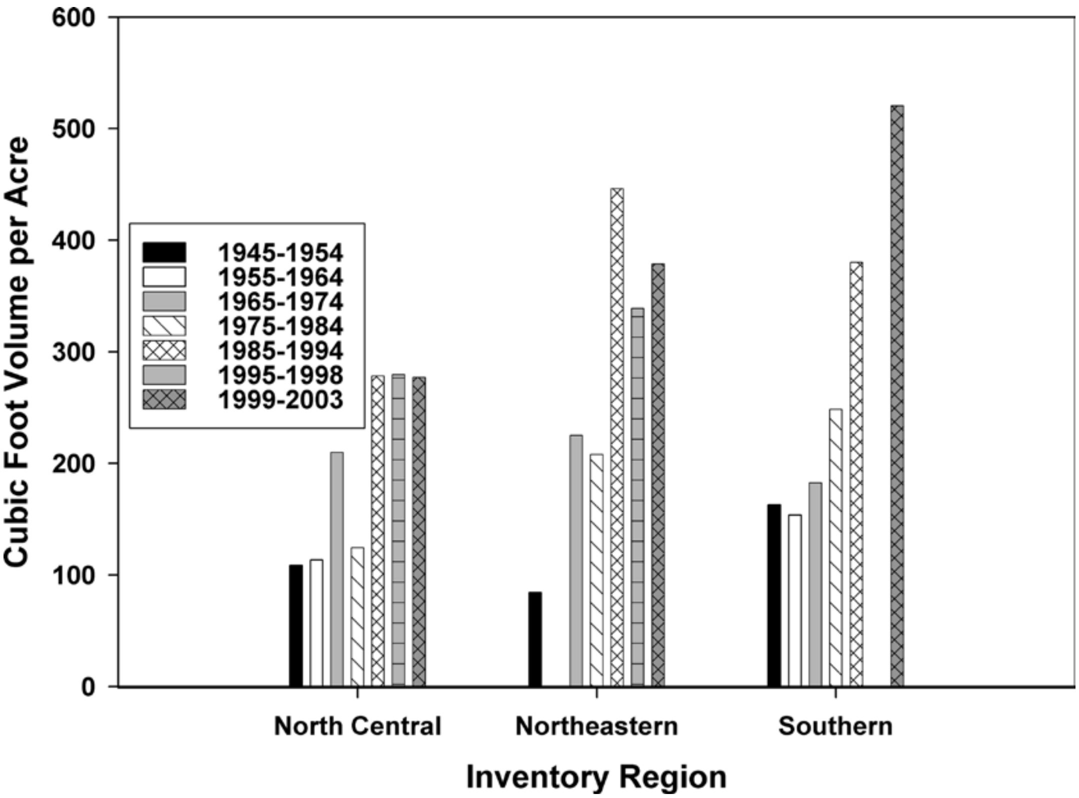

Figure 2.—Average volume of oak species per acre of timberland (ft^3 ac^{-1}) by FIA program. North Central states: Illinois, Indiana, Iowa, Kansas, Michigan, Minnesota, Missouri, Nebraska, North Dakota, South Dakota, and Wisconsin; Northeastern states: Connecticut, Delaware, Maryland, Maine, Massachusetts, New Hampshire, New Jersey, New York, Ohio, Pennsylvania, Rhode Island, Vermont, and West Virginia; Southern states: Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, Oklahoma, South Carolina, Tennessee, Texas, and Virginia.

Atlantic, Central, Gulf and Mid-Atlantic states. Other upland deciduous groups cover 67.7 million acres or 18 percent of eastern timberland. The Gulf states had the most oak, with 32 percent of the total eastern U.S. upland oak timberland area and 66 percent of the lowland oak timberland.

Oak volume has generally increased across the region (Fig. 2), in some cases at a declining rate. In upland oak stands with at least 20 ft^2 ac^{-1} of oak basal area, most acreage was in the sawtimber-size stands (Table 2). There are several explanations for this distribution. First, oaks are long-lived species and thus spend a lower proportion of their total life in the seedling-sapling or poletimber size classes. Second, according to FIA protocol, mixed-age stands often are assigned to the size class of the largest component. For example, a stand that has one-third of the stocking in each size class is called sawtimber.

Some authors have suggested that the age-class distribution of oak stands is unbalanced and skewed toward older stands (Abrams and Nowacki 1992; Abrams 2005; Healy et al. 1997; Lorimer 1993). The FIA stand-size variable can provide some indication of the stages of stand development (Oliver and Larson 1996), but the correlation with stand or tree age is less robust, because the classification is based solely on tree diameter (McWilliams et al. 2002). Each FIA plot has several "age" trees that are used to develop productivity equations. Because only the most dominant overstory trees are sampled, the ages may not represent all plot trees; those data are not considered here.

FIA has timberland area delineated by forest types which, in turn, are combined into forest type groups. We examined the components of upland oak type forest-type groups: oak-pine and oak hickory by forest type and stand size class (Table 2). The white oak/red oak/hickory

Table 2.—Timberland area of upland oak forest type groups by forest type and stand size class, for the states east of the Great Plains

Forest type	Total	Seedling-sapling		Poletimber		Sawtimber		Nonstocked	
		Area	Percent of Total	Area	Percent of Total	Area	Percent of Total	Area	Percent of Total
Oak/pine group	6.9	0.7	10	4.3	62	1.9	28	-	-
White pine/red oak/white ash	3,836.90	420.6	11	1,175.70	31	2,240.60	58	-	-
Eastern redcedar/hardwood	2,895.70	769.2	27	1,481.50	51	644.9	22	-	-
Longleaf pine/oak	1,248.90	639.6	51	330.7	26	275.9	22	2.7	0
Shortleaf pine/oak	4,398.20	579.9	13	1,438.80	33	2,379.50	54	-	-
Virginia pine/southern red oak	2,178.90	439.1	20	820.5	38	919.3	42	-	-
Loblolly pine/hardwood	15,792.10	6,104.70	39	3,722.50	24	5,959.80	38	5.1	0
Slash pine/hardwood	1,797.30	697	39	441.7	25	653.5	36	5	0
Other pine/hardwood	2,846.60	707.2	25	978.7	34	1,157.80	41	2.9	0
Total oak/pine group	35,001.50	10,358.00	30	10,394.40	30	14,233.20	40	15.70	0
Oak/hickory group	1,444.10	922.5	64	243.7	17	277.9	19	-	-
Post oak/blackjack oak	5,680.30	808.5	14	2,460.90	43	2,410.90	42	-	-
Chestnut oak	5,728.70	165.4	3	1,750.10	31	3,813.20	67	-	-
White oak/red oak/hickory	49,419.70	5,586.10	11	13,927.10	28	29,906.40	61	-	-
White oak	4,779.10	72.6	2	1,183.10	25	3,523.50	74	-	-
Northern red oak	3,782.20	110.5	3	805.5	21	2,866.20	76	-	-
Yellow-poplar/white oak/red oak	7,450.90	739.7	10	1,674.50	22	5,036.60	68	-	-
Sassafras/persimmon	965	299.3	31	390	40	275.7	29	-	-
Sweetgum/yellow-poplar	6,649.20	2,614.80	39	1,798.60	27	2,235.80	34	-	-
Bur oak	582.1	26.7	5	118.2	20	437.2	75	-	-
Scarlet oak	527.6	30.9	6	207.9	39	288.8	55	-	-
Yellow-poplar	1,250.20	176.5	14	378	30	695.6	56	-	-
Black walnut	653.8	81.5	12	186.5	29	385.9	59	-	-
Black locust	474.9	168.2	35	206.9	44	99.8	21	-	-
Southern scrub oak	1,281.10	904.8	71	326.6	25	39.5	3	10.2	1
Chestnut oak/black oak/scarlet oak	3,051.70	177	6	944.4	31	1,930.40	63	-	-
Red maple/oak	2,381.60	368	15	965.1	41	1,048.50	44	-	-
Mixed upland hardwoods	29,166.60	7,640.70	26	8,851.60	30	12,652.10	43	22.2	0
Total oak / hickory group	125,268.80	20,893.70	17	36,418.70	29	67,924.00	54	32.40	0
Total (Thousand acres)	160,270.30	31,251.70	19	46,813.10	29	82,157.20	51	48.1	0
Total(Thousand hectares)	64,861.40	12,647.60		18,945.30		33,249.00		19.5	

forest type was by far the largest with approximately 50 million acres. The next largest forest type was in mixed upland hardwoods, with slightly less than 30 million acres. The loblolly pine/hardwood forest type was the third largest at around 15 million acres.

Using the most recent data, oak timberland area had a stand-size distribution of 19 percent seedling-sapling, 29 percent poletimber and 51 percent sawtimber. The oak/hickory forest type group summary was similar at 17, 29, and 54 percent, respectively. The largest component of this group was the white oak/red oak/hickory, a forest type common in parts of the Southern Appalachians and the Ozark Plateau (McWilliams 2002). Sawtimber accounted for the majority of the acreage in this forest type, with 61 percent of the area classified as sawtimber-sized stands and only 11 percent classified as seedling-sapling. Other individual oak-hickory forest types with percentages of the total area in seedling/saplings that were less than the oak/hickory group as a whole included chestnut oak (3 percent), post oak/blackjack oak (14 percent), white oak (2 percent), northern red oak (3 percent), bur oak (5 percent) and scarlet oak (6 percent).

The oak-pine forest type group had a more balanced stand structure, with 30 percent of the area in seedling-saplings, 30 percent in poletimber, and 40 percent in sawtimber. This group's largest component, loblolly pine/hardwood, distributed 39 percent seedling-sapling, 24 percent poletimber, and 38 percent sawtimber. The forest types with percentages of the total area in seedling/saplings that were less than the oak/pine group as a whole included white pine/red oak/white ash with 11 percent seedling-sapling, shortleaf pine/oak 1(3 percent) and Virginia pine/southern red oak (20 percent).

It is interesting to note both the higher average percentage of seedling-sapling timberland area in oak-pine forests vs. oak-hickory forests and the greater percentage of regeneration among those oak-pine types that are considered below the forest-type group average compared to oak-hickory below-average forest types. McWilliams et al. (2002) stated that these young oak-pine stands commonly convert to pine stands as the pine species outgrow their competitors, so how much of these oak-pine seedling-sapling areas will result in mature forests with a significant component of oak remains to be seen.

Oak Overstory: Status And Trends

To examine overall trends in oak forest species in the overstory, we divided the estimated eastern U.S. timberland into three categories based on FIA plot-level data: stands with oak basal area greater than or equal to 20 ft^2ac^{-1} (OAK 20 PLUS), stands with oak basal area less than that amount but greater than zero (OAK LT 20), and stands with no oak basal area (OAK ZERO) (Fig. 3). The results vary across the region, but there is a slight decline in the percentage of plots in the "oak basal area greater than or equal to 20 ft^2ac^{-1}" category and the proportion of plots with zero oak basal area is increasing. We can conclude that there is a downward trend in oak basal area but as yet no precipitous decline across the region.

We also examined upland oak overstory growing-stock volume according to individual species. Table 3 shows the volumes in descending order for species in the red and white oak groups. Northern red oak had the largest red oak volume followed by black oak. White oak had the largest white oak group volume followed by chestnut oak.

Mortality estimates are the principal indicator of health from FIA inventories and are calculated on an equivalent annual basis (McWilliams et al. 2002). Overall oak mortality was 0.79 percent (Table 3). The mortality of white oak species averaged 0.55 percent while red oak species' mortality averaged 0.98 percent of growing-stock volume. Those red oak species with the highest mortality included pin oak (1.78 percent), Nuttall oak (1.74 percent) and scarlet oak (1.34 percent). The white oak group species with the highest mortality rate was overcup oak at 0.81 percent.

McWilliams et al. (2002) noted that species with the highest mortality are subjected to the most stress agents and gave examples of American elm and Dutch elm disease and balsam fir and spruce budworm. While there could be a host of localized factors affecting mortality,

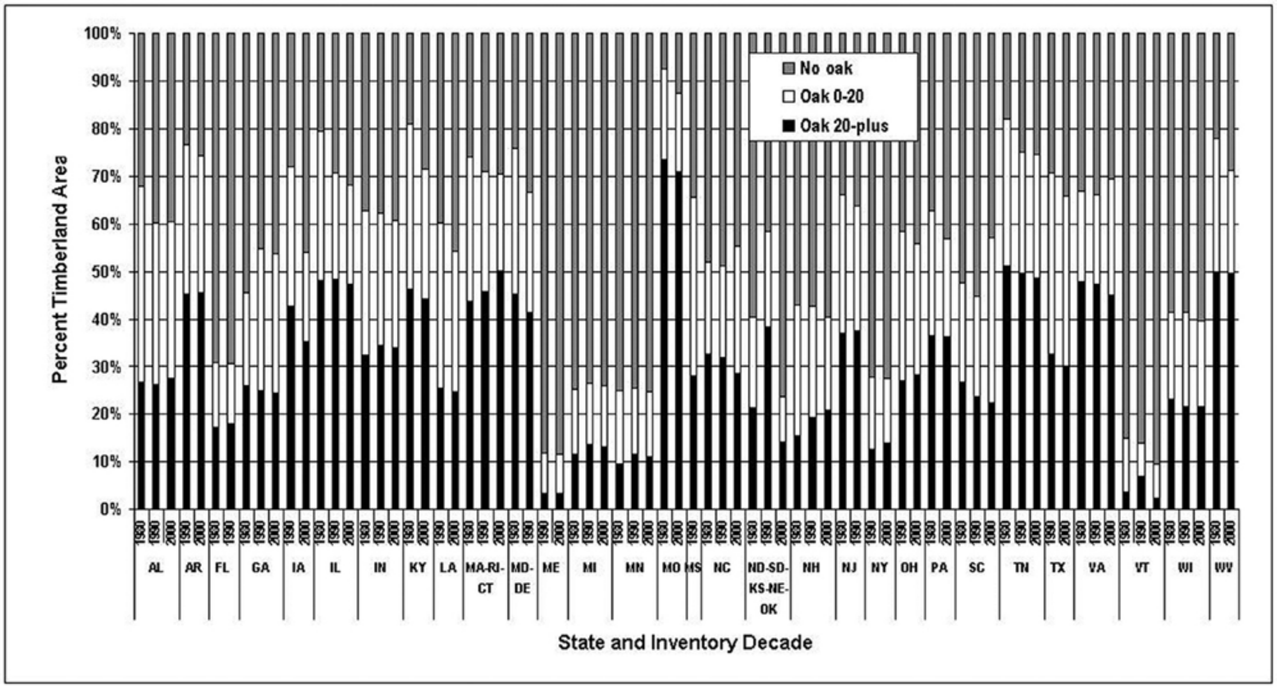

Figure 3.—Percentage of timberland area by oak category class, state and decade of inventory. Multiple inventories in one decade are averaged. “No oak” = 0 ft^2 ac^{-1} oak species basal area; “Oak 0-20” = 0-19.9 ft^2 ac^{-1} oak basal area; “Oak 20-plus” = 20+ ft^2 ac^{-1} oak species basal area.

disease situations that are density- and age-mediated, such as oak decline in Missouri (Lawrence et al. 2002), could help explain the apparent high mortality of scarlet oak and pin oak.

Net Growth and Removals

The presence of a particular species is influenced not only by environmental considerations but also by how human activity impacts the species. A useful indicator of oak resource dynamics (McWilliams et al. 2002) is the ratio of growth to removals. Ratios less than 1.0 indicate overcutting of the resource while values above 1.0 indicate inventory expansion. We examined the latest estimates of net volume growth and removal volume by species (Table 4). The ratio for all oak species was positive; species that were most closely matched in terms of growth and removals included southern red oak (1.05), Nuttall oak (0.88), willow oak (1.17), and scarlet oak (1.18), suggesting that utilization of the oak resource is regionally sustainable. The high proportion of sawtimber stands implies that net growth will be primarily from increases in current tree diameter rather than ingrowth (McWilliams et al. 2002).

Status of Proportion of Oaks in Eastern U.S. Forest Overstory

In most of those states with select red oak (Table 4) stands that met the previously-mentioned criteria, mean overstory basal area of select red oaks increased from the 1980’s through 2003. Because oak regeneration potential is at least partially related to the proportion of oaks in the overstory (Johnson et al. 2002), it would be helpful to examine trends in the proportion of oaks in the total stand basal area. As the underlying theme of this paper is “where oak is going,” we examined trends in the percentage of total basal area that is in select red oaks (Fig. 4). We looked at changes between inventories in the 1980’s vs. 1990’s, 1980’s vs. 2000’s, and 1990’s vs. 2000’s. While some states showed an increase in the proportion of overstory basal area represented by SRO, most states showed an overall decline during this period.

The situation was similar for white oaks. Regionwide, there was an increase in white oak basal area in stands with at least 20 ft^2ac^{-1} of white oak, but this basal area represents a decreasing proportion of total stand basal area (Fig. 5).

Table 3.—Growing-stock volume, mortality, growth and removals, in cubic feet, of oak species groups in the Eastern United States (excluding the Plains states); data based on the latest inventory for each state

Species	Growing-stock volume	Mortality	% Mortality	Net growth	Removals	Growth removals ratio
Red Oak						
northern red oak	20,655,454,559	153,296,330	0.74	538,477,979	329,792,788	1.63
black oak	12,630,532,249	153,531,016	1.22	332,118,409	231,452,931	1.43
water oak	8,018,169,993	83,017,464	1.04	344,814,570	257,593,597	1.34
southern red oak	6,535,917,882	56,645,858	0.87	235,162,961	224,973,434	1.05
scarlet oak	6,460,923,509	86,851,528	1.34	157,857,759	134,157,620	1.18
laurel oak	3,374,331,616	35,516,992	1.05	137,585,927	99,398,151	1.38
willow oak	3,018,642,800	35,342,884	1.17	112,112,411	95,500,704	1.17
cherrybark oak	2,695,838,154	18,276,233	0.68	114,284,906	91,056,971	1.26
pin oak	804,771,728	14,337,991	1.78	21,554,964	15,717,424	1.37
northern pin oak	709,967,448	4,309,812	0.61	21,454,369	6,079,116	3.53
Nuttall oak	634,755,257	11,015,356	1.74	12,282,211	13,956,389	0.88
Shumard oak	557,716,992	3,466,047	0.62	23,187,254	16,453,650	1.41
live oak	513,642,097	1,082,584	0.21	11,480,662	3,167,426	3.62
shingle oak	402,528,673	2,815,132	0.70	17,596,495	2,769,956	6.35
blackjack oak	280,545,827	2,851,693	1.02	10,777,416	722,612	14.91
Total red oak	67,293,738,784	662,356,920	0.98	2,090,748,293	1,522,792,769	1.37
White Oak						
white oak	28,401,475,433	139,793,091	0.49	843,535,496	568,816,692	1.48
chestnut oak	12,831,623,025	96,296,070	0.75	259,564,479	160,100,511	1.62
post oak	6,273,795,886	33,244,411	0.53	212,414,329	136,672,484	1.55
bur oak	1,936,207,204	4,064,910	0.21	93,947,682	15,584,760	6.03
overcup oak	1,522,137,119	12,326,322	0.81	40,340,051	29,043,911	1.39
chinkapin oak	904,100,004	2,287,263	0.25	28,954,892	6,237,172	4.64
swamp chestnut oak	883,000,453	5,083,805	0.58	24,765,460	20,524,158	1.21
swamp white oak	417,660,508	1,159,890	0.28	14,165,984	4,016,859	3.53
Durand oak	24,008,130		0.00	470,805		--
Delta post oak	20,239,202		0.00	833,243	514,152	1.62
dwarf post oak	7,357,431		0.00	674,570		--
Total white oak	53,221,604,395	294,255,762	0.55	1,519,666,991	941,510,699	1.61
Total oak	120,515,343,177	956,612,682	0.79	3,610,415,284	2,464,303,469	1.47

Table 4.—Select red and white oak species

Select red oaks	Select white oaks	
Northern red oak	White oak	Chinkapin
Cherrybark oak	Swamp white oak	Durand oak
Shumard oak	Swamp chestnut oak	Bur oak

Oak Past

Historical disturbance patterns such as clearcuts, fire, or land clearing contributed to oak's current dominance in upland forests (Liptzin and Ashton 1999; Rogers and Johnson 1998; Johnson et al. 2002; Van Lear and Waldrop 1989; Van Lear 1991). However, human land-use changes altered disturbance type and intensity (Abrams 2005; Smith 2005), creating environmental conditions more favorable to other species (Parker and Merritt 1995; Larsen and Johnson 1998; Smith 2005.

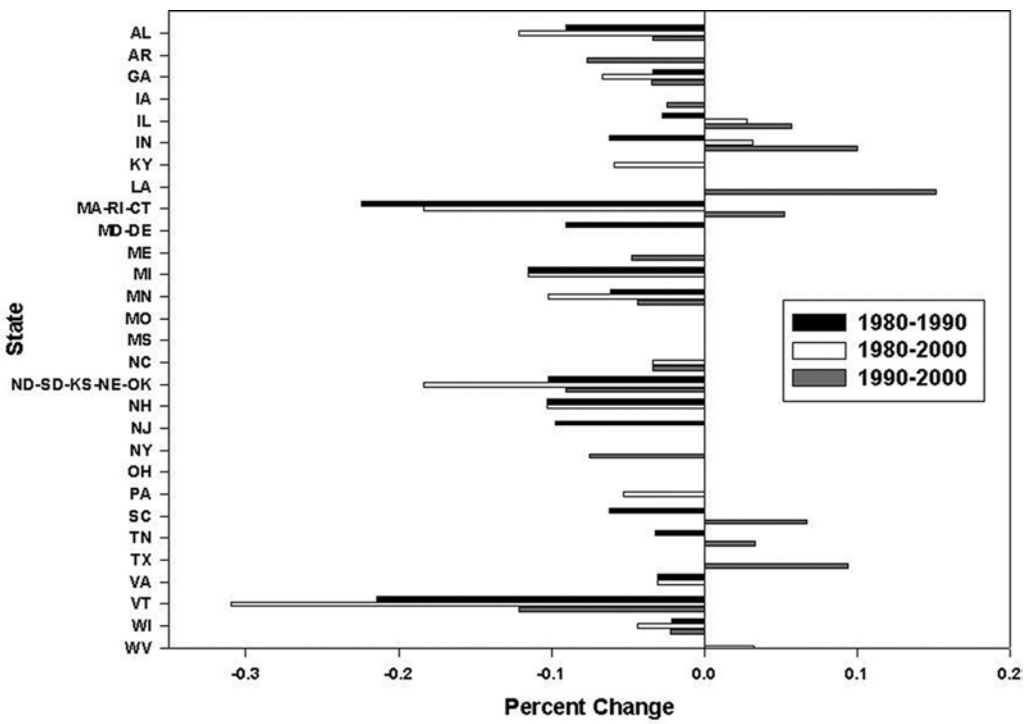

Figure 4.—Percent change in proportion of total basal area of select red oak species in stands with at least 20 ft^2 ac^{-1} of select red oaks, by state and inventory decade.

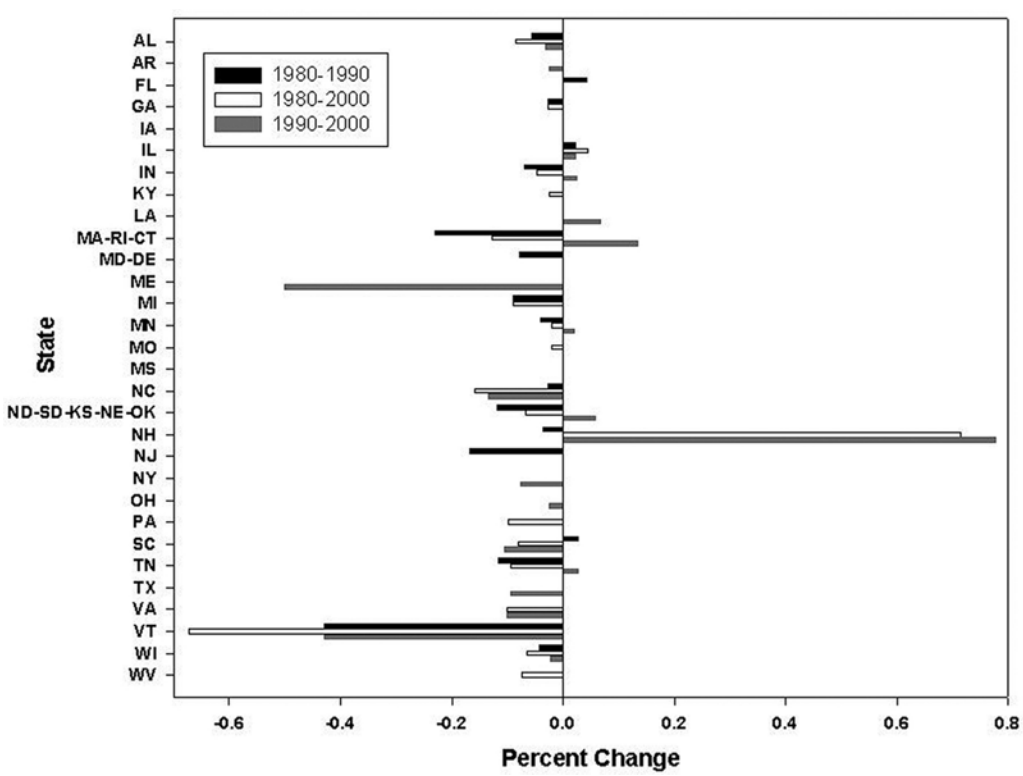

Figure 5.—Percent change in the proportion of total basal area of select white oak species in stands with at least 20 ft^2 ac^{-1} of select white oaks, by state and inventory decade.

We examined temporal trends in the oak resource as a component of total timberland acreage. The dynamic nature of property ownership in this region affects the amount of land area in the timberland base and these changes impact all species. Timberland acreage was highest in the early decades followed by a gradual decline to a low in the 1970's that was sparked by the intensification of land use, particularly the conversion to agriculture in the 50's and 60's, the decline of open space, particularly forestland, to urban development (See Smith, this volume). From 1970 to present, timberland area increased due to the gradual abandonment and reforestation of farmland, particularly during periods of economic downturn in the farm sector.

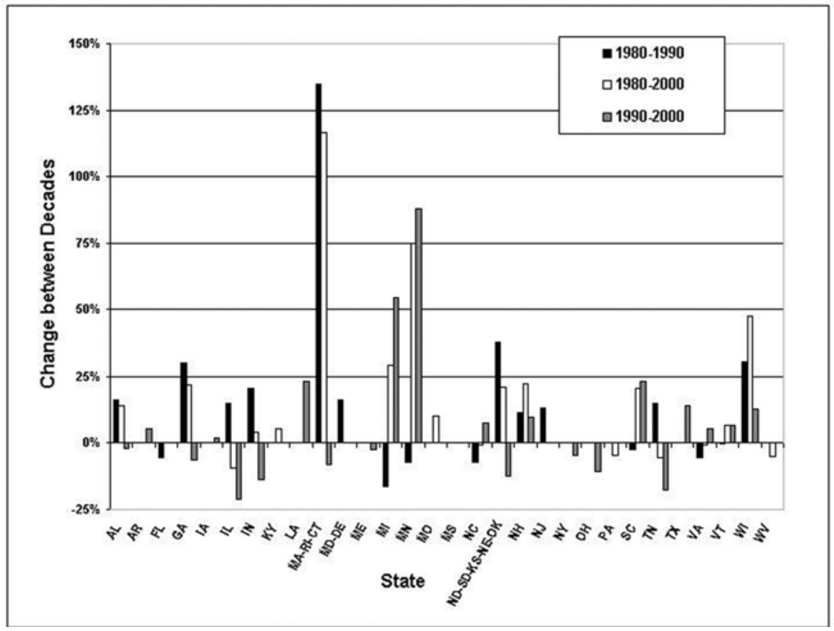

Figure 6.—Percentage change in trees per acre of all species of seedlings/saplings on all timberland, between inventory decades.

Oak Future

A common technique for regenerating oaks is the two-cut shelterwood method (Johnson et al. 2002). To take advantage of oak's life history strategy, the first cut removes enough overstory to create growing space (light on the forest floor) for regeneration. While seedlings/saplings may not grow rapidly in this partial tree shade in the understory (Johnson 1993; Larsen and Johnson 1998), the seedlings/saplings will maintain and expand a root system even while being topkilled repeatedly. With the established root systems, many of the seedlings/saplings are able to grow faster after removal of the residual overstory during the final cut of the shelterwood.

This silvicultural treatment mimics the perceived natural disturbance regime, exemplified by fire, that facilitated natural oak establishment in eastern hardwood stands (Johnson et al. 2002). Oak seedlings/saplings become established in the understory, sometimes in gaps (Ashton and Larson 1996; Rentch et al. 2003) and then undergo a series of topgrowth and dieback until some disturbance releases them to grow to the canopy.

Regeneration is heavily influenced by the size and composition of the forest overstory (Smith et al. 1996). We have seen a declining trend in the proportion of total timberland in oak overstory, with several exceptions, throughout the Eastern United States. Given the relatively long lives of certain oak species, the current oak presence likely reflects disturbance conditions far in the past. However, oak regeneration, should reflect more recent disturbances (or the lack of them). Accordingly, we looked at oak seedling/sapling data from the last one to three inventories in each state to gain insight into the future of oak forests.

The Eastern U.S. forests are not lacking for regeneration. In most states, we have observed increasing seedling/sapling densities of all species between the 1980's and 2000's (Fig. 6). Observed declines in seedling/sapling numbers occurred in certain states between inventories in the 1990s and in the 2000s, and may merely reflect the severe drought conditions in the Midwest (Lawrence et al. 2002) rather than a long-term trend.

Red Oak Regeneration

We first looked at seedlings/saplings of select red oak species in what we defined as select red oak stands. Of these species, northern red oak is the most important component of this category and apparently benefited from the large-scale anthropogenic influences of the 19th and 20th centuries (Abrams 2005) and other factors. Across the Eastern United States we observed increases or slight decreases in the number of select red oak seedlings/saplings per acre over time (Fig. 7). While several states showed dramatic gains in the number of seedlings, most of the changes were more modest.

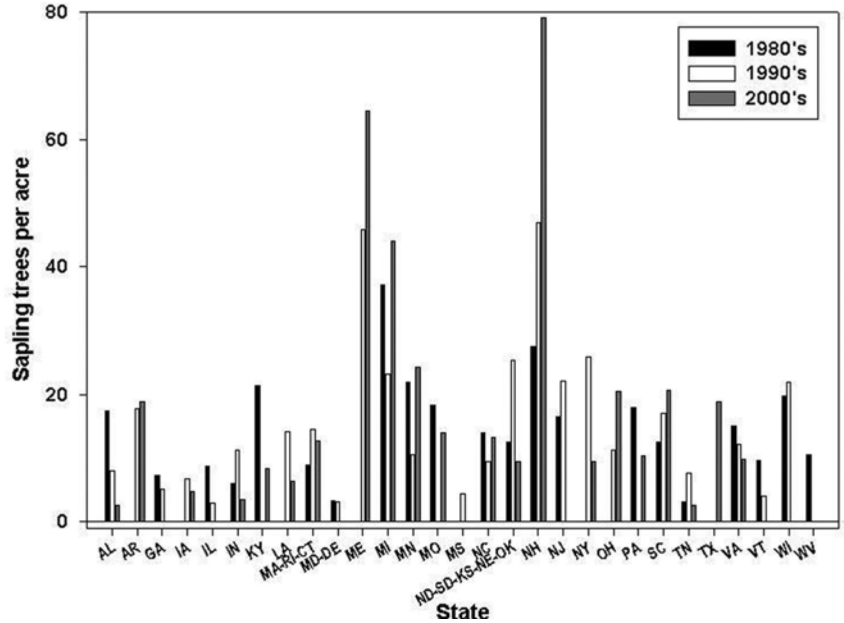

Figure 7.—Select red oak seedlings/saplings per acre, by state and inventory year.

As with the analysis of red oak in the overstory, we were interested in temporal trends in the proportion of red oak seedlings/saplings to the total number of seedlings/saplings. Across most of the states, we have observed a decline in the percentage of all seedlings/saplings represented by select red oak species over time (Fig. 8).

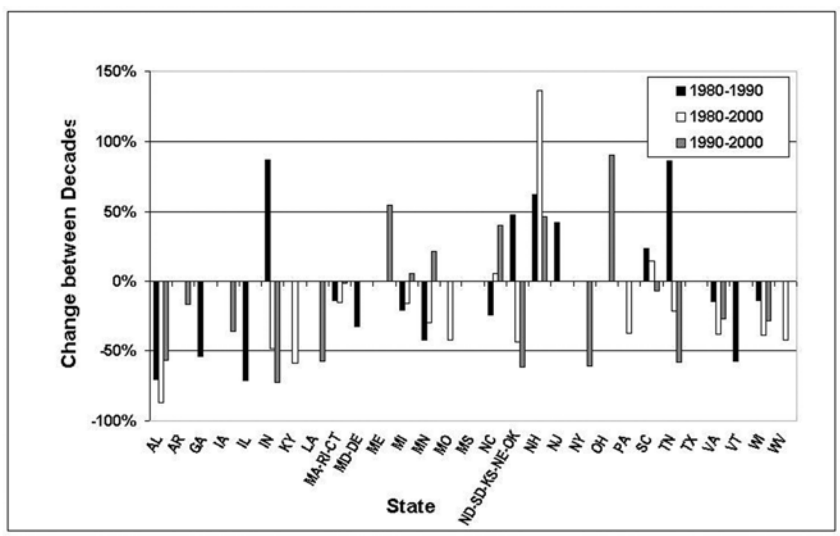

Figure 8.—Percent change in the proportion of all seedlings/saplings that are select red oak species between decades, in stands with at least 20 ft^2 ac^{-1} of select red oak overstory. For example, “1980 – 1990” compares an inventory conducted in the 1990’s with the same state’s inventory conducted in the 1980’s.

White Oak Regeneration

As with the red oak seedlings/saplings, we have white oak regeneration information from the last 20 or so years, with some states better represented than others. White oaks were an extremely important component of the pre-settlement landscape (Abrams 2005) and still have the greatest growing-stock volume of any oak species (Table 4). White oaks are more shade tolerant than red oaks, so given the increasing density of deciduous forests in the eastern U.S., we thought there might be evidence of seedling/sapling accumulation in the understory.

While the total number of seedlings/saplings in the understory of stands with select white oak basal area greater than 20 ft^2 ac^{-1} has been increasing, the proportion of all seedlings/saplings that are select white oak seedlings/saplings has been declining over the last several inventories (Fig. 9). The decline in white oak proportion, extending across the entire eastern United States and over several decades, suggests that this trend is neither temporary nor an anomaly of the data, but a representation of an established trajectory in oak regeneration.

SUMMARY

After reaching its low point in the 1960's and 1970's, timberland in the eastern U.S. has recently started to increase. The oak volume per acre of timberland has increased over the last four to five decades as overstory trees increased in size. While total oak volume has been increasing, there is a decline in the proportion of total timberland with at least 20 $ft^2 ac^{-1}$ of select red or white oaks (the "select oak" stands). Within these stands, the select red or white oak basal-area component increased slightly. While the oak basal area has been increasing, it represents a decreasing proportion of the total basal area in the stand.

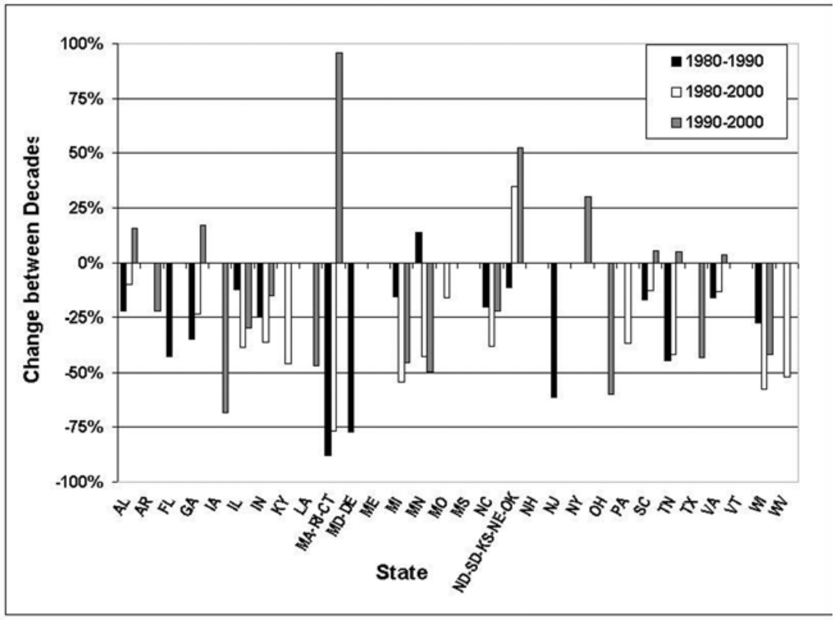

Figure 9.—Percent change in the proportion of all seedlings/saplings that are select white oak species between decades, in stands with at least 20 ft^2 ac^{-1} of select white oak overstory.

Johnson et al. (2002) detailed the importance of accumulating oak regeneration in the understory and outlines the disturbances, anthropogenic and natural, that encourage this accumulation. Such disturbances promote two processes: the concentration of early growth on the oak seedling/sapling root system resulting from repeated dieback of the above-ground component, and the elimination of less fire-resistant species that otherwise would compete successfully for resources. Two of the most prominent disturbances are harvesting and fire. Both disturbances have been declining in eastern oak forests in recent times. Contributing to decreasing proportions of oak seedlings/saplings and seedlings/saplings as reflected in our data across the eastern United States.

The declining proportion of regeneration represented by oak species is particularly disquieting as it provides a foretelling of future forest overstory composition. Given these regeneration trends, it is hard to imagine a future eastern U.S. forest landscape with the current proportion of oaks in the overstory. While reinstituting disturbances such as fire will be problematic in the highly populated landscape of this region, it should be considered part of the toolbox that resource managers use as they seek to maintain the many benefits that oak forest provides.

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WHY SUSTAIN OAK FORESTS?

David Wm. Smith[1]

Abstract.—A brief overview and some personal thoughts are offered that deal with the implications of our social and political systems on the long-term sustainability of our forest resources. The connection of the most recent climatic events, in a geologic-time context, to the development of present day oak dominated forests of the Eastern United States is discussed. The impacts of human activity and human infrastructure during the recession of the Wisconsin Glacier that began about 15,000 years ago to the present are reviewed. Changes in eastern oak forests since European settlement in the early 1600s, and more specifically in the last half century are presented in greater detail. Also discussed are important characteristics of the more that 30 oak species native to the Eastern United States, the complexity of the oak dominated eastern forests, the uniqueness of oak species for a variety of forest products and uses to satisfy human needs, and the critical importance of the oaks for wildlife food and cover. Finally, seven reasons for sustaining oak forests are presented.

INITIAL THOUGHTS

How we will achieve sustainable management of our oak forests is complex and encompasses much more that just understanding the biological aspects of trees and forests. It has become evident how differently various groups look at or perceive forests. For most foresters and land owners who grew up on farms or have been intimately associated with their forested land, we think about the evolution and changes that occur over periods of tens and hundreds of years. We plan and carry out management activities, the full results of which we may not live to see. There is a significant body of scientific knowledge and professional experience that we use to make these long-term decisions. Is this knowledge complete? Certainly not, but it is sufficient for us to move forward and be relatively sure that we are moving toward our goal of sustaining our forested ecosystems. There are several obstacles that make the road to success a bit difficult. We live in a democratic country and our government functions through a well-established political system. It is through this system that all legislation (including that related to forests, forestry and the environment) is formulated, discussed, and enacted or rejected. With this political system there are some drawbacks. We often are faced with new legislation that results in what I term "political silviculture." This is forest resource management policy legislation that often includes silvicultural practices or constraints on silvicultural practices that denies or disregards established scientific knowledge and professional experience, or when passage is influenced by issues and circumstances that are not related to the legislation, is partisan, or caters to special interests. Another concern is that there is an ever growing disconnect between what we use and where it comes from. This disconnect has worsened as we moved from a predominately rural to a predominately urban population over the past 100 years. The concept that toilet paper comes in a roll and not from a tree and that milk comes in a container and not from a cow pasturing in a field has far reaching implications when it comes to the need to practice forestry that will provide people across the country and world with the products, and values and uses they depend on and demand. We also must deal with a general public that has little understanding of the dynamics of forest establishment, growth and development. For the majority of our society forests are viewed like a picture or a "snapshot in time"—a view of a forest as an entity that is static and does not change. The fact is that the only thing constant in a forest is change.

OAK FORESTS IN THE RECENT GEOLOGIC PAST

To gain ideas and a perspective on how to develop and organize this presentation, I talked to several people who were knowledgeable about and interested in eastern hardwoods and the implications of fire on forest stand development in oak and mixed-oak stands in particular. One person suggested that I explore what may be known about oaks and oak species evolution

[1]Emeritus Professor of Forestry, Department of Forestry, Virginia Tech, Blacksburg, VA 24061, 540-552-6705, email: smithdwm@vt.edu

in terms of geologic time. I thought for a moment and the idea of tens of millions of years did not appear to be relevant to this particular conference. In fact, just thinking about what might have happened during the Pleistocene geologic epoch (1.8 million to 8000 years ago) seemed to be more than we really needed to be concerned about. When you consider that glaciers came southward an estimated 17 times during the Pleistocene and our eastern forests migrated north and south the same number of times (Goudie 1992), I do not think it is necessary to concern ourselves with what happened prior to that geologic period. What seems to be most important to the question of whether we should attempt to sustain the oak dominated forests of the Eastern United States is what has happened since the last ice age and what has been the influence of humans on the development of today's forests.

The Wisconsin Glacier, the most recent glacial period of the Pleistocene, began about 100,000 years ago and reached its southern most extent in the Eastern United States about 15,000 years ago. (Nilsson 1983; Porter 1988). If we were to look at the Eastern United States 15,000 to 18,000 years ago, the sequence from north to south would have been ice sheets/glaciers, tundra and cold steppes, then white spruce forests, followed by jack pine forests, southern pine-oak forests, and finally sand dune-scrub communities in the lower half of Florida (Bonnicksen 2000). The spruce forests extended into North Carolina (Whitehead 1967; Wright 1981) and as far south as central Missouri, Arkansas, and southeastern Oklahoma. Some hardwoods including hornbeam and northern red oak may have been interspersed with the spruce in some places (Davis 1981; Delcourt and Delcourt 1991, 1979). A band of mixed hardwoods, including oak species, bordered the lower Mississippi River and major tributaries in protected areas adjacent to the spruce forests (Delcourt and Delcourt 1979, Wells 1970). The jack pine forests were just to the south of the spruce forests and consisted of jack pine and red pine with a few temperate hardwood species scattered throughout the pine on favorable sites. The jack pine forests extended southward to central Georgia, Alabama, and Mississippi and westward to Memphis and the lower Mississippi River Valley (Delcourt and Delcourt 1991; Whitehead 1973,;Wright 1981; Davis 1981). The mixed-hardwood, and southern pine-oak forests, lying to the south of the jack pine forests and containing most of the present day oak species, were relegated to the central and northern parts of Florida, and the southern parts of Georgia, Alabama, Mississippi, Louisiana, and Texas (Delcourt and Delcourt 1981; Porter 1988; Webb 1987). The southern half of Florida was too dry and hot to support trees (Davis 1981; Wright 1981).

When considering this historical distribution of forests 15,000 to 18,000 years ago, it is evident that those mixed hardwood, oak-pine, and pine forests occurred only in the southern portion of the country and occupied only about 20 percent of the land area they occupy today. In addition, the species composition, stand structure and distribution patterns probably were different from those of the present. Since the last glaciers started to recede, there has been a dramatic migration northward coupled with a rapid evolution of species composition and community structure, and adaptation to complex climatic and geographic changes. Since the end of the last Ice Age, the only thing constant in our Nation's forests has been change. I think the following quote puts the evolution of our forests in perspective. "Forests only exist in human minds. Groups of animals and plants that we call forests come together for a short time; then each species goes its separate way when conditions change. Constant warming and cooling of the climate, and the ebb and flow of glaciers, caused the disassembly of old forests and the reassembly of new forests. Some species thrive in glacial cold while others do best in the warm periods between glaciations. So different forests dominated the land under different climates" (Bonnicksen 2000).

The Holocene geologic epoch that we are presently in, began 8,000 to 10,000 years ago (USGS 2005). It is during this warming period that the glaciers retreated, our eastern forests began their migration north, and the evolution of today's forests began. Oaks have been a part of the forests throughout this evolutionary process. As an example, oak pollen from sediment core samples of Holocene deposits in Cliff Palace Pond, a small woodland pond, located below a sandstone ridge in the northeastern part of Jackson County in southeastern Kentucky, revealed the presence of oak pollen 9,500 years ago (Ison 2000; Delcourt et. al. 1998). Oak pollen showed a general increase to about 4,800 years ago, then declined dramatically for about a thousand years and then began to increase about 3,000 years ago. The oak pollen has remained at its present high for most of the past 3,000 years, with pollen from American chestnut and pine also being significant (Delcourt et. al. 1998). Delcourt and others measured charcoal particles and the relative abundance of fire-tolerant and fire-intolerant trees and shrubs. This study and similar studies have proved to be extremely helpful in investigating the role of fire, both natural and anthropogenic, in the regeneration of oak species in the upland hardwood and mixed oak-pine forests of the Eastern United States.

HUMAN MEDIATED CHANGE IN OAK FORESTS

In a Geologic Time Context

The presence of humans and the impacts of man's activities have resulted in forests that are different than those of the previous glacial cycles when forests migrated north and south as the climate changed. Humans, migrating eastward from Siberia, across Beringia (now lying below the Bering Strait) and into Alaska, have been present in North America for at least 15,000 years (Bonnicksen 2000; Fiedel 1987; Fagan 1991, 1987). It may have taken another 2,000 to 3,000 years for these Paleoindians to work their way south and east into what is now the Eastern United States. Paleoindians probably reached southeastern Wisconsin between 13,400 and 12,300 years ago (Bonnicksen 2000; Hall 1998). During the last 8,000 to 10,000 years, as the Wisconsin ice sheet retreated, human activity has had an ever increasing influence on how hardwood forests developed as they migrated north. The First Americans migrated south as the climate warmed. By 12,000 years ago, Paleoindians had settled in Florida and had moved as far west as St. Louis, Missouri (Canby 1979; Graham et. al. 1981). People were here and they were here to stay. For the next 11,600 years, until the arrival of Europeans in the early 1600s, fire was the primary anthropogenic tool for mediating change in eastern forests. We do not know how much change was directly or indirectly attributed to human caused fire, but we are confident that fire was used and for several thousand years. With the initial European settlements along the East Coast and the rapid migration west came land clearing for farming, the construction of towns and cities, and a rapid increase in the human population of a growing nation. The building of America had begun and the forests of the East were in rapid transition—not from the gradual climate changes measured in geologic times of tens of thousands of years but from human-induced changes in a matter of decades.

Since European Settlement

Think for a moment about some of the human-caused or mediated events of the past 300 years that have modified or permanently changed the oak dominated forest landscapes of the Eastern United States. Probably the first and most widespread was the conversion of millions of acres of forest land to pasture for domestic animals and the production of agricultural crops.
The chestnut blight, a disease caused by the fungus *Cryphonectria parasitica,* virtually eliminated the American chestnut from oak-chestnut forests of the east—to the point where we changed the forest cover type name to oak-hickory. The exclusion of periodic understory fire in the past 100 years has had a significant negative influence on the recruitment of oak species and a positive influence on the recruitment of species e.g., maple, beech, and blackgum, in many central and eastern hardwood forests (Abrams 2000, 2005). One of the most interesting and perhaps a most significant event was related to the annual migration of the now extinct passenger pigeon. Little is known about the ecological impacts of the passenger pigeon and it is rarely mentioned in the context of oak forests and changes in these forests that must have occurred when one considers the almost unbelievable numbers that were present. Scott Weidensaul (1994) wrote: "In 1808, in Kentucky, ornithologist Alexander Wilson watched the sky blacken with birds—a flock he estimated at a mile wide that passed him for four hours. They were passenger pigeons, easily the most abundant species of bird in North America and likely the most numerous land bird in the world. Based on his observations, Wilson calculated the flock's number at 2.25 billion birds. They, and their kin, were the single greatest living expression of the bounty of the wooded sea." Weidensaul noted that the pigeons existed in a relatively few enormous flocks, with each flock numbering in the hundreds of millions

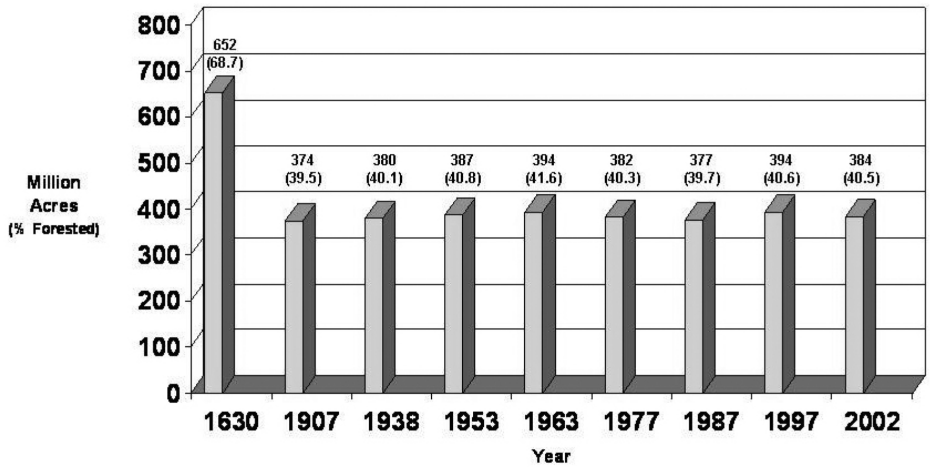

Figure 1.—Trends in U.S. forest-land area 1630-2002.

of birds. "Several billion birds feeding through a forest would reduce the mast available to other nut-eaters and would seriously reduce tree regeneration; the fractured, mangled trees, collapsed under the sheer weight of the roosting or nesting birds (to say nothing of the vast quantities of guano), would have provided a flush of sunlight and fertilizer for new plant growth, benefiting grazers and browsers like elk and deer." Passenger pigeons were still nesting by the millions in the 1870s, by the turn of the century they were almost gone from the wild, and the last known passenger pigeon died in the Cincinnati Zoo on September 1, 1914 (Weidensaul 1994). Think about this for a moment: a bird species composed of billions becoming extinct in less than 50 years. It was during this same period that major forest harvests of mature and old-growth oak forest types were taking place within the natural range of the passenger pigeon in the Eastern United States. Here were two major landscape altering events occurring at the same time—and both closely linked to oak forests. What were the individual and the combined effects of these two agents of change on the oak forests of today? It is certainly worthy of some thoughtful conversation and contemplation.

Eastern Oak Forests and Recent Trends

The total land area of the Eastern United States, which includes Minnesota, Iowa, Missouri, Oklahoma, Texas, and all states to the East, is 948 million acres, or 42 percent of the U.S. land area. There are 384 million acres of forest land in the East, or 52 percent of the total U.S. forest land area. Today, the eastern U.S. is 40.5 percent forested. How has this forest land area changed over time? In 1630 it is estimated that the eastern U.S. was nearly 70 percent forested (Fig. 1). By 1907, primarily as a result of land clearing for agricultural crops and pasture, the forest-land base had declined to 39.5 percent (652 to 374 million acres) of the total land area. It is believed that this was the low point in forest coverage in the eastern U.S. since sometime during the early Holocene epoch when the ice sheets were receding—some 8,000 to 10,000 years ago. Since the first part of the 20th century, the eastern U.S. forest-land base has remained remarkably constant at about 40 percent (Fig. 1).

The forest-land base has remained essentially constant but what about forest types and species? When looking at land area trends of the oak type groups, both upland and bottomland, over the past 50 years, there appears to have been an increase in the acreage of upland types and a decrease in the bottomland types (USDA For. Serv. 2003). The area occupied by the oak-pine type group increased by 30 percent and the oak-hickory type increased by 23 percent, while the bottomland oak-gum-cypress type declined by 17 percent (Table 1). The conversion of productive bottomland sites from forest to agriculture uses and the oak-gum-cypress types to pine plantations are significant factors in the decline in bottomland hardwood acreage. The increase in upland oak type groups likely is the result of the reversion of agricultural land back to forest cover, especially land that was cleared but proved to be marginal for agricultural uses. Overall, there was a 15-percent increase in the total acreage of the oak type groups in the past 50 years (Table 1). It is important to note that these forest-type groups are broad general classifications and species composition within a type group could vary by 15 to 20 percent or more. For example, the oak-hickory type group is defined as: "forests in which upland oak or hickory, singly or in combination, comprise a plurality of the stocking except where pines comprise 25 to 50 percent, in which case the stand is classified as oak-pine. Common associates include yellow-poplar, elm, maple, and black walnut" (USDA For. Serv. 2003). Therefore, these data tell us little about possible shifts in the oak component within these type groups.

Table 1.—Timberland area in the Eastern United States[a] of oak type groups, 2002, 1997, 1987, 1977, and 1953

Year	Oak-pine	Oak-hickory	Oak-gum-Cypress	Oak types total
	-------------------------million acres-------------------------			
2002	32.6	124.3	29.8	186.7
1997	33.4	124.0	29.3	186.7
1987	31.1	117.0	28.0	176.1
1977	34.6	108.6	26.6	169.8
1953	25.0	101.3	35.7	162.0

[a] Includes Minnesota, Iowa, Missouri, Oklahoma, Texas, and all states to the East.

An analysis of the hardwood resource data from the Forest Service's Forest Inventory and Analysis unit over the past 40 years revealed some interesting and compelling trends that may be helpful in obtaining a better understanding of what has been happening in eastern U.S. oak forests. The net volume of all hardwood growing stock on timberland has shown a remarkable increase of nearly 70 percent between 1963 and 2002 (Table 2). A significant part of this impressive net volume increase probably is related to the skewed age-class distribution in eastern hardwood forests. Major harvests of original forests occurred in the Northeast, Lake States, Appalachian Mountains, and in the South between 1850 and 1920 (Sedjo 1991) The result is a significant bulge in the 80- to 150-year age classes where forest stands are vigorous and mortality is relatively low. The rapid increases in net volume growth during the past 40 years probably will level off in the next 10 to 20 years. Since 1963, the proportion of select white and red oak net growing stock volume has remained about the same at 16 percent. However, the proportion of growing-stock volume of all other oaks has shown a decline of 16 percent. By contrast, the proportion of soft maple net volume has increased by 60 percent and that of yellow-poplar has increased by 35 percent. The proportion of net growing-stock volume of "all other hardwoods" has shown a slight decline of less than 5 percent (Table 2). The proportion of hard maple and ash (both species were included in the "all other hardwood" category of Table 1) net growing-stock volume has also increased in the past 40 years (USDA For. Serv. 2003). If we consider the oaks to be an important component of eastern U.S. hardwood forests, then the general proportional decline of oak net growing-stock volume and the proportional increase in volume of associated species in the relatively short period of 40 years should be of major concern.

THE OAKS
Species and Distribution

Quercus is the classical Latin name of the oaks and is thought to be of Celtic derivation meaning ***fine*** and ***tree*** (Little 1979). The oaks are the largest tree genus in the United States and arguably the most important. Their worth is enormous when you consider the large number of species, the vast potential for forest products, the importance for wildlife food, cover and habitat, their importance for aesthetics and scenic beauty, their functional use in urban forests, and probably most important, their ecological value as significant functional components of most forested ecosystems, especially in the Eastern United States. Little (1979) recognized 58 native oak species in the United States and Canada, and about 10 native oak shrubs. Over half of the 58 native species are found in the Eastern United States. Of the 24 oak species described in significant detail in *Silvics of North America—Hardwood, Volume 2* (Burns and Honkala 1990), 20 are considered eastern species and 4 are only found in the western United States. Smith (1993) reviewed some of the regeneration-related silvical characteristics of 31 oak species, 25 of which were

Table 2.—Net volume of hardwood growing stock on timberland in the Eastern United States[a] of selected species, 2002, 1997, 1987, 1977, and 1963

Year	Total, all hardwood species	Select white oak and red oak	Other oaksmaple	Soft poplar	Yellow-hardwoods	All other
	100 million ft^3	-------------------------Percent of all hardwood species-------------------------				
2002	327.7	15.9	18.4	10.2	6.9	48.6
1997	316.4	16.2	18.2	10.2	6.7	48.7
1987	281.4	16.0	19.9	9.2	5.8	49.1
1977	241.3	16.4	21.0	7.9	5.7	49.0
1963	193.6	15.8	22.0	6.5	5.1	50.6

[a] Includes Minnesota, Iowa, Missouri, Oklahoma, Texas, and all states to the East.

native to the Eastern United States and 6 to the western states. It is interesting that most eastern oak species occur in association with a wide variety of overstory and understory species. The combination of individual oak species and associated overstory and understory species are dependent on specific site conditions, and one or more species of oak is adapted to virtually every forest site condition that is found in the Eastern United States. In fact, if you were to overlay the ranges of just five species, northern red oak, white oak, bur oak, overcup oak and live oak, you would cover all sites in the Eastern United States except for several counties in northern New York. The overstory and understory species greatly influence oak regeneration distribution and success, and oak growth and development throughout the rotation. By contrast, the oaks of the Western United States generally are smaller in stature and in the total area that they occupy, Western oak species generally are found on more arid sites and in clumps or small pure stands. Competition from grasses and shrubs and drought often are major factors in regeneration success and subsequent seedling development on many western sites.

In general, oak species native to the Eastern United States are strong, durable, and relatively long-lived compared to associated species. Ages of 150 to 300 years for northern red oak, white oak, and chestnut oak are not uncommon, and ages up to 700 years have been recorded (Hora 1981). Very large oaks have been documented in the east. Probably the best example is a white oak that was cut in 1913 near Lead Mine, West Virginia. This giant was 13 feet in diameter 16 feet from the base and 10 feet in diameter 31 feet from the base (Clarkson 1964). That amounts to 19,200 board feet (International ¼ inch log rule) or 23,100 board feet (Doyle log rule) of lumber from the first log.

Uses and Values

The oaks serve as a raw material for a host of forest products and as such have a significant impact on local and regional economies throughout the eastern United States. The wood from oak is noted for its beauty, durability, strength, and decay resistance. Just think for a moment about the many products from oak that we take for granted, but use virtually every day: fine furniture, flooring, paneling, structural timbers, pallets, charcoal, veneer, molding, railroad ties, boxes and crates, cabinets, cooperage, boat building materials, handles, musical instruments, architectural woodwork, fuelwood, wood composites, and mulch.

The oaks also are important in the urban forest where they provide shade, food, and cover for urban wildlife, protection from wind, visual buffer zones, and aesthetically pleasing landscape components in parks, along streets, in yards, and in public areas.

A shift in forest tree species composition from the oaks to species including the maples, yellow-poplar, blackgum or beech has serious implications for many wildlife species in eastern hardwood forests. Acorns (hard mast) from oaks represent a significant food source for numerous mammals and birds, especially for fall and winter diets when sufficient food is critical for survival

(Martin et al. 1961). Wildlife, including deer, bear, wild turkey, quail, grouse, squirrel, mice and many songbirds depend on acorns for part of their diet. Some examples of these songbirds include the red-bellied woodpecker, tufted titmouse, blue jay and white-breasted nuthatch (Martin et al. 1961; Smith and Scarlett 1987; Grubb and Pravosudov 1994). Martin and others (1961) noted that acorns rate at or near the top of the wildlife food list, and listed more that 90 species of wildlife that use acorns. Acorns are a highly digestible, high-energy, low-protein food for many wildlife species (Kirkpatrick and Pekins 2002). Compared to many other seasonal food sources, acorns are relatively slow to decompose and therefore available for an extended period when other food sources are limited or absent. Acorns ripen and are dispersed in the fall and are available throughout the winter while the seeds of red maple and silver maple ripen and are dispersed in late spring and early summer. The seeds of sugar maple and yellow-poplar ripen and are dispersed in the fall.

The annual production of acorn crops is highly variable and unpredictable and varies by species, region, and environmental conditions (Koenig and Knops 2002). Since hard mast often is a critical source of wildlife food, the variability in annual acorn production is a factor in limiting wildlife populations and greatly influences periodic population fluctuations. The consequences of declining oak species composition has serious ecological implications. The results of a recent study in Pennsylvania (Rodewald and Abrams 2002) is perhaps the first study to suggest that shifts in forest composition from oak toward maples may reduce bird species richness and abundance within forest bird communities and have a negative influence on certain species. The most likely species to be impacted are long-distance migrants, residents and bark-gleaning species. Few birds consume maple seeds (Martin et al. 1961). When the food value, quantity (weight), season of dispersal (soft maples), and decomposition rates of maple seed are compared with those of acorns, the latter clearly are superior.

Other physical factors of oaks versus maples and other species, such as bark thickness, maximum tree size, and longevity, also are important when evaluating shifts in species composition. The rough and deep furrowed nature of most oaks compared to the relatively smooth bark of red maple is an important and distinct advantage for insect-foraging bird species (Jackson 1970). The generally greater tree size, longer life, and the decay resistance of the oaks are important in their enhanced value as den and cavity trees. The larger size makes them more adaptable for bear and larger animals, and the fact that they live longer and are more resistance to wind fall and other climatic events adds to the stability of wildlife species that are dependent solely or in part on den cavity trees.

For a more comprehensive discussion of the importance of oak ecosystems to wildlife and the interactions of a host of wildlife species with these oak dominated forests see *Oak Forest Ecosystems: Ecology and Management for Wildlife* (McShea and Healy 2002).

WHY SHOULD WE SUSTAIN OAK FORESTS?

There is little question that change is the only thing that is constant in forests. I believe that one of the overriding issues is not the fact that changes in oak composition and distribution are probably occurring all across the eastern forests but that the rapid rate of change is unprecedented in the context of biologic and ecologic time frames of the past. We are seeing changes in less than a single generation of oaks that we would have expected in 10 to 20 or more generations. Ecological balances are not being reached as a result of human induced changes. It is imperative that we strive to enhance our understanding of the short and long-term, direct and indirect implications of human activity on our forested ecosystems. We must identify the information needed to understand the biological processes and functions that will result in forest resource management decisions that will ensure the future biological quality and ecological integrity of the forested-land base. We must diligently identify, prioritize, plan, conduct, and analyze, research programs that will fill the gaps in our present knowledge. Above all, we must report research results and translate and communicate them to people on the ground who are working to solve real-time problems. Finally, we must ensure that we have a built-in evaluation and follow-up system so that future information needs and associated research directions can be identified and considered effectively and efficiently. Forests always will be in a state of transition and what we learned yesterday will be helpful but not totally applicable to the conditions today and in the future. We have no choice but to work toward sustaining our oak forests. We human beings have mediated global ecological changes and we have by no means grasped the magnitude of the unintended consequences of our actions. We are a significant part of the problem so we must diligently and deliberately be part of the solution. We should continue our efforts to sustain oak forests because:

1. The oaks have ecological "standing." I borrow the word "standing" from the legal profession where it implies "a connection." The oaks have been "connected" for millions of years. They have been displaced, replaced, and then returned over a notable length of geologic time. There is little question that they are of significant ecological importance in an evolutionary context.

2. There are at least 31 oak species in the Eastern United States and at least one oak species, (and often several species) is adapted to or will grow on virtually every forest site in the East.

3. The wood from most oak species is noted for its strength, durability, rot resistance, high energy value, and visual appeal that make it ideal for a host of important wood products. Think about a world without oak floors and fine furniture, railroad ties, bourbon and wine barrels, firewood, and pallets.

4. The oaks are among the longest living tree species in the East and, therefore, tend to stabilize forest communities that are subjected to outside forces—both natural and anthropogenic. The presence of oaks in forested communities tends to make those same communities more resilient.

5. The oaks provide essential habitat components for many wildlife species, including large and small mammals, birds, and reptiles. The volume and quality of food provided by acorns are essential for a host of species. The durability and size of oak cavity trees are seldom matched by associated tree species.

6. Oaks add color, shape, contrast and stability to virtually every landscape, in all four seasons and in both rural and urban settings.

7. The oak tree is a symbol of American forests. It has stature among tree species.

We should continue our efforts to sustain our oak forests and greatly expand our efforts to achieve that goal.

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Vegetative and Forest Floor Responses to Fire

ECOLOGICAL AND ECOPHYSIOLOGICAL ATTRIBUTES AND RESPONSES TO FIRE IN EASTERN OAK FORESTS

Marc D. Abrams[1]

Abstract.—Prior to European settlement vast areas of the eastern U. S. deciduous forest were dominated by oak species. Evidence indicates that periodic understory fire was an important ecological factor in the historical development of oak forests. During European settlement of the late 19th and early 20th century, much of the Eastern United States was impacted by land clearing, extensive timber harvesting, severe fires, the chestnut blight, and then fire suppression and intensive deer browsing. These activities had the greatest negative impact on the once dominant white oak, while temporarily promoting the expansion of other oaks such as red oak and chestnut oak. More recently, recruitment of all the dominant upland oaks waned on all but the most xeric sites. Mixed-mesophytic and later successional hardwood species such as red maple, sugar maple, black birch, beech, black gum and black cherry are aggressively replacing oak. At Fort Indiantown Gap in southeast Pennsylvania, periodic burning over the last 50 years resulted in sites with lower tree density and a higher proportion of overstory oak species than unburned stands. Oak saplings averaged 875 per ha in burned forests and 31 per ha in unburned forests. Red maple had overstory importance of 7 percent and 24 percent in burned and unburned stands, respectively. The leaf litter of many oak replacement species (e.g., red maple, sugar maple, black birch, beech, black gum and black cherry) is less flammable and more rapidly mineralized than that of the upland oaks, reinforcing the lack of fire. The trend toward increases in non-oak tree species will continue in fire-suppressed forests, rendering them less combustible for forest managers who wish to restore natural fires regimes. Moreover, many of the oak replacement species are now growing too large, both above and below ground, to readily kill with understory fire. This situation greatly differs from the western United States, where fire suppression during the 20th century has made a variety of conifer-dominated forests more prone to stand-replacing fire. Thus, forest supervisors in the East who wish to use fire as a management tool in oak forests need to act sooner rather than later.

KEY WORDS: historical ecology, disturbance, succession, fire suppression, oak replacement

INTRODUCTION

The increased importance of oak (*Quercus*) during the Holocene epoch was associated with warmer and drier climate and elevated fire frequency after glacial retreat (reviewed in Abrams 2002). Significant levels of charcoal influx occurred almost routinely with oak pollen in lake and bog sediments throughout the Holocene (Delcourt and Delcourt 1997). As American Indian populations increased throughout the Eastern United States so did their use of fire, land clearing, and other agricultural activities (Whitney 1994). Thus, low to moderate levels of biotic and abiotic disturbance and climate change were an intrinsic part of the Holocene ecology, resulting in a dynamic equilibrium in regional forests.

The magnitude of anthropogenic disturbances in North American forests changed dramatically following European settlement. These included extensive logging, land clearing, and catastrophic fire, followed by fire suppression and the introduction of exotic insects and diseases (Brose et al. 2001). All of these have led to unprecedented and rapid changes in forest composition and structure. This is particularly true for the Eastern United States, which has seen the extirpation of the once dominant chestnut (*Castanea dentata*) overstory from blight, loss of vast white pine (*Pinus strobus*) forests from logging followed by intense fires, a virtual cessation of oak regeneration from fire suppression and intensive deer browsing, and a rapid increase in native and exotic invasives (Keever 1953; Abrams 1992, 1998; Whitney 1994). Some authors have characterized the landscape as undergoing a near complete transformation over the last 350 years (Whitney 1994; Foster et al. 1998).

Oak was the dominant genus in the pre-European settlement forests throughout much of the Eastern United States (Abrams 1992; Whitney 1994). However,

[1]Ferguson Building, School of Forest Resources, Pennsylvania State University, University Park, PA 16802, 814-865-4901, email: agl@psu.edu.

there has been little recruitment of oak during the 20th century (Cho and Boerner 1991; Abrams et al. 1995), with the possible exception of the southwest portion of the eastern deciduous forest (parts of Missouri, Oklahoma, and Arkansas) that lacks many of the later successional, oak-replacement species (Abrams 1992). There is evidence of a dramatic decline in oak forests from presettlement to the present for much of the Eastern United States (Glitzenstein et al. 1990; Fralish et al. 1991; Whitney 1994; Abrams and Ruffner 1995). Anthropogenic impacts during the late 19th and early 20th centuries included both the height and the end of the clearcutting era, catastrophic wildfires, the beginning of the fire suppression era, and chestnut blight. In this paper I synthesize studies of land-use history, witness trees (from early land surveys), dendroecology (tree-ring studies), and fire history to investigate the major ecological and environmental changes that have occurred in the eastern deciduous forest following European settlement. Due to the large increase in non-pyrogenic, mixed-mesophytic tree species, the window of opportunity to restore natural fire regimes in eastern oak forests may be closing in the foreseeable future.

OAK ABUNDANCE IN THE PRESETTLEMENT FOREST

Oak distribution in the pre-European settlement forests of New England differed dramatically from north to south. There is little or no oak in northern New England except along the Connecticut River Valley (Table 1; Burns and Honkala 1990; Cogbill et al. 2002). However, in southern New England and eastern New York, white oak was typically the first-rank species, with composition percentages ranging from about 17 to 36 percent (Table 1). Other dominant tree species in these forests included white pine, hickory (*Carya*), chestnut, and hemlock (*Tsuga canadensis*).

Oak species occurred in the southern and central regions of the Lake States (Burns and Honkala 1990). White oak represented 19 to 26 percent of certain presettlement forests in southern and central Michigan and Wisconsin, in some regions occurring with red pine (*P. resinosa*) and white pine, as part of bur oak (*Q. macrocarpa*) savannas, or with sugar maple (*Acer saccharum*) and red oak (*Q. rubra*) (Cottam 1949; Kilburn 1960; Nowacki et al. 1990) The peak distribution for oak species in the presettlement forest clearly was in the oak-hickory, oak-pine, and former oak-chestnut regions of the Mid-Atlantic, central Appalachians and Piedmont, Midwest, and Central States (Table 1). In the Mid-Atlantic region, oak was the first- or second-rank species in 16 of 18 studies reviewed here. In 12 of these examples, white oak was the dominant species, with a frequency of 21 to 49 percent. White oak was second to black oak (*Q. velutina*; 33 percent) with a frequency of 17 to 30 percent in the Piedmont of southeastern Pennsylvania (Mikan et al. 1994; Black and Abrams 2001). In the Ridge and Valley of Pennsylvania, white oak was the first rank dominant on valley floors, but was a codominant behind pine species and chestnut oak on ridges (Nowacki and Abrams 1992; Abrams and Ruffner 1995).

In the Midwest and central regions, white oak followed by black oak were the dominant species in six of eight examples (Table 1). However, fine till soils in northeastern Ohio and western New York were dominated by sugar maple and beech (*Fagus grandifolia*), with lesser amounts (5 to 14 percent) of white oak (Seischab 1990; Whitney 1994). White oak and black oak typically grew with hickory on drier and less fertile sites throughout the region.

There is much less information on pre-European forest composition for the South and Southeast (Table 1). The few existing studies suggest that oak species were not typically dominant but did achieve frequencies of 5 to 18 percent in forests with *Magnolia*, beech, maple, pine and other oak species (Table 1). However, more numerous studies of 20th century forests and old-growth remnants farther north in the Piedmont and central and southern Appalachians suggest that oaks were a dominant in the original forest (Braun 1950, Peet and Christensen 1980, Monk et al. 1990; Barnes 1991).

OAK DECLINE FOLLOWING EUROPEAN SETTLEMENT

Significant changes in the composition of oak forests occurred in most regions from presettlement the to present. In 18 of 26 examples reviewed here, white oak experienced a decline in frequency of 10 percent or more (Table 2). Six examples reported no significant change,

Table 1.—Percent composition of witness tree species in pre-European settlement forests in the Eastern United States

Region and location	Presettlement forest composition	Reference
Northeast		
northern VT, NH	beech(32), spruce (14), maple (12), hemlock (12), oak (5)	Cogbill et al. 2002
western NY	beech (32), sugar maple (18), basswood (12), white oak (11)	Seischab 1990
eastern NY	white oak (36), black oak (15), hickory (10), elm (6)	Glitzenstein et al. 1990
central MA	white oak (27), black oak (26), pine (18), hickory (9)	Whitney and Davis 1986
central MA	white oak (20), pine (20), hemlock (10), chestnut (8)	Foster et al. 1998
CT and RI	white oak (33), hickory (10), chestnut (9),	Cogbill et al. 2002
MA	white oak (25), pine (16), maple (6), hemlock (6)	Cogbill et al. 2002
eastern NY	white oak (17), beech (16), hemlock (10), pine (9)	Cogbill et al. 2002
Lake States		
central MI	jack pine (20), red pine (19), white pine (11), white oak (2)	Whitney 1994
central MI	red pine (40), white oak (19), white pine (15), aspen (12)	Kilburn 1960
southern WI	bur oak (60), white oak (26), black oak (13)	Cottam 1949
central WI	sugar maple (37), white oak (25), red oak (16), elm (12)	Curtis 1959
central WI	pine (28), aspen (17), larch (12), white oak (10)	Nowacki et al. 1990
Mid-Atlantic		
northern NJ	white oak (34), black oak (18), hickory (15), red oak (9)	Russell 1981
northern NJ	white oak (31), hickory (25), black oak (19), chestnut (12)	Ehrenfeld 1982
northwest PA	white oak (21), beech (13), maple (17), black oak (6)	Whitney and Decant 2001
southeast PA	black oak (33), white oak (17), chestnut (15), hickory (15)	Mikan et al. 1994
southeast PA	black oak (33), white oak (30), hickory (28)	Black and Abrams 2001
southwest PA	white oak (40), black oak (9), hickory (9), dogwood (8)	Abrams and Downs 1990
central PA		
Allegheny Mts		
plateaus	white oak (26), chestnut (19), pine (19), maple (10)	Abrams and Ruffner 1995
stream valleys	hemlock (24), maple (21), white pine (15), birch (15)	Abrams and Ruffner 1995
Ridge and Valley		
ridges	chestnut oak (14), white oak (12), pine (19), chestnut (11)	Abrams and Ruffner 1995
valleys	white oak (30), pine (25), hickory (17), black oak (10)	Abrams and Ruffner 1995
ridges	pine (27), chestnut oak (18), white oak (11), chestnut (13)	Nowacki and Abrams 1992
valley	white oak (41), white pine (12), hickory (12), black oak (9)	Nowacki and Abrams 1992
eastern WV		
ridges	white oak (35), chestnut (15), chestnut oak (13), black oak (12)	Abrams and McCay 1996
valleys	white oak (23), maple (22), pine(15), basswood (10)	Abrams and McCay 1996
southern WV	white oak (24), chestnut (12), hickory (9), chestnut oak (6)	Abrams et al. 1995
northern VA	white oak (49), red oak (26), hickory (7)	Orwig and Abrams 1994
southwest VA	red oak (25), white oak (18), chestnut (9)	McCormick and Platt 1980
western Virginia	white oak (26), pine (13), chestnut oak (9), hickory (9)	Stephenson et al. 1992
Mid-west and central region		
central MO	white oak (32), black oak (11), sugar maple (9), elm (8),	Wuenscher and Valiunas 1967
eastern IL	white oak (27), black oak (18), hickory (6), elm (10)	Rodgers and Anderson 1979
southern IL		
south slopes	white oak (81)	Fralish et al. 1991
ridge tops	white oak (45), black oak (33),	Fralish et al. 1991
northeast OH		
fine till	beech (36), sugar maple (17), white oak (14)	Whitney 1994
coarse till	white oak (37), hickory (13), black oak (6)	Whitney 1994
north-central OH	hickory (34), white oak (30), bur oak (11), black oak (11)	Whitney 1994
southeast OH	white oak (40), hickory (14), black oak (12), beech (8)	Dyer 2001
South and southeast		
north FL	magnolia (21), beech (14), maple (7), white oak (5)	Delcourt and Delcourt 1977
central GA	pine (27), black and red oak (21), post oak (18), white oak (7)	Cowell 1995
southeast TX	pine (25), white oak (18), pin oak (10), red oak (9)	Schafale and Harcombe 1983
eastern AL	white oak (13), beech (9), pine (9), maple (5)	Black et al. 2002

Table 2.—**Percent frequency of present-day forest composition in the Eastern United States (examples chosen based on availability of corresponding pre-European settlement witness tree data)**

Region and location	Present day	Reference
Northeast		
MA		
Connecticut Valley	maple (30), oak (22), hemlock(15), pine (11), birch (11)	Foster et al. 1998
Pellham Hills	maple (27), oak (21), hemlock(15), birch (15), pine (11)	Foster et al. 1998
Central Uplands	maple (24), oak (23), pine(16), birch (12), hemlock (12)	Foster et al. 1998
Eastern Lowlands	oak (35), maple (23), pine(21), birch (8)	Foster et al. 1998
central MA	white pine (23), black oak (21), red oak (19), white oak (9)	Whitney and Davis 1986
eastern NY	maple (30), chestnut oak (14), red oak (10), pine (9), white oak (4)	Glitzenstein et al. 1990
Mid-Atlantic		
northwest PA	red maple (22), black cherry (14), hemlock (7), white oak (3)	Whitney and Decant 2001
southeast PA	chestnut oak (26), red maple (18), black oak (15), white oak (4)	Black and Abrams 2001
southeast PA	box elder (23), red maple (19), ash(8), elm (7), white oak (1)	Kuhn (unpublished data)
central PA		
Allegheny Mts.	red maple (35), white oak (19), red oak (11), chestnut oak (9)	Abrams and Ruffner 1995
Ridge and Valley	chestnut oak (28), red maple (14), red oak (14), white oak (13)	Abrams and Ruffner 1995
valleys	white oak (43), red maple (15), black cherry (10), pine (7)	Nowacki and Abrams 1992
ridges	chestnut oak (43), red oak (19), red maple (14), white oak (1)	Nowacki and Abrams 1992
southwest PA	red maple (30), beech (23), tulip poplar (17), white oak (5)	Abrams and Downs 1990
eastern WV	chestnut oak (15), red oak (14), red maple (12), white oak (9)	Abrams and McCay 1996
northern VA	white oak (30), hickory (13), poplar (13), dogwood (11)	Orwig and Abrams 1994
southwest VA	hickory (14), red oak (12), chestnut oak (8), white oak (5)	McCormick and Platt 1980
Midwest and Lake States		
southeast OH	white oak (15), black oak (14), tulip poplar (11), hickory (8)	Dyer 2001
northeast OH	beech (11), white oak (11), hickory (9), black cherry (8), red maple (6)	Whitney 1994
southern WI	sugar maple (28), elm (14), basswood (11), white ask (10), white oak (5)	Whitney 1994
southern WI	(savanna)white oak (34), hickory (30), black oak (24), black cherry (12)	Whitney 1994
southern WI	(savanna)white oak (54), black oak (25), black cherry (17)	Cottam 1949
central Wisconsin	red oak (46), white oak (19), red maple (16)	Nowacki et al. 1990
southern IL		
south slope	white oak (30), black oak (22), post oak (18), hickory (13)	Fralish et al. 1991
ridge top	white oak (53), black oak (17), hickory (14), post oak (7)	Fralish et al. 1991
north slope	white oak (21), red oak (22), sugar maple (13), black oak (13)	Fralish et al. 1991

whereas two cases actually showed an increase or more than 10 percent for the species. The latter examples are rather special cases that involved the conversion of bur oak savannas to closed oak forests in southern Wisconsin following Euro-American settlement and fire suppression (Cottam 1949; Whitney 1994).

The magnitude of decline in the once dominant white oak has been dramatic. For example, in central Massachusetts oak (mainly white oak) decreased in frequency by more than 20 percent in the Connecticut Valley and Eastern Lowlands (Table 2; Foster et al. 1998). In the Hudson Valley of eastern New York, white oak declined mroe than 30 percent (Glitzenstein et al. 1990). Similar declines were noted for white oak in northwest and southeastern Pennsylvania, eastern West Virginia, and northern Virginia (Orwig and Abrams 1994; Abrams and McCay 1996; Black and Abrams 2001; Whitney and

Decant 2001). However, the largest decline in white oak (from 81 percent to 30 percent) was reported on south slopes in southern Illinois (Fralish et al. 1991).

Currently, oak is the first rank tree species in only 14 of the 26 examples reviewed here compared to 24 of these examples at the time of European settlement (Tables 1 - 2). There is a greater tendency for present-day oak dominance in the Midwest and Lake States, where they increased in former bur oak savannas and logged and burned-over pine forests (Cottam 1949; Whitney 1994; Nowacki et al. 1990), or in the former prairie peninsula outside the range of red maple (Fralish et al. 1991). Apart from these exceptions, white oak generally experienced a significant decline in frequency even when it maintained the dominant ranking in modern forests (Fralish et al. 1991; Orwig and Abrams 1994; Foster et al. 1998; Dyer 2001).

Large increases in red oak and chestnut oak also occurred from presettlement to the present (Tables 1 - 2). Red oak increased from 7 to 19 percent in central Massachusetts and from 2 to 22 percent on north-facing slopes in southern Illinois (Fralish et al. 1991; Whitney and Davis 1986). Red oak has importance values of 37 to 51 percent in present forests of north-central Wisconsin, where it formerly represented less than 1 percent of the original northern hardwood-conifer forest (Nowacki et al. 1990). Increases in red oak ranging from 9 to 19 percent and in chestnut oak from 7 to 25 percent occurred in the Allegheny Mountains and Ridge and Valley of Pennsylvania and West Virginia (Nowacki and Abrams 1992; Abrams and Ruffner 1995; Abrams and McCay 1996). In western Virginia, red oak increased from 11 to 57 percent (from 1932 to 1982), while red maple increased from 1 to 11 percent (Stephenson et al. 1992). Chestnut oak increased in frequency from less than 1 percent to 26 percent in the Piedmont lowlands of southeastern Pennsylvania (Black and Abrams 2001). Red oak and chestnut oak apparently benefited from the death of overstory chestnut (on ridges), selective logging of white oak, and the intensive logging and burning of both high- and low-elevation forests.

The witness tree studies reviewed here indicate that the once dominant white oak grew on a wider range of sites and in greater numbers than any other eastern oak species (Braun 1950; Peet and Christensen 1980; Monk et al. 1990; Barnes 1991). Red oak appears frequently in the witness tree record but only occasionally had a frequency exceeding 5 percent. It is likely that there were small populations of red oak across the eastern forest on most landforms that provided adequate nutrients and some protection from fire and drought.

After the clearcut era of the late 1800s and the chestnut blight of the early 1900s, the fast-growing and opportunistic red oak expanded dramatically from its sheltered areas and grew over vast areas of the eastern forest previously dominated by chestnut, pine, and white oak (Keever 1953; Crow 1988; Nowacki et al. 1990; Barnes 1991; Stephenson et al. 1992). It appears that red oak flourished in response to large-scale anthropogenic disturbances that were much less common in the pre-European settlement forest. Moreover, the invasion of red oak into relatively undisturbed old-growth white oak forests probably was facilitated by its increase in surrounding forests that were more highly disturbed (Abrams et al. 1995; Abrams and Copenheaver 1999).

The post-settlement expansion of chestnut oak can be best explained by the loss of overstory chestnut, pitch pine (*Pinus rigida*) and white oak, and it being tolerant of severe fires that occurred during and after the major clearcut era in the late 1800s (Keever 1953; McCormick and Platt 1980; Abrams and Ruffner 1995; Abrams and McCay 1996). Despite often growing on ridge sites and being a low quality timber species, chestnut oak did not escape extensive cutting during the 19th century. It was an important fuelwood for domestic uses and the charcoal iron industry, and a source of tannin for the tanbark industry (Stephenson et al. 1992; Whitney 1994). On sites in which chestnut oak and white oak codominated in the original forest, chestnut oak was the apparent victor following the catastrophic disturbances. However, many chestnut oak forests now are being invaded by red maple (Table 2).

RISE IN RED MAPLE IN EASTERN OAK FORESTS

By far, the largest increases in species frequency on present-day upland oaks sites are from red maple (Tables

1 - 2). Red maple now represents the first or second rank dominant in 12 of the 17 examples from southern New England, eastern New York, and the mid-Atlantic region. This is even more impressive when you consider that little red maple was recorded in the presettlement forests of these areas (Table 1). Moreover, many old-growth oak forests now have abundant young red maple as a dominant tree (Abrams and Downs 1990; Mikan et al 1994; Shumway et al. 2001; Abrams et al. 1995). The dramatic rise in red maple in oak forests during the 20th century has been attributed to the extensive logging of oak in the late 19th and early 20th century, the chestnut blight, and the suppression of understory burning (Abrams 1992, 1998; Nowacki and Abrams 1992; Stephenson et al. 1992; Mikan et al. 1994). In the prairie regions of Illinois, Iowa, and Missouri, outside the range of red maple, sugar maple now is the dominant later successional, oak replacement species on mesic, nutrient-rich sites (Pallardy et al. 1988; Fralish et al. 1991).

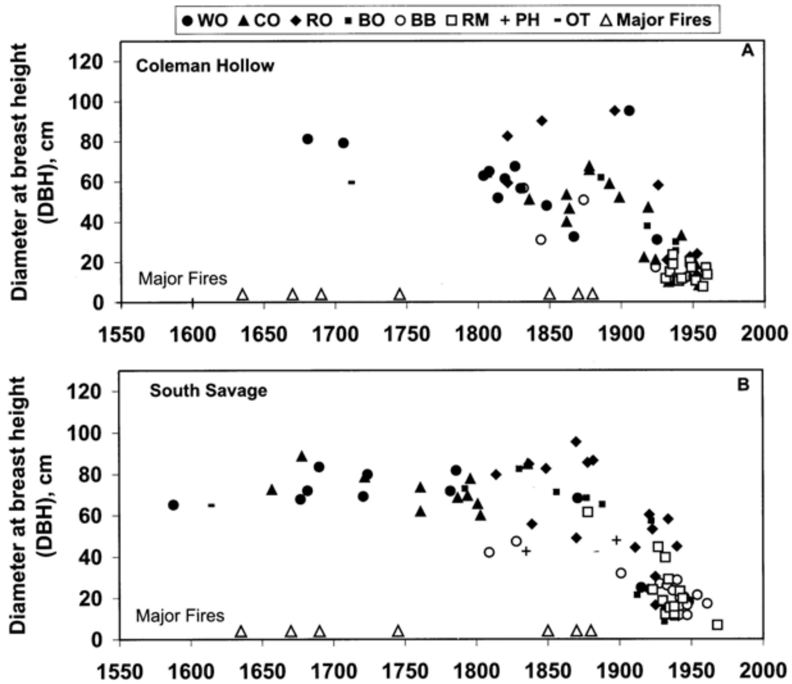

Figure 1.—Age-diameter relationships for all cored trees and the major fire years (> 25% of sampled trees scarred; indicated by triangles along x-axis) in two old-growth forests on Savage Mountain in western Maryland (adapted from Shumway et al. 2001). WO = white oak; CO = chestnut oak, BO = black oak, BB = black birch, RM = red maple, PH = pignut hickory, OT = other tree species.

The situation for red maple is somewhat analogous to the rise in red oak during the 20th century in the Eastern United States (Abrams 1998). Before European settlement, red maple generally was limited to swamps and other areas sheltered from fire. Red maple is much more sensitive to fire than red oak, and would have been less common on uplands. After 1900, red maple quickly expanded out of the protected areas and began to dominate most forest understories throughout its range. The selective logging of the highly prized white oak gave a clear opportunity to less common and/or less desirable species, e.g., red oak, red maple, black birch, and black cherry (Whitney 1994; Whitney and Decant 2001).

A CASE STUDY OF FIRE HISTORY AND OAK RECRUITMENT IN AN OLD-GROWTH OAK FOREST

Fire history and dendroecology (based on tree ring analysis) were investigated for two stands in an old-growth, mixed-oak stands in western Maryland (Shumway et al. 2001). I believe the ecological history and dynamics of these stands are representative of oak forests throughout much of the eastern forest. One stand (Coleman Hollow) is dominated by red maple (24 percent), chestnut oak (23 percent), white oak (20 percent), red oak (14 percent), and black oak (9 percent), whereas the South Savage stand is dominated by chestnut oak (20 percent), black birch (*Betula lenta*; 18 percent), red oak (17 percent), red maple (17 percent), black oak (11 percent), and white oak (6 percent). A greater abundance of rock outcroppings at South Savage may have allowed for more black birch and less white oak in the stand. The presettlement forest on Savage Mountain contained white oak (27 percent), hickory (18 percent), black oak (12 percent), chestnut oak (11 percent), chestnut (10 percent), and red oak (5 percent).

Basal cross sections were obtained from a partial timber cut in 1986, which provided evidence of 42 fires from 1615 to 1958. Fires occurred on average every 8 years during the presettlement (1600 to 1780) and early post-settlement (1780 to 1900). These included seven major fires year in which at least 25 percent of the sample trees were scarred in a given year (Fig. 1). No major fire years occurred after 1900 and no fires were recorded after 1960. The South Savage stand had a larger component of older trees, including a 409-year-old white oak, and exhibited continuous recruitment of oaks from the late 1500s until 1900. Interestingly, white oak and chestnut oak dominated recruitment from 1650 to 1800, whereas red oak and black oak dominated recruitment from 1800 to 1900. The lack of red oak and black oak recruitment prior to 1800 may be due to their relatively short longevity at the site. However, it is difficult to explain the large reduction in white oak and chestnut oak recruitment after 1800, though they might have been outcompeted by the other oaks. After 1900, red oak was the only oak species to recruit in significant numbers. This was associated with the loss of overstory chestnut from the blight.

Coleman Hollow differed from South Savage in species composition and the fact that it contained only two very old oaks—white oaks 290 and 320 years old (Fig. 1). Moreover, the abundant oak recruitment during the 19th century included large amounts of chestnut oak and white oak not seen on Savage South. From 1900 to 1950, recruitment of chestnut oak and red oak was joined by large numbers of red maple and black birch. There was little recruitment of white oak in either stand after 1900.

The results of this study indicate that periodic fires burned through the forest understories between 1600 and 1900, and that some degree of burning continued until 1960 because of its remote location. The fire rotation at Savage Mountain is consistent with mean fire intervals of 4 to 20 years in other oak forests in the Eastern and Central United States (Guyette and Dey 1995; Sutherland 1997; Schuler and McClain 2003). The long history of periodic burning at the study site was associated with continuous oak recruitment. A similar result was reported for a fire history and red oak recruitment study in West Virginia (Schuler and McClain 2003). Fires at Savage Mountain likely played an important role in oak ecology, such as preparing a thin litter seedbed, increasing sunlight to the forest floor, and suppressing red maple and black birch. Indeed, red maple and black birch were absent from the witness tree record and among the older trees in the forests, even though they can live for more than 200 years. Large amounts of red oak, chestnut oak, red maple, and birch recruitment were associated with the chestnut blight period from 1910 to 1950. A reduction and eventual cessation of fire further facilitated red maple and black birch invasion in the forest while retarding the recruitment of all oak species.

IMPACTS OF RECENT, RECURRING FIRE ON OAK FOREST COMPOSITION AND RRECRUITMENT

The National Guard Training Center at Fort Indiantown Gap near Harrisburg, Pennsylvania, has experienced frequent fires since the 1950's on the ridges and since the 1980's in the valleys as a result of military training exercises. This represented an unique opportunity to investigate the role of recent and repeated fire in oak forests in the Eastern United States. We investigated four frequently burned and two unburned sites replicated in Ridge and Valley ecosystems (Signell et al. 2006).

The cross sections collected from stand VB1 indicate that it initiated following a fire in 1941; a second fire occurred in 1953 (Fig. 2). Most cross sections had pith dates of 1953 but several individuals initiated in 1942 and one of these had a 1953 fire scar. VB1 has also experienced major fires in 1984, 1986, 1990 and 1992, with mean fire return interval (FRI) of 2.7 years for the period. The diameter distribution of VB1 is bell shaped rather than positively skewed as in VU (Fig. 3). Oak dominated every size class and shade-tolerant/fire sensitive species were uncommon. Total oak importance value exceeded 94 percent in this stand. Cross sections from stand VB2 showed recruitment dates from the early 1940s. Like VB1, this stand experienced a fire-free period following stand initiation. No fire scars were found until the early 1980s, after which at least three fires occurred, in 1982, 1985 and 1990, resulting in a mean FRI of 4 years. VB2 has a bell-shaped diameter distribution similar to that for VB1, and oak dominated all size classes (Fig. 3). However, VB2 has more shade tolerant-species present in the smaller diameter classes, including red maple and blackgum, than were observed in stand VB1. Total oak importance value in stand VB2 was 81.1 percent.

Stands RU and RB1 were part of an uninterrupted forest stand in 1952. Tree-ring data show that the

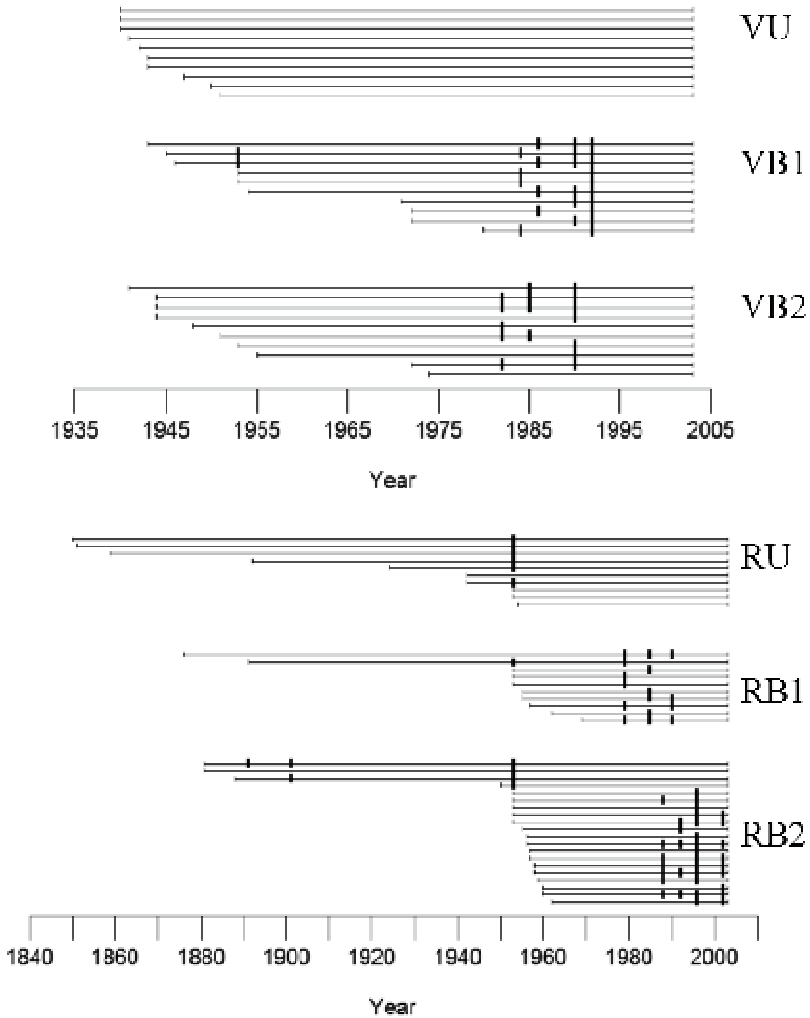

Figure 2.—Fire scar records obtained from tree basal cross sections at Fort Indiantown Gap, Pennsylvania. Horizontal lines represent the visible growth rings for each tree, and vertical bars represent fire scars (adapted from Signell et al. 2005).

oldest trees in both stands originated around 1845 (Fig. 2). No fire scars were found in any of the cross sections obtained from these stands until 1952, when both stands experienced a fire severe enough to scar many of the approximately 100-year-old chestnut oaks present at the time. Between 1952 and 1956, a road was constructed between the two stands. Following the road construction, RB1 experienced fires in 1979, 1983 and 1991, while RU has been free of fire (Fig.2).

RU had high stem density, mostly from chestnut oak, red maple, and black birch (Fig. 4), but few individuals more than 30 cm in diameter at breast height, and they were mostly chestnut oak. Nonetheless, these large trees contributed greatly to the high basal area in this stand. Red maple and black birch dominated the smaller size classes (Fig. 4). RB1 had lower stem density than stand RU. Of the major species, only scarlet oak had more stems in stand RB1 versus RU. Most black birch and red maple were concentrated in rocky patches. Chestnut oak had the highest importance in this stand; overall oak importance was 63.7 percent. Stand RB2 had several large chestnut oaks and red oaks that were clustered on a rocky boulder field. Fire occurred in 1890 and 1900, soon after stand establishment (Fig. 2). No fire scars were recorded between 1900 and 1952. Many of the smaller trees recruited following a fire in 1953. RB2 had additional fires in 1986, 1991, 1996, and 2000 (Fig. 2).

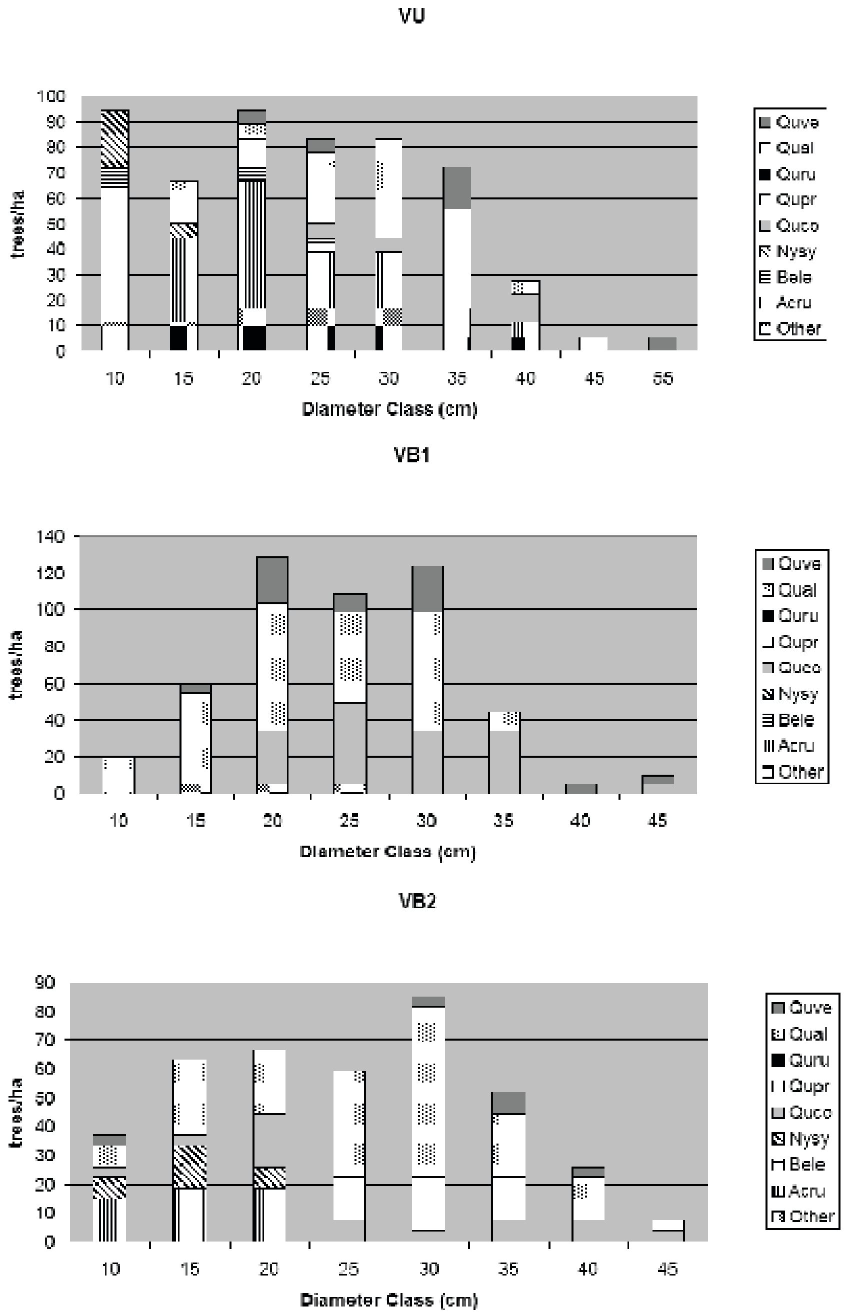

Figure 3.—Diameter distributions for unburned (VU) and burned valley stands (VB1, VB2) at Fort Indiantown Gap, Pennsylvania. Quve = *Quercus velutina*, Qual = *Quercus alba*, Quru = *Quercus rubra*, Qupr = *Quercus prinus*, Quco = *Quercus coccinnea*, Nysy = *Nyssa sylvatica*, Bele = *Betula lenta*, Acru = *Acer rubrum* (adapted from Signell et al. 2005).

RB2 has high tree density, with oak species dominating all diameter classes (Fig. 4). There were substantial numbers of red maple and black birch, primarily on rocky patches. Oak importance was 69.2 percent in this stand.

Burned sites generally had lower tree density and a higher proportion of overstory oak species (64 to 92 percent relative importance value) than unburned stands (47 to 49 percent importance). Oak saplings averaged 875 per ha in burned forests and 31 per ha in unburned forests. Red maple, the most aggressive oak replacement species in the Eastern United States, had overstory importance of 7 percent and 24 percent in burned and unburned stands, respectively. Oak saplings ranged from 824 to 1,545 per ha in three of the four burned

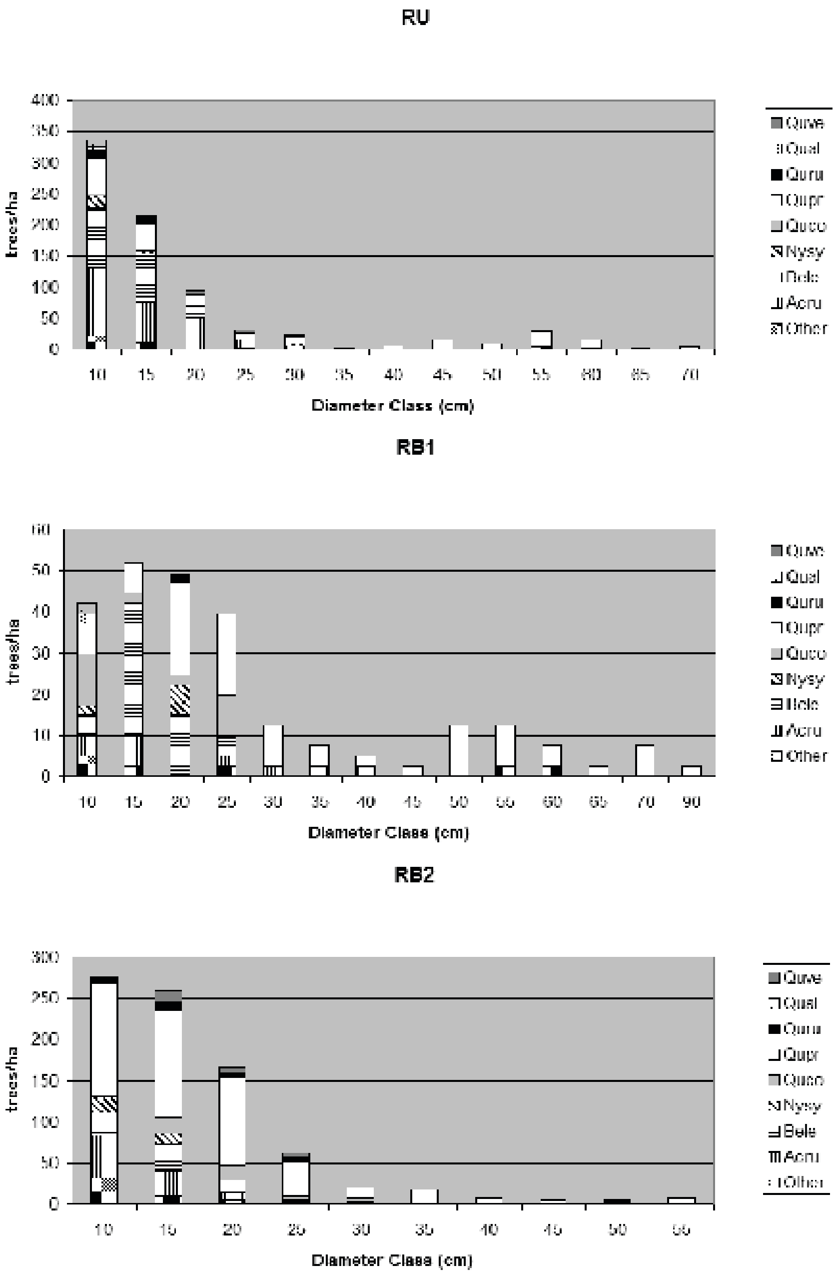

Figure 4.—Diameter distributions for unburned (RU) and burned ridge stands (RB1, RB2) at Fort Indiantown Gap, Pennsylvania. Quve = *Quercus velutina*, Qual = *Quercus alba*, Quru = *Quercus rubra*, Qupr = *Quercus prinus*, Quco = *Quercus coccinnea*, Nysy = *Nyssa sylvatica*, Bele = *Betula lenta*, Acru = *Acer rubrum* (adapted from Signell et al. 2005).

stands and 0 to 62 per ha in the unburned stands. Oak sapling density was only 62 per ha in one recently (2002) burned ridge stand that had the highest tree density. There were no red maple saplings in three of the four burned stands. Oak saplings were most abundant when overstory density was less than 400 trees per ha and understory tree density was less than 200 trees per ha. When overstory or understory tree density exceeded 400 and 200 trees/ha respectively, oak regeneration was virtually absent. The results of this study suggest that periodic fire often reduces overstory and understory stand density and promotes successful regeneration of relatively shade intolerant oak species in the Eastern United States. However, high tree density in forests will retard the development of oak understories and subsequent recruitment even if periodic burning occurs.

Table 3.—Adaptations to fire, drought and understory conditions in upland oaks relative to non-oak hardwood species (adapted from Abrams 1996)

Fire	Drought	Understory
Thick bark	Deep rooting	Low-moderate shade tolerance
Rotting resistance after scarring; tyloses in white oaks	High predawn leaf water potential	Large seed and cotyledons
Deep and extensive rooting	Thick leaves	Large initial seedling and roots
Vigorous sprouting in young trees	High leaf mass per area	Relatively high net photosynthesis
Improved fire-created seedbed	High stomatal density	Relatively low respiration
Increased germination	High leaf nitrogen	Low light compensation point
Increased seedling survival	High net photosynthesis	Medium light saturation values
Lowered seed and seedling predation	High leaf conductance	Slow seedling shoot growth
	High drought threshold for stomatal closure	High c-based phenolics
	High osmotic adjustment	Shoot dieback
	High wilting threshold	Intense seed and seedling predation
	Low nonstomatal inhibition of photosynthesis	Often overtopped by competing species
	Leaves not drought deciduous	Responsive to canopy gaps
		Increase in mixed-mesophytic hardwoods decreasing litter layer flammability

ECOPHYSIOLOGICAL ATTRIBUTES OF UPLAND OAK SPECIES

Upland oaks are prevalent across a broad array of sites, including drought-prone areas, have evolved with periodic understory burning and are transitional to later successional species in the absence of fire on most sites. Thus, upland oaks presumably possess a suite of ecophysiological adaptations for drought and fire but not for competing in a closed forest understory dominated by more shade-tolerant species. In this section I summarize the ecophysiological attributes of upland oaks as they relate to its success in the presettlement forest and subsequent decline following European settlement (Table 3).

Fire Adaptations

In an early opinion survey of the fire resistance for 22 northeastern tree species, oaks (chestnut oak, black oak, white oak, and scarlet oak) were rated in four of the top six positions (Starker 1934). A ranking of increasing bark thickness and fire resistance reported that bur oak > black oak > white oak > red oak (Lorimer 1985). White oaks (subgenus *Leucobalanus*), including bur oak, chestnut oak, and white oak, can produce tyloses, eccentric outgrowths of cell walls in response to wounding, which allow for effective compartmentalization of fire scar injuries. In this respect, white oaks should be more resistant to fire than red oaks (Table 3). Fire also may be beneficial to oaks relative to other hardwood species because of their deep and extensive rooting, vigorous sprouting ability, and increased germination and survival on fire-created seedbeds (Table 3).

Drought Adaptations

Oaks are among the most deeply rooted tree species in the Eastern United States, which allows them to maintain relatively high predawn shoot water potential from superior overnight rehydration (Table 3). Oak leaves typically are more xerophytic, having greater thickness, mass per area and stomatal density, and higher nitrogen content than leaves of non-oak species. These factors contribute to the relatively high photosynthesis and transpiration in white oak and other oak species. Oaks need to develop the necessary tissue water relations to support high level of gas exchange, and they often exhibit low osmotic potentials, low relative water content at zero turgor, and low water potential threshold for stomatal closure than non-oak species (reviewed in Abrams 1992).

Adaptations to Forest Understory Conditions

Most upland oaks are considered to have intermediate shade tolerance (Table 3; Burns and Honkala 1990), though Crow (1988) considered red oak to have low shade tolerance. Oaks produce a large acorn that allow for high initial growth, though white oaks typically exhibit low shoot growth after the first year in forest understories (Cho and Boerner 1991). Sapling density is low for most oaks, especially for white oak. This indicates that there is a severe "bottleneck" between the oak seedling and sapling stages (Crow 1988; Nowacki et al. 1990).

The physiological mechanisms for slow growth in oak seedlings are complex. Photosynthetic rates of oak in shaded conditions are often higher than in non-oak species, while oak respiration rates are low to moderate (Table 3). Oak seedlings often produce large root systems but experience recurring partial or complete shoot dieback (Crow 1988; Abrams 1992, 1996). Allocating carbon in this manner has its limitation, particularly for the once dominant white oak. White oak is reported to have slower height growth than chestnut, red oak, and black oak (Whitney 1994). Following a 1985 tornado, white oak seedling/sprouts had among the lowest rates of height growth for seven species (including black oak, red oak, and black cherry; Kauffman 2002).

Upland oaks are well adapted to dealing with a range of moisture conditions, and their moderate shade tolerance is conducive to gap-phase regeneration in fire-maintained forests. However, upland oaks grow slowly in deeply shaded forest understories dominated by later successional species and in heavily disturbed areas, where they are overtopped by competitors. Intensive deer browsing, despite the production of carbon-based defense chemicals, only exacerbates the situation (Table 3). Moreover, the increase in later successional, mixed-mesophytic hardwoods is reducing the flammability of the litter layer, which likely will result in less frequent and intense surface fires in the future. These ecological stressors represent a serious threat to the survival of upland oaks.

CONCLUSION

Paleoecological and dendroecological evidence suggest that the process of fire and oak recruitment in upland forests went on for many hundreds and thousands of years. This cycle was broken during the 19th and early 20th century, leading to a dramatic decline in the once super-dominant white oak followed by declines in other oak species. It is doubtful that oaks on mesic sites represent a true self-perpetuating climax in the absence of fire (Lorimer 1985; Abrams 1992; Schuler and McClain 2003). Thus, the broad ecological distribution of oaks in the Eastern United States can be attributed directly (probably in large part) to extensive understory burning in the pre-European and early settlement forests. If not for these fires, most oak sites would have been dominated by red maple, sugar maple, birch, beech, and black gum, a trend that now is apparent throughout most of their range.

There has been a nearly complete cessation of oak recruitment over the last 50 to 100 years in all but the most xeric and nutrient poor sites. There now is strong competitive pressure on oak regeneration from a number of later successional and gap opportunistic trees. The conversion of flammable oak litter, with high lignin content, in forest understories to less combustible and more rapidly decomposed litter of mixed-mesophytic and later successional tree species (Melillo et al. 1982; Lorimer 1985; Washburn and Arthur 2003) is rendering eastern oak forests less prone to burning. By contrast, fire suppression in the Western United States and resultant increases in live and dead fuel, stand density, and changes in species composition have made many conifer-dominated forests more prone to fire (Biswell 1967; Parsons 1976; Brown et al. 2000). The declining combustibility of eastern forests may be further exacerbated by intensive deer browsing, which has reduced the leaf litter from most woody species, especially oaks. Thus, forest managers wishing to restore historical burning regimes to eastern forests in the hope of encouraging more oak regeneration while reducing native invasive tree species should act sooner rather than later as the window of opportunity may be closing.

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RESEARCH EFFORTS ON FUELS, FUEL MODELS, AND FIRE BEHAVIOR IN EASTERN HARDWOOD FORESTS

Thomas A. Waldrop[1†], Lucy Brudnak[1], Ross J. Phillips[1], and Patrick H. Brose[2]

Abstract.—Although fire was historically important to most eastern hardwood systems, its reintroduction by prescribed burning programs has been slow. As a result, less information is available on these systems to fire managers. Recent research and nationwide programs are beginning to produce usable products to predict fuel accumulation and fire behavior. We introduce some of those tools and examine results from regional studies of hardwood fuel distribution.

INTRODUCTION

A broad array of natural disturbances has contributed to the development of eastern hardwood ecosystems. The role played by natural and anthropogenic fire has not been appreciated until recent years. There is considerable knowledge on the fire-dependent pine ecosystems of the Western United States and the Southeastern Coastal Plain. However the role of fire is not well recognized for the Central Hardwood region, Southeastern Piedmont, and Southern Appalachian Mountains. In these areas, prescribed burning programs and supporting research has lagged behind that of the West and Southeastern Coastal Plain because of the perceived damage to hardwoods, difficulty of controlling high-intensity fires on slopes, and the potential for soil/site damage (Van Lear and Waldrop 1989). Consequently, basic fire-planning information such as fuel models and fire behavior models are missing or poorly calibrated.

Although fire was never entirely missing from the eastern hardwood region, prescribed burning only began to be used in hardwood stands during the 1980's for site preparation (Phillips and Abercrombie 1987) and in the 1990's for restoration of individual species (Waldrop and Brose 1999) or entire ecosystems (Sutherland and Hutchinson 2003). Fire managers use a number of tools to describe fuels and predict fire behavior. Many of these tools were developed for western systems and have not been tested for eastern hardwoods. The purpose of this paper is to examine several existing models of fuels and fire behavior, describe several ongoing projects that will develop new models, and describe fuel characteristics specific to the eastern hardwood region.

National Projects with Hardwood Fuel Components

Models have been developed for application in numerous ecosystems throughout the United States. Probably the most well known and used in eastern hardwood systems are BEHAVE (Burgan and Rothermel 1984), FARSITE (Finney 1998), and FOFEM (Reinhardt and others 1997). Newer projects include LANDFIRE (http://www.landfire.gov), a photo series for major natural fuel types of the United States (http://www.fs.fed.us/pnw/fera/), and the USDA Forest Service Forest Inventory and Analysis (FIA) National Assessment of Fuel Loadings (Woodall 2003). We present a brief description of each that is intended as an introduction rather than an exhaustive evaluation of all available fuel and fire behavior models.

BEHAVE and BEHAVEPlus

The BEHAVE fire prediction system was developed by Forest Service researchers to predict fire spread and intensity (Burgan and Rothermel 1984; Andrews 1986; Andrews and Chase 1989; Andrews and Bradshaw 1990). Three later versions of this model (BEHAVEPlus v.1, v.2, and v.3) were developed to allow use in a Windows®-based environment and to incorporate updates in fire and modeling technology. This series of programs uses 13 fuel models that describe common fire-prone fuel types like grasslands, chaparral, coniferous forests, and logging slash (Anderson 1982). A set of 40 additional models, described by Scott and

[1]USDA Forest Service, Southern Research Station, 233 Lehotsky Hall, Clemson, SC 29634-0331; [2]USDA Forest Service, Northern Research Station, PO BOX 267, Irvine PA 16329; [†]Corresponding author, 864-656-5054, email: twaldrop@fs.fed.us.

Burgan (2005), was added in the most recent version of BehavePlus (3.0.1) that allow both static and dynamic simulations (that is, simulations that involve seasonally changing fuelbeds). The most commonly used fuel models for eastern hardwood systems are models 6 (hardwood slash), 8 (closed timber litter), and 9 (hardwood litter) (Brose 1997). The system also allows input of local fuel measurements by developing a customized fuel model. These are used for areas where fuel conditions are not adequately described by the standardized fuel models (Burgan and Rothermel 1984). A description of each version of BEHAVE and BEHAVEPlus is found at http://fire.org. Free program downloads are available at this site.

Published tests of BEHAVE in eastern hardwood systems are rare but have been presented by Brose (1997) for oak-dominated (*Quercus* sp.) shelterwood stands in central Virginia and by Grabner and others (1997) for oak savannas in Missouri. Both studies used the original BEHAVE model. No published tests of the 40 new fuel models are available for eastern hardwood systems. Brose (1997) compared actual flame length (FL) and rate of spread (ROS) measurements from winter, spring, and summer burns to values predicted by BEHAVE using fuel models 6, 8, 9, and a customized fuel model. Of the four, the customized fuel model most accurately predicted both flame length and rate of spread. Model 6 overestimated these measures of fire behavior while models 8 and 9 consistently underestimated them. The custom fuel model provided accurate predictions with r^2 values of 0.88 for flame length and 0.96 for rate of spread. Grabner and others (1997) also compared observed and predicted values for FL and ROS. They tested BEHAVE fuel models 1 (short grass), 2 (timber and grass), 3 (tall grass), 9 (hardwood litter), and a customized fuel model. In this comparison, the customized fuel model and fuel model 2 provided reasonable estimates of ROS but all models poorly estimated FL. In both studies, the authors concluded that BEHAVE was a useful tool for eastern hardwood systems, especially when site-specific fuel measurements can be used as input for a customized fuel model. When those types of measurements are impossible, BEHAVE fuel models can be substituted with a reasonable degree of confidence.

A limitation of BEHAVE is that it was developed to produce information on the intensity and rate of spread of point source fires. It does not take into consideration changes in topography, weather, or the influences of the fire itself on its environment that can affect observed fire behavior. The original 13 models are not dynamic but some of the new models have been designed that way. Currently, BEHAVE is intended to be used by an experienced fire manager who can combine the outputs with experience to develop an expectation of fire behavior.

Fire Area Simulator

The Fire Area Simulator, FARSITE (Finney 1998), is a GIS-based fire growth model originally developed for planning and management of prescribed natural fires. Its use has since expanded to suppression efforts for wildfires, evaluating fuel treatments (Finney 2001; Stephens 1998; Stratton 200;, van Wagtendonk 1996), and reconstructing past fires (Duncan and Schmalzer 2004). Developed in the Western United States, the model has been validated on fires in Yosemite, Sequoia, and Glacier National Parks (Finney 1993; Finney and Ryan 1995). While the use of this model is growing in the Southwest, Midwest, Florida, and in other countries, it has received little attention in the Eastern United States. FARSITE uses the same spread and intensity models as BehavePlus and, therefore, is subject to some of the same limitations. It requires a minimum of five raster layers to generate simulations: elevation, aspect, slope, fuel model, and canopy cover. The landscape data (elevation, slope, and aspect) are used for making adiabatic adjustments for temperature and humidity as well as for computing slope effects on fire spread and solar radiation effects on fuel moisture.

Phillips and others (2005) tested FARSITE on study sites in the Southern Appalachian Mountains that had been treated with mechanical fuel reduction (chainsaw felling of small trees and shrubs) or left untreated. They used measurements from systematically placed temperature sensors during prescribed burns to recreate test fires and compared fire behavior simulated by FARSITE with observed behavior. Initial simulations using default settings resulted in overpredictions for all fire behavior fuel models (FBFM) because the standard fuel models estimate fire behavior during the western fire season when fuel moisture contents are low (Anderson 1982). Actual fuel moisture values collected prior to burning were input into FARSITE, and subsequent simulations underpredicted fire spread for FBFM 9 and FBFM 11 but still overpredicted spread for FBFM 6. After adjustments, the rate of spread, flame length, and fire intensity for FBFM 9 appeared to adequately represent fire behavior in the leaf litter of oak-hickory (*Carya* sp.) forests. FBFM 11 seemed appropriate for modeling the mechanical treatment. For FBFM 6, adjustments allowed realistic rate of spread. However, the other variables of flame length and fire intensity were excessive, with predicted flame lengths of up to 20 m and fire intensities up to 27,000 kW/m, which were not observed during the prescribed burn. The authors suggested that FARSITE should be viewed as an option for fire modeling in the Southern Appalachians, but that work is needed on developing fuel models that better represent existing conditions of fuels in this region. In particular, fuel moisture the and presence of ericaceous shrubs presented difficulties for simulations. New fuel models introduced by Scott and Burgan (2005) were not incorporated into FARSITE at the time of this test.

First Order Fire Effects Model

FOFEM, the First Order Fire Effects Model, has been under development by the USDA Forest Service's Fire Laboratory at Missoula, Montana. It has been used widely by fire and land managers across the United States and has been endorsed by the National Wildfire Coordinating Group. FOFEM synthesizes the results of many fire effects studies into one computer program that can be easily and quickly used by novice or expert resource managers. Currently, FOFEM (v. 5.2) provides quantitative fire effects predictions for tree mortality, fuel consumption, smoke production, and soil heating from prescribed fires and wildfires. Although still under development, FOFEM provides the framework for predicting fire effects in most forest cover types listed by Eyre (1980) in the United States (http://fire.org/index.php?option=com_content&task=view&id=57&Itemid=31).

Tests of FOFEM for eastern hardwood systems have not been published, though independent use of the program suggests that mortality and litter consumption are highly overpredicted and that duff consumption is underpredicted. Although the scope of FOFEM is intended to be national, most of the models within FOFEM currently are based on western conifer forests. Mortality predictions are fitted to bark thickness and percent crown scorch relationships described by Ryan and Reinhardt (1988) for western conifers. Studies in the Southeast indicate that crown scorch is a poor predictor of mortality of southern pines (Wade and Johansen 1986; Waldrop and Lloyd 1988) and is not relevant to mortality prediction for dormant-season burns in eastern hardwoods (Yaussy and others 2004; Waldrop and Mohr 2005). Litter consumption is assumed to be 100 percent in FOFEM; this does not reflect the patchy nature of burns in eastern hardwood systems (Waldrop and others 2004). FOFEM uses models of duff consumption developed by Hough (1978) for southern pine communities but no models are available for eastern hardwoods. The Joint Fire Science Program recently funded a proposal to modify FOFEM for use in the Southeastern Coastal Plain (unpublished). However, calibration for use in eastern hardwoods has not been scheduled.

The Landscape Fire and Resource Management Planning Tools Project

The Landscape Fire and Resource Management Planning Tools Project, or LANDFIRE Project, was initiated by federal land agencies that asked principal investigators to develop maps needed to prioritize areas for hazardous fuel reduction. The objective of the project is to provide the spatial data and predictive models needed by land and fire managers to prioritize, evaluate, plan, complete, and monitor fuel treatment and restoration projects essential to achieving the goals targeted in the National Fire Plan. This project will generate consistent, comprehensive maps and data describing vegetation, fire, and fuel characteristics across the United States. The consistency of LANDFIRE methods ensures that data will be nationally relevant, while a 30-m grid resolution assures that data can be locally applicable. LANDFIRE is a multiagency project among the USDA, USDI, and The Nature Conservancy. The principal investigators are located at the Rocky Mountain Research Station's Fire Sciences Laboratory (MFSL) and the USGS National

Center for Earth Resources Observation and Science (EROS), in Sioux Falls, South Dakota) (http://www.landfire.gov/index.html).

The first step began with the selection of two prototype areas with which to develop the best methods for mapping the rest of the country (Schmidt and others 2002). Selected areas included the central Rockies of Utah and the Central Rockies of Montana, and north-central Idaho. Protocols for producing comprehensive digital maps of current vegetation composition and condition, wildland fuel, historical fire regimes, and fire-regime condition class were developed using the Utah study area as a test case. Lessons learned during this process were applied to the study area in Montana and Idaho and completed in May 2005 (Keane and others 2002). The Wildland Fire Leadership Council chartered national implementation of LANDFIRE in October 2003. A national organizational structure was developed in addition to a project plan and associated schedule. The LANDFIRE technical teams at MFSL, EROS, and The Nature Conservancy have developed strategies and procedures for building on the approaches used in the LANDFIRE Prototype. National delivery of LANDFIRE products will occur incrementally over the next 5 years as follows: Western United States in 2006; Eastern United States in 2008; and Alaska/Hawaii in 2009 (http://www.landfire.gov/ABOUT_Milestones3.html).

LANDFIRE includes a national Rapid Assessment component which will deliver a "first pass," coarse- to midscale assessment of fire regime conditions that is moderately accurate but quickly delivered. The Rapid Assessment will map and model potential natural vegetation, historical fire regimes, and fire regime condition class via existing datasets and expert opinion. The Rapid Assessment completed 12 workshops to create succession models and mapping rules for potential natural vegetation groups across the country. Participants included land managers and scientists from numerous organizations. Succession models were developed in the workshops for numerous biophysical settings defined as communities of vegetation that persist for long periods at landscape scales under natural disturbance regimes. Vegetation models currently are undergoing scientific and technical review (http://www.landfire.gov/ABOUT_Milestones2.html).

Photo Series for Major Natural Fuel Types of the United States

Most managers have little available fuels data of the extent, detail, or resolution needed for fire behavior and effects prediction. A sequence of photos called a "photo series" provides a quick, easy means for quantifying and describing existing fuel properties for selected sites across a landscape. To measure fuels in the field, the user matches observed field conditions to the fuels shown in one of the photographs. Measurements of fuels given for each photograph can be used as a reasonable estimate for that area. Photo series are available for western systems but few exist for the East (Wade and others 1993; Scholl and Waldrop 1999). Only two have been completed for the eastern hardwood region (Wilcox and others 1982; Sanders and Van Lear 1988). The Fire and Environmental Research Applications (FERA) team from the Pacific Northwest Research Station initiated a nationwide effort to produce photo series in major natural fuel types in 1995. The work was conducted in three phases. Phase I was commissioned by the USDI and covered 18 fuelbed types that were published in six volumes. Represented fuelbeds were western types with the exception of longleaf pine (*Pinus palustris* Mill.), pocosin, and marshgrass types in the Southeast. Phase II was funded by the Joint Fire Science Program in 1998. Eastern types included sand hill pines, hardwood/conifer mixtures, marshgrass, and pine/palmetto. Hardwood/conifer sites were measured in north Georgia and oak/hickory sites were measured in Kentucky and Ohio. Phase III was funded by the Joint Fire Science Program and the National Fire Plan. Work began in 2001. Eastern types included mixed hardwoods, cutover mixed hardwoods, pitch pine, and additional sand hill pines. Publications and additional information are available on the FERA website: http://www.fs.fed.us/pnw/fera/.

An interesting development in newer photo series is the digital photo series. The Natural Fuels Photo Series is designed for field use and is a source of high quality fuels data for a variety of forest and range ecosystems throughout the United States. These data were developed for on-the-ground assessments and are not fully utilized in the planning environment. Technological advances since the inception of the original Photo Series projects coupled with development of new fire- and natural resource-based software applications highlight the need to bring the Photo Series concept into the electronic age. The Digital Photo Series will be a software application that will include a fuels database with a user-friendly interface that will leverage the already high value of the Photo Series data. The Digital Photo Series is nearing completion (http://www.fs.fed.us/pnw/fera/).

FIA National Assessment

The Forest Inventory and Analysis program (FIA) of the Forest Service has been monitoring forest resources for 75 years and represents the most comprehensive, consistent, and current assessment of U.S. forests available. FIA added measurements of fuel loading to a subset of its permanent inventory plots in 2001. The inventory, known as the Down Woody Materials (DWM) Indicator, has been implemented in 38 states to provide a regional-scale estimation of fuel complexes. The process measures 1-, 10-, 100-, and 1,000-hr fuels annually using planar intercepts, fixed radius sampling, and point sampling. The goal of the project is to provide a seamless link to fire and fuel models and to serve as a robust source of fuels data for national and regional fire scientists and managers (Woodall 2003). Fuel profiles are available for Minnesota, South Carolina, and the North Central States at http://www.ncrs.fs.fed.us/4801/national-programs/indicators/dwm/.

Chojnacky and others (2004) examined data on DWM collected in 2001 by FIA on plots in several states. DWM data from 778 plots were used to compute biomass for coarse woody material, fine woody material, litter, duff, and shrub/herb cover. Regression equations were used to predict DWM components for extension to FIA's more intensive plot network. Seven regression equations were applied to the FIA data to create maps of DWM biomass. General trends showed that woody fuels and duff were most heavily loaded in the northern states and decreased from north to south. The litter layer and the shrub and herb layer followed an opposite pattern with heavy abundance in the south and decreasing to the north. The authors suggested the patterns were due to differences in climate and decomposition and that these patterns could be used to assess regional fire-fuels conditions.

Regional Projects with Hardwood Fuel Components

Hardwood fuels have been measured in numerous individual studies conducted for various purposes throughout many eastern hardwood systems. Examples include low-quality hardwood stands on the Cumberland Plateau (Muncy 1981; Waldrop 1996), pine-hardwood mixtures in the Southern Appalachians (Phillips and Abercrombie 1987), white pine-hardwood mixtures in North Carolina (Clinton and others 1998), central hardwoods in the Missouri Ozarks (Hartman 2004; Kolaks and others 2003; Kolaks and others 2004; Shang and others 2004), and mixed-oak stands in Central Ohio (Riccardi 2002; Sutherland and Hutchinson 2003). Hardwood fuels were described for studies of stand-replacement prescribed burning in the Southern Appalachians by Swift and others (1993), Waldrop and Brose (1999), and Welch and others (2000). A photo series was developed for Southern Appalachian pine-hardwood clearcuts by Sanders and Van Lear (1987). The National Fire and Fire Surrogate Study (NFFSS) (Youngblood and others 2005) has three sites in eastern hardwoods: the Ohio Hill Country, Southern Appalachians, and the Southeastern Piedmont. Fuel characteristics for these sites were described by Iverson and others (2003), Phillips and others (2005), and Waldrop and others (2004), respectively. The Great Smoky Mountains National Park is conducting a parkwide vegetation mapping project using photogrammetric and GIS techniques to support an all-taxa biodiversity inventory. As a component of this project, fuels are being mapped using the 13 fuel categories described by Anderson (1982). A fuel module was developed by He and others (2004) for the LANDIS forest landscape model (Mladenoff and He 1999) and used by Shang and others (2004) to simulate fuel dynamics in the Missouri Ozarks.

Each of these studies provides valuable information about localized fuel loads and response to one or more variables. However, there remains a lack of understanding of how hardwood fuels are distributed across the landscape, an important and beneficial concept for fire management planning. Studies by Iverson and others (2003), Kolaks and others (2004), and Waldrop and others (2004) suggest that loading of dead and down fuels is controlled by the varying inputs associated with different species and productivity levels across the landscape and varying decomposition rates at different sites. At any time since disturbance, fuel loads are a function of the amount input from dying vegetation minus the amount lost from decay. Waldrop and others (2004) also showed that fuels can be distributed across the landscape by gravity or by cultural treatments. Several studies of fuel and landscape interaction are described here.

LANDIS

He and others (2004) developed the LANDIS fuel module to simulate fuel dynamics for the Missouri Ozarks. The model simulates fuel inputs and decomposition for each of 24 land and cover types over a 200-year simulation. Species input data were from extensive inventories measured on the Mark Twain National Forest in the Missouri Highlands. Decomposition rates were those reported by Trofymow and others (2002) for Canadian forests. In a test of the model, Shang and others (2004) described fuel buildup and wildfire frequency that would occur under three fuel management treatments. A valuable product of the work is a GIS map of fuel accumulation across the study area over time. This map will allow fire managers to better predict where fuel amelioration is needed. LANDIS has been applied and tested successfully in a number of forest types in Missouri, Wisconsin, California, Canada, and China. A current effort by researchers from Texas A&M University and the Forest Service is the parameterization of LANDIS for ridgetop pine-hardwood communities in the Southern Appalachians. Although untested, local site-specific input and decomposition rates may be necessary to simulate the diverse topography of those mountains.

FORCAT

Another model designed to study dynamics of fuel loading was reported by Waldrop (1996). He used the FORCAT model (Waldrop and others 1986) to simulate inputs and decomposition of coarse fuels for xeric and mesic forest types on the Cumberland Plateau of East Tennessee. FORCAT is a gap model of forest succession that is a member of the family of models produced by Botkin and others (1972) and Shugart (1984). The simulation was set to examine fuel loads after a major disturbance; in this case, clearcutting was used. A uniform decomposition rate of 6 percent was used for both sites as suggested by Harmon (1982).

The estimated fuel load immediately after clearcutting was 49 Mg/ha on the xeric site and 69 Mg/ha on the mesic site (Fig. 1). On both sites, these levels were higher than at any other time during the 200-year simulation period. Fuels decompose rapidly in clearcuts at a period when inputs are small. In this simulation, decomposition exceeded inputs through year 32. Between years 30 and 75 there was a rapid increase in fuels for both simulated sites. FORCAT predicted decreases in stand basal area during this period as crown closure occurred and several large trees began to die. The period of rapid accumulation on the xeric site lasted until the stand was about 70 years old. Fuels continued to accumulate but at a slower rate, to a maximum at year 91. For the remainder of the 200-year simulation period, decomposition slightly exceeded inputs and fuel loads decreased gradually. Tree growth on the simulated mesic site exceeded that on the xeric site, producing a more rapid rate of accumulation (Fig. 1). On this site, fuels accumulated rapidly from years 30 through 89. Between years 90 and 200, fuel loading decreased much

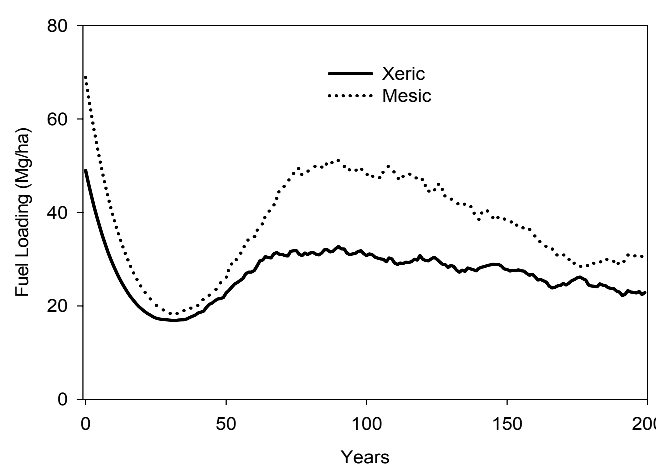

Figure 1.—Fuel dynamics after clearcutting xeric and mesic sites (predicted by FORCAT using a 6 percent decomposition rate for both sites).

more rapidly than on the xeric site. Species on the mesic site were longer lived than those on the xeric site and the trees continued to grow. Mortality was higher on the xeric site during this period due to moisture stress. Therefore, inputs were less on the mesic site than the xeric site. Loads of coarse woody fuel projected by FORCAT were similar to those reported by Muller and Liu (1991) for old-growth stands on the Cumberland Plateau.

The work of Abbott and Crossley (1982) indicates that decomposition rates are higher on moist sites. By assuming a decomposition rate of 8 percent on the mesic site and 6 percent on the xeric site, the difference in simulated fuel loads between sites was greatly reduced (Fig. 2). Even though loading was much higher on the mesic site in year 1, it decomposed to a smaller amount than the xeric site by year 32. By year 75, fuel loading again was greater on the mesic site, but beyond that point the lines converged. During the last 50 years of the simulation, fuel loads on the two simulated sites were nearly identical. This comparison (Fig. 2) illustrates the observation of Abbott and Crossley (1982) that differences in decomposition rates between sites can be more important than differences in sizes or input of woody debris. Although the mesic site produced far more biomass than the xeric site, the relatively small difference in decomposition rates (8 vs. 6 percent) produced similar fuel loading throughout the 200-year simulation. This study is limited in scope, but suggests that fuel loads may be somewhat uniform across site types.

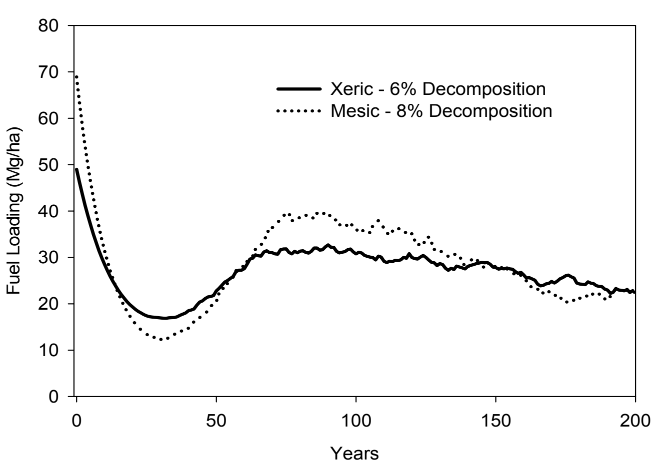

Figure 2.—*FORCAT predictions of fuel loads by site type and decomposition rate.*

Field Measures in the Missouri Ozarks

Kolaks and others (2003) collected fuels data in the Southeast Missouri Ozarks to determine whether aspect had an effect on fuel loading in previously undisturbed stands. Aspect ranges included exposed slopes (135 to 315 degrees), ridges with no aspect, and protected slopes (315 to 135 degrees). Fuel measurements were made at 15 points within each of 12 experimental units. They found no difference in loading of 1-, 10-, and 100-hour fuels or vertical structure of fuels across aspects, suggesting that the different input and decomposition rates across the landscape balance the loading of these fuels. However, the authors found significantly higher loading of 1,000-hr fuels on protected slopes. They attributed this to lower decomposition rates on moist sites, but we suggest (from experience gained from the study described below) that this is an anomaly due to the limited size of the study. The study was continued to determine whether aspect affects fuel loading after thinning and prescribed burning (Kolaks and others 2004). Both thinning and burning increased fuel loads across all aspects, though the increase followed the pattern of protected > ridge > exposed. This result agrees with the general pattern that moist sites produce greater quantities of fuel. Burns consumed fine fuels more than those more than 3 inches in diameter. Total consumption did not differ among aspects but the data suggest that a greater proportion of fine fuels was consumed on exposed slopes, likely due to differences in fuel moisture.

Field Measures in the Southern Appalachian Mountains

A large-scale study was funded by the Joint Fire Science Program in 2001 to develop hyperspectral image maps of fuel loads for study sites in Tennessee, North Carolina, Georgia, and South Carolina (http://jfsp.nifc.gov/JFSP_Project_Info_4.htm). The remote sensing component of the study will be completed by summer 2006. The field component is complete and has similarities to the studies by Waldrop (1996) and Kolaks and others (2003). Preliminary analyses were reported by Brudnak and Waldrop (2005) and Stottlemyer and others (2005). Additional analyses are presented here.

Table 1.—Fuel characteristics by slope position and aspect in the Southern Appalachian Mountains of TN, NC, GA, and SC

Slope/Aspect	Litter	1 hr	10 hr	100 hr	1,000 hr	Fuel Ht	Kalmia	Rhododendron
	--------------------------t/acre				--------------------------	Inches	Percent	Percent
Northeast								
Lower	1.68 a[1]	0.32	0.91	3.8	24.0	4.3	10.6 a	37.0 c
Upper	1.82 b	0.30	0.91	3.5	18.0	4.6	13.6 a	19.7 b
Ridge	1.83 b	0.29	1.04	4.2	16.2	4.6	13.1 a	6.1 a
Southwest								
Upper	1.75 ab	0.30	0.97	3.7	17.3	4.3	21.0 b	6.8 a
Lower	1.70 a	0.29	0.92	3.4	18.3	4.1	15.6 a	15.4 b

[1]Means followed by the same letter within a column are not significantly different at the 0.05 level.

Fuels data were collected at 250 randomly selected plots in each of four study areas (10 square miles in size). Study areas are on the Chattahoochee National Forest in northeastern Georgia, Nantahala National Forest in western North Carolina, Sumter National Forest in northwestern South Carolina, and Great Smoky Mountains National Park in southeastern Tennessee. Plot locations were generated randomly within each 10-square-mile study area using ArcView® GIS software and were stratified by slope position and aspect. Fifty plots each were located on middle and lower slopes on northeast (325 to 125 degrees) or southwest (145 to 305 degrees) aspects. An additional 50 plots were on ridgetops for a total of 250 plots in each of the four study areas. A GPS receiver was used to locate the plots in the field. Additional plots were included when necessary to provide adequate representation of all slope position/aspect combinations. Methods for measuring dead and live fuels followed standard protocols and were described by Brudnak and others (2005). The resulting dataset had measurements from 1,008 plots.

Downed woody fuels showed few differences in fuel loading across aspect/slope position plots (Table 1). Differences were observed only in the litter layer. The litter on the 1,008 sample plots tended to be heaviest along the ridges and decreased going downhill on both southwest and northeast slopes, suggesting that decomposition exceeded leaf litter inputs on the wetter sites. Although this difference among site types was significant, the relative differences were small. There was approximately 8 percent less litter on northeast lower slopes (1.68 t/acre) than on ridges (1.83 t/acre). There were no significant differences among slope/aspect combinations for loading of 1-, 10-, 100-, and 1,000-hr fuels or average fuel bed depth. These are preliminary data because analyses of impacts such as disturbance or cover type are not yet complete. However, these findings closely agree with those of Kolaks and others (2003), indicating that down woody fuels tend to be uniformly distributed across slopes and aspects.

Another component of fuels in eastern hardwood systems that must be considered is live fuel cover, particularly from ericaceous shrubs. Waldrop and Brose (1999) and Phillips and others (2005) reported a strong relationship of fire intensity to cover of mountain laurel (*Kalmia latifolia*). Van Lear and others (2002) discussed the importance of rhododendron (*Rhododendron* sp.) on the ecology of the Southern Appalachians. Although generally moist-site species, they occasionally ignite and can act as vertical fuels. In this study, both mountain laurel and rhododendron were missing from most measured plots but occurred in thick clumps where they were found. Mountain laurel was found at all aspect/slope position combinations but was significantly more abundant on southwest upper slopes (Table 1). Wildfires that might occur could reach dangerous intensities if they burned uphill on dry southwest slopes and ran into thickets of mountain laurel. Rhododendron also was present at all slope/aspect combinations but was more common at lower slope and northeast-facing plots.

Fire Behavior in Eastern Hardwoods

Prediction of fire behavior in eastern hardwood systems is complex and includes too many variables to describe in this paper. Factors such as fuel moisture, temperature, relative humidity, windspeed, wind direction, slope, and aspect contribute to this complexity in ways that are similar to other ecosystems across North America. Eastern hardwoods have several differences because most trees are deciduous. After leaffall, abundant sunlight reaches the forest floor changing the microclimate there differently than would be the case under a protective canopy of conifers. Hardwood leaves generally are flat and can be difficult to ignite once decomposition is underway. However, leaves of oaks typically curl more than do those of maple (*Acer* sp.), beech (*Fagus* sp.), and birch (*Betula* sp.), making hardwood fuelbeds more complex than those of conifers. Fires that occur in dry weather soon after leaffall often have a high rate of spread but are of low severity due to a moist duff layer.

Data and other research discussed in this paper indicate that dead woody fuels have a fairly uniform distribution across different slope positions and aspects. Although the reason for this is unknown, it is suggested that the heavy fuel production that occurs on productive sites is balanced by rapid decomposition on those same sites. Regardless, managers should be more concerned with live fuels, such as ericaceous shrubs, jackpots (localized heavy loads due to a limb or tree falling), and past disturbances rather than the overall loading of downed woody fuels. Waldrop and Brose (1999) and Waldrop and others (2005) compared fires of several intensities for successful stand replacement of Table Mountain pine (*Pinus pungens* Lamb.). They found that fires that did not burn into jackpots or mountain laurel remained relatively cool throughout a 900-acre burn. Where flames reached a jackpot, lower limbs were scorched and some tree mortality occurred. If mountain laurel cover was over 40 percent, flames climbed into the crowns of trees and consumed all of the leaves. In areas where cover of mountain laurel was more than 85 percent, flame heights were twice that of overstory trees and carried from one crown to the next.

Additional work is necessary to better understand the relationships of hardwood fuels to fire behavior. Ongoing studies include the NFFSS sites in Ohio, North Carolina, and South Carolina. Numerous temperature sensors are scattered throughout the sites to monitor fire behavior. Both groups of scientists are beginning to use sensors located at different heights above ground to produce a three-dimension view of test fires.

Studies funded by the National Fire Plan and Joint Fire Science Program are being conducted in Ohio to examine landscape-scale fire behavior. The studies entail: 1) predicting the spatial and temporal variability in key drivers of fire behavior, particularly fuel moisture and loading, and 2) monitoring fires using both airborne remote sensing and in-fire monitoring technologies. The first part of the work will provide a better understanding of fuel characteristics and better predictive capabilities as to when and how fuels will burn (that is, their frontal intensity, rate of spread, residence time, fuel consumption, and extinction). The monitoring data will be used to test and calibrate fire behavior models, including BEHAVE and a coupled fire-atmosphere model (http://www.fs.fed.us/ne/delaware/4153/4153.html).

SUMMARY

Although research on fire in eastern hardwoods has lagged behind that of other U.S. regions, there are numerous ongoing studies and several prediction products already available. The BEHAVE system tested well in two hardwood systems for predicting rate of spread, especially when customized fuel models were available. FARSITE has not been used extensively in the region but shows promise as local fuel measurements become available. FOFEM includes the computer code to allow prediction of fire effects in eastern hardwoods. However, it has not been calibrated for the region. LANDFIRE is an aggressive nationwide effort to provide fuel and vegetation maps of the entire country. Products for eastern systems should be available in 3 years. Phase III of the Photo Series for Major Natural Fuel Types of the United States is nearing completion and should include fuel models for eastern hardwoods. And the Forest Service FIA is measuring fuels across the United States and providing valuable information about regional fuel patterns.

In addition to nationwide programs, numerous local studies have been completed that describe local fuel types. Three independent studies compared hardwood fuel loads for different slope positions and aspects. All found few differences and suggested that higher productivity on moist sites was balanced by higher decomposition on those same sites. Major sources of fuel, and thus areas of concern for fire managers, were localized jackpots and presence of mountain laurel and rhododendron. Additional work is needed to better predict fuel loads throughout the eastern hardwood region and to model fire behavior within complex interactions of weather, fuel, and topography.

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SOIL, FIRE, WATER, AND WIND: HOW THE ELEMENTS CONSPIRE IN THE FOREST CONTEXT

Ralph E.J. Boerner[1]

Abstract.—Reviews our current understanding of the impact of fires typical of eastern oak forests on soil properties, soil organisms, and water quality. Most oak ecosystem fires are dormant-season fires whose intensity falls at the low end of the range of wildland fires. Direct heating of the mineral soil generally is minor except where accumulations of woody debris smolder for lengthy periods. Considerable proportions of nitrogen, phosphorus, and cations may be lost during fire through a combination of volatilization and ash convection. Post-fire precipitation events govern the return of nutrients in ash to the mineral soil, and the interaction of the soil exchange capacity, geomorphology, and weather control the proportion of nutrients from ash that will be retained for later plant use. Exposure of mineral soil by fire may lead to increased sheet erosion, but soil hydrophobicity does not seem to be important in oak ecosystems. Nitrogen availability and organic carbon content of soils may increase after fire, though both appear to be lesser in magnitude and duration than in other ecosystems. Impacts on fungi, bacteria, and microarthropods in the mineral soil are small, whereas those in the forest floor are proportional to the degree of consumption and the extent of heating due to smoldering woody fuels. Microbial activity and microarthropod populations recover quickly except after repeated annual burning. Little is known of effects of fire on other groups of soil organisms (e.g., nematodes and earthworms). The geomorphology of much of the eastern oak region is complex and heterogeneous. In such terrain, the difficulties inherent in scaling up plot-based studies to land areas of management scale are considered in the context of demonstrated landscape-scale variations in belowground effects of fire. GIS-derived landscape-scale metrics can be used to help generate management-scale recommendations from smaller scale research studies.

INTRODUCTION

Analysis of pollen and charcoal deposits taken from water bodies along the plateaus of the Appalachian Mountains demonstrates that fire has been a frequent and consistent part of the forests, savannas, and grasslands of Eastern North America for at least the last 4,000 years (Delcourt and Delcourt 1997). Further, evidence from macrofossils and pollen indicates that forests dominated by fire-dependent trees such as oak (*Quercus*) and pine *(Pinus)* have covered much of Eastern North America for at least 10,000 years (Delcourt and Delcourt 1987). Thus, the history of fire in eastern forests precedes the development of significant human populations in the region.

During and after European settlement, many observers noted the use of fire by Native Americans. Whitney (1994) summarized more than 20 historical references to the use of fire by Native Americans in oak forests and savannas, from Massachusetts to Missouri, and from the early 1600's to the early 1800's. Some of these early observers even commented on the positive (Lorain 1825) or negative (Coxe 1794; Lorain 1825) effects of fire on forest soil productivity.

The key role that fire plays in the ecology of oak-dominated forests was recognized by ecologists as early as the 1920's and 1930's (e.g., Daubenmire 1936; Cottam 1949). Ironically, this is the same time period in which organized fire suppression policies came into force and became effective, at least in Ohio (Sutherland and others 2003).

Quantitative studies of the effects of fire on the soils of oak forests, and on the waters draining such forests, did not become common until after World War II. Thus, current and recent studies of the effects of fire on forest soil and water have taken place not in the context of the long, continuous history of fire that preceded the 20th century but in the artificial context of forests already affected by many decades of fire suppression.

[1]Department of Evolution, Ecology, and Organismal Biology, The Ohio State University, Aronoff Laboratory, 318 West 12th Avenue, Columbus, OH 43210, 614-292-6179, email: boerner.1@osu.edu.

FIRES IN OAK FORESTS

The term "forest fire" conjures images of fires in western pine forests, with flames rising 30 meters or more, trees exploding, and fire storms carrying burning brands for miles. Although such intense fires can occur in our region, especially during lengthy droughts in forest types with a significant pine component (Forman and Boerner 1981), such intense, stand-replacing fires are uncommon in oak forests. Most fires in the oak forests of the central hardwoods occur during the dormant seasons (spring and autumn). Such fires typically move along the forest understory without climbing into the canopy (Sutherland and others 2003).

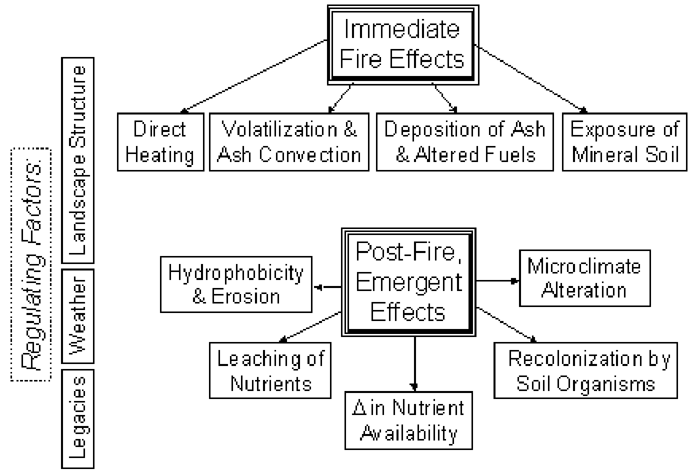

Figure 1.—Schematic of immediate fire effects and post-fire emergent effects on soil physical, chemical, and biological properties, with regulating factors indicated at left.

Fire has both immediate, direct effects as well as delayed effects on soil, soil organisms, and water that emerge in the days, weeks, and months after fire (Fig. 1). The degree to which those effects are felt is, in turn, regulated by weather, by the structure of the landscape, and by the legacies of past management, fire history, and land use (Fig. 1). In reviewing the available information on these topics I will rely where possible on studies from eastern oak forest ecosystems, and will supplement with studies from other temperate ecosystems (particularly pine forests and shrublands) only where there is insufficient direct evidence from eastern oak forests. Those interested in reviews of fire effects on ecosystems in general may consult recent, broadly based reviews, such as those of Neary and others (1999), Ice and others (2004) and Certini (2005).

SOIL HEATING

As a generalization, surface fires moving across the floor of a forest do not present a strong potential for severe soil heating because only a small percentage of the heat generated by the fire is partitioned downward into the soil. Further, mineral soil is a poor conductor of heat, especially when the soil is relatively dry (Raison 1979).

How much the temperature of the mineral soil increases during fires is the key to understanding the impact of the fire on soil organisms. Temperatures of 70°C for as little at 10 minutes will kill almost all fungi and soil microfauna, and also a significant proportion of soil bacteria (Lawrence 1956). Temperatures of 60°C are considered lethal to plant roots (Steward and others 1990) and temperatures above 60 °C denature enzyme systems of metabolically active cells of most microorganisms.

In areas where a distinct humus layer is absent or where that forest floor layer does not begin to smolder, insufficient heat is transferred downward into the mineral soil for temperatures to rise above ~60°C at depths below several centimeters (Anson and Gill 1976; Steward and others 1990). Saa and others (1993) measured soil temperatures at a depth of 5 cm in the mineral soil during surface fires in pine forests and shrublands in Spain. They did not record soil temperatures ≥ 50°C either during or after fire. Similarly, Hayward (1938) recorded soil temperatures during a fire in a longleaf pine (*Pinus palustris* Mill.) forest in North Carolina and found that temperatures at 2.5 cm into the mineral soil did not exceed 40°C at any time during or after the fire. Thus, fires moving across the forest floor are unlikely to produce increases in mineral soil temperatures sufficient to cause significant soil organism mortality.

One situation in which temperatures may rise sufficiently to induce considerable mortality is when localized concentrations of organic materials, either as woody debris (Miller and others 1955; Busse and others 2004) or as a particularly thick, dry humus layer (Miyanishi 2001), smolder in place for an extended period. Miller and others (1955) monitored surface and mineral soil temperatures during and after a fire in a New Zealand shrubland. aThere was no significant change in temperature at a depth of 5 or 10 cm in the mineral soil under this fire as it moved across the site. However, in areas where localized wood heaps burned for several hours, sensors recorded maximum temperatures of 100°C at 5 cm and 60°C at 10 cm into the mineral soil. Similarly, temperatures of 100 to 200°C have been recorded under smoldering *Arctostaphylos* residues at a depth of of 7.5 to 12.5 cm in a mixed-conifer forest in California (Busse and others 2004). Thus, localized heavy fuel accumulations such as slash piles from harvesting or detrital accumulations from storm damage present a situation in which direct heating effects may have significant negative effects on soil biota, including plant roots.

VOLATILIZATION AND ASH CONVECTION

Two pathways by which nutrients may be lost from an oak forest during and shortly after fire are direct volatilization (conversion of elements in fuel to gas phase) and ash convection (conversion of elements to solid, inorganic ash subject to wind action). Which nutrients are subject to being volatilized depends on the element's chemical properties, the properties of the compound in which that element is found, and the behavior of the fire. Similarly, losses via ash convection depend on the local weather conditions, the behavior of the fire, and the characteristics of the fuels being combusted.

Under controlled laboratory conditions, organic compounds containing nitrogen (N), phosphorus (P), and sulfur (S) often begin to volatilize when temperatures approach or exceed 200°C (Table 1). The temperatures reached in the fuelbed (~650°C) during flaming or smoldering after the flames have passed are considerably higher than that critical temperature. This helps explain why the amount N lost by volatilization is so closely dependent on the amount of fuel consumed (e.g., Raison and others 1985).

One of the few estimates of the loss of N via this pathway in oak forests is the report of Hubbard and others (2003) of losses of 55 kg N/ha from litter + wood and 6 kg N/ha from humus as the result of prescribed fires in oak-pine stands in Georgia and Tennessee. They indicated that losses of this magnitude could be replaced by atmospheric deposition of N in a relatively short time.

One also can gain perspective on the potential importance of this pathway of N loss from estimates generated in other temperate ecosystems dominated by woody plants. Losses of N from vegetation and detritus to the atmosphere have been estimated at 70 percent in a North Carolina longleaf pine savanna (Christensen 1977), 33 percent in a mixed pine plantation in South Carolina (Wells 1971), 39 percent in a mixed conifer forest in Washington (Grier 1975), and 39 percent in a chaparral shrubland in California (DeBano and Conrad 1978). In general, losses of N to the atmosphere are proportional to fuel mass loss during fire (Raison and others 1985).

Compared to N, base cations such as calcium (Ca), magnesium (Mg), and potassium (K) and inorganic forms of P have considerably higher laboratory volatilization temperatures (Table 1). Nonetheless, losses of cations and P from vegetation and detritus during a fire can be significant. Estimates of Ca loss range from 11 percent in a Washington mixed conifer forest (Grier 1975) to 15 percent in a longleaf pine savanna (Christensen 1977), to 17 percent in a mixed-eucalypt forest (Harwood and Jackson 1975). The same three studies report losses of K ranging from 9 to 46 percent and losses of Mg ranging from 13 to 17 percent. Similarly, estimates of the loss of P through this pathway range from 10 percent in a mixed-eucalypt forest in Tasmania (Harwood and Jackson 1975) to 46 percent in the longleaf pine site described by Christensen (1977).

The degree of combustion of the fuels and the resultant nature of the particulate materials available for transport can influence the relative importance of volatilization and ash convection as pathways for nutrient export. Raison and others (1985) determined that low intensity fires in three subalpine *Eucalyptus* forests resulted a ratio

Table 1.—Estimates of temperatures at which various elements/compounds of ecological importance volatilize under laboratory conditions; elements/compounds are listed in order of increasing minimum volatilization temperature

Element	Form	Volatilization
		°C
N	organic forms	175-200[a]
N	fumaric acid dinitrile	189[b]
N	alanine	<200[b]
N	adenine	<200[b]
P	phosphoric acid	203[b]
P	phosphobenzoic acid	249[b]
N	aminobenzoic acid	249[b]
P	organic forms	340-360[a]
S	organic forms	375[a]
K	inorganic forms	550[a]
P	inorganic forms	770[a]
K	base metal	774[b]
Ca	$CaCO_3$	825-898[b]
Mg	$MgCO_3$	900[b]
Mg	base metal	1107[b]
P	potassium hexametaphosphate	1320[b]
Ca	base metal	1487[b]

[a]Agee (1993).
[b]Weast (1969).

of 57 percent particulate to 43 percent nonparticulate loss of P when combustion produced black ash, and a ratio of 73 percent particulate to 27 percent nonparticulate loss when combustion proceeded all the way to very fine grey ash.

Although small losses of cations via direct volatilization may occur even in low intensity fires (e.g., Raison and others 1985), losses from the ecosystem of P and cations of the magnitudes listed also must represent exports of ash and other particulate combustion products due to the action of either ambient wind or air drawn (entrained) into the combustion zone because of the buoyancy of hot flame gases. Fire intensity (the rate of heat release from flaming per unit length of fire line, kW/m) determines both flame length and the velocities of air entrainment. Thus, it would be expected that losses of nutrients by ash convection would increase with fire intensity, a pattern suggested by Raison and others (1985). In addition, fire intensity tends to increase when fires burn in wind, so wind and fire intensity often may act together to increase nutrient losses in ash. Given the strong spatial heterogeneity in fuel consumption and intensity typical of most oak ecosystem fires, there likely are large spatial differences in nutrient losses either from volatilization or ash convection. Thus, one can generalize that losses of nutrients to the atmosphere may result in large losses of nutrients from the ecosystem, what proportion of the ecosystem nutrient capital is lost and via which atmospheric pathway depends on the element of interest, its biochemical context, the local weather conditions, and the behavior of the fire.

ASH DEPOSITION AND SOIL NUTRIENT AVAILABILITY

The material that remains after fire has consumed part or all of the fuel it can access is a combination of patches of uncombusted material, partially combusted woody material, and ash, the end product of complete combustion of the carbon (C) skeleton of the fuel materials. The inorganic materials that comprise the ash/partially combusted materials and whatever soluble organic compounds are present in the latter are all easily dissolved by rainfall. Whether these materials remain in the mineral soil or are lost from the system through leaching or overland runoff depends on several factors, most notably weather patterns, topography, and the properties of the mineral soil at that site.

During the first month after an early spring fire in a South Carolina pine plantation complex, approximately 80 mm of rainfall resulted in ~70 percent of the Ca and K in the ash being dissolved and transported downward into the soil (Lewis 1974). Similarly, in a study of post-fire nutrient dynamics in a mixed conifer forest in Washington, Grier (1975) observed that 90 percent or more of the ~150 kg Ca/ha, 50 kg K/ha, and 85 kg Mg/ha present in ash were dissolved and transported during the first precipitation event after the fire: 67 cm of snowmelt. In both of these cases, there was little actual loss from the ecosystem as the vast majority of the nutrients dissolved from the ash remained in the mineral soil.

The impact of ash deposition and dissolution on soil nutrient status is most obvious when fire has been both frequent and concentrated in space. For example, in their study of an oak forest site in Pennsylvania that had been used as a charcoal hearth from 1771 to 1884, Mikan and Abrams (1995) reported that Ca and Mg availability in the soils under the former hearth were 12.1 and 4.8-fold greater, respectively, than in the surrounding forest. Thus, repeated deposition of concentrated ash caused changes in nutrient status that were still apparent more than a century after the last hearth fire.

A similar example is from the study of the effects of burning concentrated slash piles (i.e., jackpot fires) after logging in an English oak forest (Jalaluddin 1969). In this study, the slash was piled in areas of < 2.0 m in diameter and burned for nearly 3 hours. Samples taken several days after the fires indicated that dissolution of the resulting ash had increased soil pH from 6.0 to 9.0. However, within 6 months, continued leaching of the soil resulted in the pH returning to ~ 6.0. Thus, the effects of fire on nutrient status are sensitive to the amount of ash deposited in a fire, the number of fires on that site, and the intensity/duration of the leaching of the site by precipitation following ash deposition.

After a single fire in a pine-oak ecosystem in Kentucky, Blankenship and Arthur (1999) reported that soil pH had increased by 0.2 to 0.3 unit, demonstrating the effect of a single fire and subsequent ash dissolution on soil base status. After two or four fires in a mixed-oak ecosystems in Ohio, Boerner and others (2004) reported fire-induced, significant and persistent increases in soil pH of 0.2 to 0.6 unit. This study also reported increases in extractable Ca^{2+}, reductions in extractable aluminum (Al^{3+}), and in increased molar Ca:Al ratios, though these differences were statistically significant only in the more nutrient-poor sites (Boerner and others 2004). In sites with significant limestone influence in the soils, changes in base cation status were transitory. Similarly, Knoepp and others (2004) observed increased soil pH, exchangeable Ca, and exchangeable Mg after site preparation (fell-and-burn) burning in western North Carolina.

Although most studies of the effects of fire on nutrient status focus on a single fire or several fires over a decade or less, two longer term experiments in oak forests may help shed light on chronic effects of burning on oak forest soils. Eivasi and Bayan (1996) assessed the nutrient status of soils in oak-hickory flatwoods in Missouri that had been subjected to annual or periodic (~ 4 year interval) burning for 40 years. They found no significant effects of fire at either frequency on soil pH, Ca, K, or Al. They did report that available P was reduced to 24 percent of controls by annual burning and to 35 percent of controls by periodic burning. Thor and Nichols (1973) reported a similar lack of change in pH from eight annual or two periodic burns in oak stands on the Highland Rim region of Tennessee. DeSelm and others (1991) resampled the Thor and Nichols (1973) sites after 27 years of annual or periodic burning, and again reported that burning had no significant effect on soil pH or the availability of K or P. Thus, although the effect of lengthy periods of burning had no effect on base-cation availability in either ecosystem, the long-term effect of low-intensity fire on P availability remains unclear.

No aspect of the belowground responses to fire has been more intensively studied than the availability of N in the soil. This is partially the result of the generally accepted view that most ecosystems are N-limited. Whether eastern oak ecosystems are still N-limited in 2005 given the chronic deposition of N from fossil fuel combustion and agricultural sources is an open question and beyond the scope of this review.

Wan and others (2001) performed a meta-analysis of the effects of fire on soil N availability and concluded that, over all ecosystem types, inorganic N availability is increased by fire. Ammonium (NH_4^+) availability peaks soon after fire but returns to pre-fire levels in less than a year. The pulse of NH_4^+ is due to a combination of liberation from organic matter degraded during the fire, activity of heterotrophic soil biota, and N-fixation by symbionts of newly colonizing plants. Nitrate (NO_3^-) availability peaks some months later (generally 6 to 12 months after fire), and is the result of enhanced NH_4^+ availability and increased activity of nitrifying bacteria.

However, when the results were stratified by fire type or intensity, Wan and others (2001) concluded that high intensity wildfires and slash fires resulted in increased NO_3^- and NH_4^+ whereas prescribed burning did not. As the majority of fires in eastern oak forests may be more like prescribed fires than wildfires or slash fires in grasslands, shrublands, or coniferous forests, one might speculate that the conclusions of Wan and others (2001) would have been different had their analysis been limited to ecosystems with a significant oak component.

Soil organic matter is a key part of a forested ecosystem. This material is an essential store of nutrients (especially N and C), is essential to exchange processes that regulate the availability of Ca, Mg, S, P, and K to plants and microbes, helps stabilize soil structure by cementing soil particles into stable aggregates, and insulates the soil against changes in microclimate. Enhancement of soil organic matter development in forests also is a pathway that is being examined as a possible mechanism for ameliorating the effects of CO_2 released by fossil fuel combustion.

Johnson and Curtis (2001) performed an extensive meta-analysis of the effects of disturbance on soil organic matter in forests. They concluded that ecosystems that had experienced fire approximately 10 years earlier had an average gain in soil organic C of approximately 8 percent. They attributed this change to a combination of factors, including infiltration into the mineral soil of organic matter from partially combusted fuels, conversion of labile (easily decomposed) organic matter into recalcitrant (stable) organic matter, and the effects of the colonization of burned areas by plants which harbor N-fixing symbionts.

Despite a rich literature from coniferous ecosystems that demonstrates significant loss of soil organic matter during and after fire, few studies have demonstrated major changes in soil organic-matter content following fire in eastern oak ecosystems. Knighton (1977) observed no significant change in soil organic C following one to three fires in Wisconsin oak forests, and Knoepp and others (2004) reported no significant change in soil C over 5 years following site preparation burning in oak-pine stands in North Carolina. We observed only slight changes in soil organic matter in Ohio oak-hickory sites subjected to one to four prescribed fires (Boerner and others 2000b; Boerner and Brinkman 2004). It is important to note that the lack of change in soil organic matter after fire may be specific to low-intensity, dormant-season fire regimes. For example, soil organic matter was reduced by as much as 50 percent following intense wildfire in an Israeli pine-oak (*Pinus halepensis* Mill-*Quercus calliprinos* Webb) ecosystem (Kutiel and Naveh 1987).

Results from single fires or from studies of several fires to the longer term are projected with considerable uncertainly. And, this is exacerbated by somewhat conflicting results from the two longer term studies of fire in oak forests that do exist. Eivasi and Bayan (1996) could not detect significant changes in soil organic-matter content in the soils of a Missouri oak flatwoods ecosystem even after more than 40 years of annual or periodic burning. By contrast, Philips and others (2000) reported significant reductions in A horizon organic-matter concentration and content after 35 years of annual fires (but not periodic fires) in a Tennessee oak forest.

Some of the material deposited on the ground surface after a fire is partially combusted, charred woody material. The blackened materials that comprise a considerable portion of this charred fuel have been termed “black carbon,” and have become a focus of research on organic matter changes after fire (Certini 2005). Black carbon can comprise as much as 40 percent of the soil organic matter in ecosystems exposed to frequent fire (Ponomarenko and Anderson 2001), and this material may have sorptive properties that are important in regulating soil solution chemistry and biochemistry for some time after fire (Wardle and others 1998). However, no estimates of the abundance or importance of black carbon in oak forests have been published.

EXPOSURE, EROSION, AND HYDROLOGY

Changes in the soil water and hydrological regimes of an ecosystem typically are proportional to the proportion of the litter, humus, coarse woody debris, and woody stems that are consumed during the fire. Fires that consume most or all of the forest floor and root systems may produce major changes in watershed hydrology. For example, Folliot and Neary (2003) reported an increase of 90 percent in watershed output after a high-intensity fire in a ponderosa pine (*Pinus ponderosa* Laws.) ecosystem in Arizona. By contrast, low-intensity fires that do not consume the entire forest floor and/or fires that affect only part of a watershed may not produce hydrological changes that can be resolved by typical monitoring methods (Bethlalmy 1974).

The volume of water leaving a forested watershed in streamflow and ground water recharge is a function of a range of factors, including interception of precipitation by the canopy and subsequent evaporation, transpiration by plants, evaporation from the ground surface, overland runoff, water storage in the soil, and transport of water through the soil profile to groundwater or surface waters. As low-intensity, dormant-season fires in eastern oak forests typically do not have a major impact on canopy structure and understory vegetation grows/regrows rapidly after fire, neither canopy interception/evaporation nor transpiration likely will be affected strongly by fire. Evaporation from the blackened soil surface may be enhanced after fire but such an effect likely will be limited to the period before the canopy leafout and understory development in the spring (Iverson and Hutchinson 2002).

Fires that consume the entire forest floor also may affect the water-holding capacity of the top several centimeters of the soil through changes in soil texture and structure (Austin and Baisinger 1955). Severe heating may cause clay-size particles to fuse into larger particles, thereby affecting soil texture and porosity; however, the intensity necessary to produce such effects is not typical of oak ecosystem fires. Changes in surface soil permeability or porosity also may occur as the result of the volatilization of organic compounds that had served to cement soil aggregates (Dyrness and Youngberg 1957). This can result in reduced porosity as impacted aggregates crumble and plug large soil pores (Moehring and others 1966). Severe and/or repeated burning also has the potential to increase soil bulk density, though such effects typically are less evident after low intensity, periodic fires (Moehring and others 1966; Agee 1973).

Whether water impacting on the forest floor/soil surface percolates into the soil or runs off over the surface depends not just on the permeability of the soil but also on whether water is free to move downward through the soil. In ecosystems where coarse-textured soils are overlain by resinous forest floor materials, combustion of the forest floor can cause volatilization of hydrophobic organic compounds. If there is significant downward transport of those volatilized materials, they may condense as hydrophobic coatings on soil aggregates, resulting in the development of what has been termed a hydrophobic or water-repellant layer (Agee 1973). Significant hydrophobicity has been observed after fire in coniferous forests and chaparral (e.g., DeBano and others 1970), and once developed can result in significantly increased erosion, as water that would normally percolate vertically through the profile is instead routed downslope through the surface soils along the water-repellant layer (Agee 1973). Although examples of fire producing nonwettable soil layers are common in coniferous ecosystems to the west and north of the oak forest region, I have found no reports of hydrophobicity developing after fires in oak forests.

As regulation of water flow and water quality are among the most important ecosystem services we expect forested watersheds to supply, the issue of the effect of fire on water quality merits consideration. Again, in ecosystems subject to large, intense fires, water quality may be affected greatly by surface runoff and particulate erosion. However, in ecosystems such as the eastern oak forests, fires do not typically produce effects on the watershed that are likely to produce such large or chronic changes in erosion and runoff. Instead, leaching of soluble materials from the ash and partially combusted fuels through the soil profile and into ground water or water bodies is of greater concern.

Knighton (1977) analyzed the effects of zero to three annual burns on the quality of water leaving an oak-hickory forest in central Wisconsin. He reported solution concentrations of Ca^{2+} and Mg^{2+} to be significantly greater in leachate below the burned areas than in leachate from unburned controls. The solution concentrations of NO_3^- and PO_4^{3-} also were greater in leachate from burned sites, and the degree of enrichment was proportional to the number of fires a site had experienced. However, Knighton (1977) pointed out that the increases in N and P concentration with fire frequency were not statistically significant due to high-among sample variability. Similarly, Lewis (1974) noted increases of 50 to 125 percent in concentrations of Ca^{2+} and K^{+} (but little change in NO_3^- and PO_4^{3-}) in soil leachate after fires in South Carolina pine plantations, but great variation among samples resulted in the differences not being statistically significant. Whether water quality is affected by low-intensity, dormant-season fires is likely to be determined by the relationship among ash deposition, post-fire precipitation, and soil nutrient storage capacity. Thus, predictions of fire effects on water quality will be site and situation specific.

SOIL BIOTA AND ECOSYSTEM PROCESSES

The assemblage of animals in the forest floor and soil of a forested ecosystem is both highly diverse and critically important to ecosystem processes. These animals form a complex food web that is responsible for much of the conversion of coarse detritus produced both below- and aboveground into fine organic materials that are suitable for subsequent microbial decay. Without these micro- and mesofauna present, leaf litter decomposition essentially ceases (Edwards and Heath 1963).

Of the many groups of arthropods present in and on the forest floor and soil of forested ecosystems, the *Collembola* (springtails) and *Acari* (mites) are the most important to the decay process (see also Rieske-Kinney, this volume). In southern Ohio mixed-oak forests, three annual burns reduced the numbers of springtails and mites by more than 50 percent, while a single fire resulted in no significant change in abundance in either group (Dress and Boerner 2004). There are even suggestions in that study that conditions after a single fire may have supported greater densities of microarthropods than did those of the unburned controls. Metz and Farrier (1971) reported that mites and springtails in the forest floor decreased in abundance in direct relation to fire frequency in a South Carolina pine plantation. When fires were applied every 4 years, there was sufficient time for recolonization and population growth to return numbers to pre-fire levels, whereas when fire was applied annually, abundances remained depressed over time. By contrast, Metz and Farrier (1971) reported no significant effect of annual or periodic fire on mites and springtails in the mineral soil.

As sparse as the database for microarthropods might be, that for other faunal groups is even more depauperate. Matlack (2001) reported no longterm effect of fire on nematode numbers or diversity in southern pine plantations. Bhadauria and others (2000) observed that annelid earthworm abundance was depressed by fire in a mixed oak-pine forest in northern India. However, they also reported that earthworm abundance returned to pre-fire levels over several months after fire as the result of migration from neighboring, unburned areas. Bhadauria and others (2000) suggested that post-fire conditions in the burned areas might be better suited for annelid growth than conditions present in unburned areas.

The effects of fire on the soil microbial community can be evaluated by examining individual species, species groups, or microbially mediated processes such as the conversion of organic, N-containing compounds to inorganic N as NH_4^+ (N mineralization). In a South Carolina pine plantation, Jorgensen and Hodges (1971) found that both fungal and bacterial abundance in the forest floor were reduced by annual fire, but abundances of both groups in the forest floor of plots burned on a 4-year cycle were greater than those of unburned controls (Fig. 2). The same pattern was apparent for bacteria in the upper 5 cm of the mineral soil, whereas

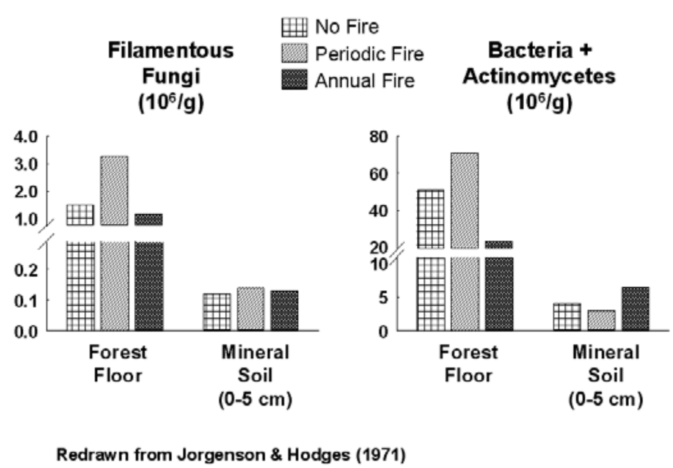

Figure 2.—Effects of fire on soil and forest floor microorganisms, redrawn from Jergensen and Hodges (1971).

fungal abundance in the mineral soil was not affected by fire treatment (Jorgensen and Hodges 1971). The notion expressed in this study that periodic fire produces conditions for soil microbes superior to those present in unburned sites parallels those from the studies of soil invertebrates described earlier (Bhadauria and others 2000; Dress and Boerner 2004.

Blankenship and Arthur (1999) demonstrated a significant, positive effect of a single winter fire in an eastern Kentucky oak-pine ecosystem on bacterial biomass, and this was accompanied by an increase in the bacteria:fungi biomass ratio in this ecosystem. A similar shift in the bacteria:fungi ratio after a low-intensity slash fire was reported by Perry and others (1984) in a northwestern coniferous forest. As the fungi in those coniferous systems are responsible for producing the compounds that make Fe available for plant uptake (hydroxymate siderophores), Perry and others (1984) believed that there was a potential for iron deficiency of tree seedlings after fire, especially where pH has been increased. Whether such short-term effects on the microbial community occur broadly in oak ecosystems and for how long such effects might persist is unknown. Fonturbel and others (1995) found no chronic or lasting effect of prescribed fire on heterotrophic bacteria or filamentous fungi in a pine ecosystem in Spain, but the applicability of this finding to eastern oaks is uncertain. Eivasi and Bayan (1996) found that 40 years of annual or periodic burning reduced microbial biomass in a Missouri oak forest by 32 and 21 percent, respectively.

Mycorrhizal fungi are key symbionts for virtually all forest plants. There are two major groups of mycorrhizal fungi found in the eastern oak regions (as well as several more limited groups of mycorrhizae *sensu latu* that are not considered here). Ectomycorrhizal fungi (ECM) are a group of higher fungi (basiodiomycetes and ascomycetes) that form symbioses with many conifers (e.g. *Pinus, Picea, Abies*), oaks (*Quercus* spp.), hickories (*Carya* spp.), and beech (*Fagus grandifolia* Ehrh.). Arbuscular mycorrhizal fungi (AMF), also referred to as endomycorrhizae, vesicular-arbuscular mycorrhizae and (VAM), are a much smaller and less diverse group of lower fungi that form symbioses with herbaceous plants and with woody plants that do not form ECM symbioses. Both AMF and ECM are critical to plants for the acquisition of P, as P is relatively immobile in most forest soils. In addition, ECM can be important in N uptake and pathogen resistance.

The colonization of roots by arbuscular mycorrhizal fungi is sensitive to soil heating, at least under laboratory conditions. Colonization of plant roots by AMF was reduced by ~50 percent when soil temperatures exceeded 50 to 60°C, and was reduced by 95 percent when soil temperature reached 90°C (Klopatek and others 1988). Whether this is important under field conditions is uncertain, as Knorr and others (2003) reported no significant effect of one to four fires on the abundance of the AMF-specific glycoprotein glomalin in oak forest soils in Ohio.

Severe fires can alter ECM community structure in bishop pine (*Pinus muricata* D.Don.) stands on the Pacific coast (Baar and others 1999). The decrease in ECM abundance and diversity that takes place after fire in western conifer forests may occur because the bulk of the fungal mycelium in those ecosystems is located in the humus layer of the forest floor, and that layer often is completely lost in severe wildfires (Harvey and others 1986). However, the applicability of this work to eastern oak ecosystems is uncertain. First, not all eastern oak forests exhibit a strongly developed humus horizon in the forest floor. Second, two studies in oak ecosystems suggest that fungi present in the mineral soil can rapidly recolonize the forest floor in the months after fire.

Tresner and others (1950) found that the species composition of fungi in a Wisconsin oak forest did not change with depth from the forest floor down through the upper mineral soil. They concluded that colonization of disturbed surface layers from deeper soil is possible without changing fungal community structure. Jalaluddin (1969) examined fungal dynamics in soils of small plots on which concentrated piles of slash had been burned. Fungal abundance in the center of those intensely burned plots was only 3, 6, and 11 percent of control levels at 1 week, 3 months, and 6 months after fire. However, at the edges of the burned plots, where recolonization from neighboring, unburned soils could occur, abundances were 17, 31, and 43 percent of control levels at those same points in time. Thus, even when intense fire effectively sterilizes the mineral soil, thus precluding recolonization from below (*sensu* Tresner and others 1950), mycelial ingrowth from surrounding areas can facilitate reestablishment of the fungal assemblage relatively quickly (Jalaluddin 1969).

Studies of changes in microbial function due to fire are more common than direct studies of community structure or abundance. In western conifer systems, a significant proportion of the N lost during fire may be offset by N_2 fixation by leguminous plants and their *Rhizobium* symbionts (Agee 1996). Among N-fixing woody species, black locust (*Robinia pseudoacacia* L.) has the most potential to contribute to the post-fire N economy in this way. Although black locust and its bacterial symbionts fix considerable amounts of N_2 and may have been common in burned areas historically, this species is now most common in postagricultural old fields in the southern portion of the oak forest region (Boring and Swank 1984); thus, there are few data with which to assess its impact after fire.

In Georgia loblolly pine forests, considerable amounts of N may be fixed by herbaceous perennials (e.g., *Desmodium* spp. and *Lespedeza* spp.) using the same symbiosis (Hendricks and Boring 1999). As both of these genera are widespread in the Eastern United States, the potential exists for this pathway to be important in oak ecosystems. Determining whether this potential is achieved awaits specific studies in oak forests.

In contrast to the sparse database on N_2 fixation in oak forests, many studies have demonstrated increases in N mineralization after single fires (reviews by Raison 1979; Boerner 1982; Wan and others 2001). These increases often are attributed to the alteration of organic matter by fire in such a manner as to render it more susceptible to microbial attack, to increases in microbial activity, and to changes in microclimate. Boerner and others (2000b, 2004) quantified soil organic matter, N mineralization rate and nitrification rate in four southern Ohio mixed-oak forests subjected to annual or periodic burning. At the full watershed scale, there was no consistent or persistent change in any index of microbially mediated N mineralization. In that study, pre-fire conditions and landscape characteristics were more important than fire behavior in explaining changes in C and N dynamics (Boerner and others 2000b). In a subsequent study of thinning+burning in a neighboring site, Boerner and Brinkman (2004) found no significant effect of a single fire on N mineralization in Ohio mixed oak-forests; a similar lack of fire effect on N mineralization has been reported for oak-pine forests in Georgia and Tennessee (Hubbard and others 2003) and oak-pine stands in North Carolina (Knoepp and others 2004).

It is important to note that the results of single fires or relatively short-term studies may not scale across longer time periods accurately. Vance and Henderson (1984) measured rates of N mineralization in the same long term Missouri stand used by Eivasi and Bayan (1996) a decade later. They found that N mineralization was reduced by longterm burning (30 years) and concluded that this change was a consequence of a change in organic matter quality rather than quantity. This is consistent with what one would expect if long-term burning results in an accumulation of black carbon over time (Ponomarenko and Anderson 2001).

Yet another approach to examining the impact of fire on the soil microbial assemblage is measuring the activity of the enzymes that microbes secrete into the soil to perform digestion processes. The activity of acid phosphatase is commonly used as a measure of overall microbial activity and as a surrogate measure for the rate at which N and P are mineralized from organic matter by microbes. Single fires in Ohio mixed-oak forests (Boerner and others 2000a) and pine forests in Spain (Saa and others 1993) and two to four fires over 5 years in Ohio mixed-oak forests (Boerner and Brinkman 2003) demonstrated reductions in acid phosphatase after fire, indicating an overall reduction in microbial metabolism. However, in the Ohio studies there were significant increases in phenol oxidase activity (Boerner and others 2000a; Boerner and Brinkman 2003). Phenol oxidase is an enzyme produced by fungi for the breakdown of recalcitrant organic compounds such as those that are likely to be left behind after the more labile organics are volatilized during fire or consumed shortly after fire.

Again, these short-term effects may not reflect what might happen over a long period of fire. Eivasi and Bayan (1996) reported significantly lower microbial biomass, acid phosphatase activity, and β-glucosidase activity in soils of Missouri oak-hickory sites exposed to annual or periodic fire for four decades.

Our overall understanding of the impact of fire in oak ecosystems on the soil microbial assemblage is growing but is not yet to the stage where a synthesis is achievable. Studies of community composition suggest that effects of one or several fires are rapidly ameliorated. Most functional studies suggest that the effects of fire are too modest to resolve analytically or are ones related primarily to changes in the relative proportions of labile and recalcitrant organic matter. However, long-term studies suggest that microbial activity and abundance may be significantly reduced if fire occurs frequently over many decades.

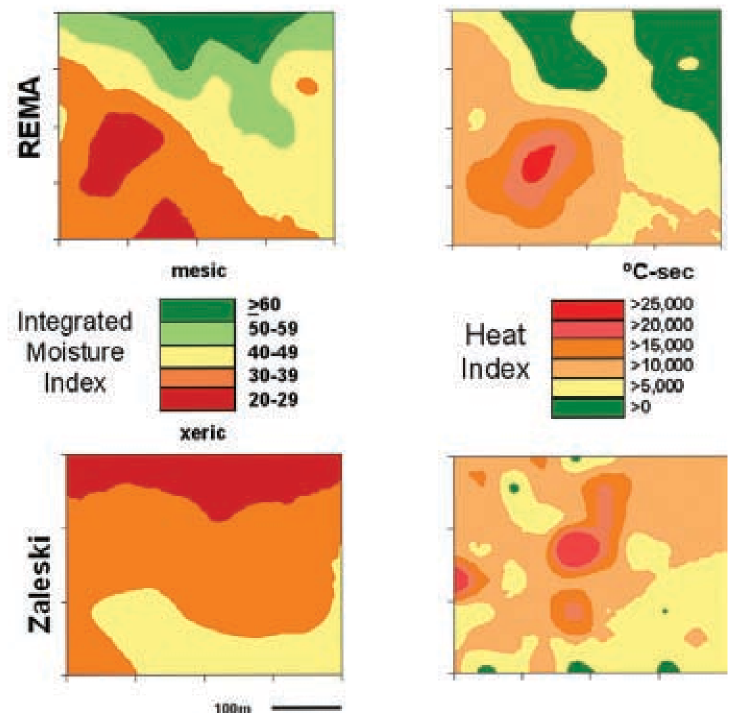

Figure 3.—Interpolated maps (semivariance analysis + kriging) of Integrated Moisture Index (IMI) and heat index in the forest floor based on 50-m grid points in two southern Ohio watersheds. The IMI data were supplied by L. Iverson and fire behavior data by M. Dickinson.

LANDSCAPE AND SCALE ISSUES

Forests with a significant oak component occur in a variety of landscape contexts in Eastern North America, from sandy coastal plains to Appalachian Piedmont and plateaus to the slopes of the Appalachian Mountains. Some of the oak region is recently glaciated, whereas other parts have not experienced glaciation for more than 10^6 years. Much of the eastern oak region consists of terrain that is dissected into ridges, hills, and valleys within which are strong gradients of microclimate (e.g., Wolfe and others 1949). In such complex landscapes, the behavior and effects of fire may vary so much among landscape positions within a watershed that considering only whole-watershed average effects may obscure important ecological patterns at the landscape scale.

To illustrate how fire behavior varies across the landscape, I present an example from the Ohio Hills Fire and Fire Surrogate (FFS) research sites in southern Ohio. The individual treatment units for the FFS project are areas of 20 to 30 ha in southern Ohio on which 50-m grids have been superimposed. Although the majority of studies that report fire behavior do so in terms of the maximum temperature experienced by thermocouple probes, this may not be the most useful metric for understanding fire effects. The potential impact of fire is better described by the integration of thermocouple probe temperature over the time a fire is actually present at a point. When expressed at a single point (above a single sensor) this metric is referred to as the heat index (Bova and Dickinson 2005) and is expressed in units such as °C-sec.

Figure 3 presents interpolated maps of two of the Ohio FFS treatment units, with a metric of landscape position and long-term soil moisture potential (Iverson and others 1997) on the left and a map of heat index on the right. The heat index map clearly portrays both the variations in fire behavior across this landscape as well as the strong patchiness or spatial heterogeneity typical of fire in these forests. Unfortunately, the time and expense involved in instrumenting a site for measurements of heat index and the unpredictability of wildfire occurrence combine to make measurements of actual fire intensity in oak forests uncommon.

The recent development of integrative, GIS derived metrics designed to represent within-watershed scale gradients of elevation, aspect, slope shape, soil type/depth, and other aspects of microclimate have made available tools to allow us to begin to understand the landscape ecology of fire in a quantitatively rigorous manner. For southern Ohio, Iverson and others (1997) developed an Integrated Moisture Index (IMI) that integrates a large number of microclimate and soil parameters into a single metric. By stratifying sampling designs based on this integrated measure, we have been able to resolve within-watershed variations in fire effects on the soil.

In Ohio mixed-oak forests where fire has been effectively suppressed, soil pH, inorganic N, extractable Ca^{2+}, Mg^{2+}, and Al^{3+}, and molar Ca:Al ratio commonly vary among landscape positions, with relatively xeric ridgetops and upper, south-to-west-facing slopes having lower pH, cation availability, and molar Ca:Al ratio than other landscape positions (Table 2). N mineralization rate, and nitrification, and inorganic N in the soil solution tend to increase with increasing long-term soil moisture potential, or from xeric through median to mesic IMI classes Table 2).

By contrast, glomalin content tends to be greatest in the most xeric landscape positions (Table 2). As the content of this AM fungal-specific compound is thought to be directly proportional to AM fungal activity and inversely proportional to P availability, this suggests that the latter varies across the landscape in a manner parallel to that of N, This has been supported by direct analyses of P availability among landscape positions (Boerner and others 2003).

In some studies, relatively xeric landscape positions tend to have greater soil organic C content than more mesic positions, though others demonstrate no significant within-watershed scale variations in organic matter (Table 2). Soil enzyme activities, as indices of microbial activity, parallel total soil organic C closely, as do the abundances of various soil arthropod groups (Table 2). Thus, variations in soil biological and chemical properties among landscape positions within the small watersheds of the Appalachian plateaus are so diverse and significant as to cast doubt on the utility of watershed-scale averages, and point out the need for sampling designed to detect treatment effects to be stratified by landscape position.

This is illustrated by comparing responses to fire at the whole watershed-scale treatment unit level to those in the various landscape positions within that treatment unit (Table 3). In some cases, watershed-wide responses parallel (at least in direction) those in the individual landscape positions, e.g., phenol oxidase activity at Arch Rock (Boerner and others 2000a, Table 3) and both soil organic matter and N availability at Arch Rock (Boerner and others 2000b, Table 3). Far more common are situations in which there are significant effects of fire in one or more landscape positions that are not reflected in the overall, watershed-scale, treatment unit effect (Table 3), and even some where the effects at different landscape positions are opposite in direction, e.g., acid phosphatase and chitinase at Young's Branch (Boerner and others 2000a, Table 3).

On level terrain with relatively small variations in topography and soils over fairly large areas (e.g., till plains and some coastal plains), extrapolating manipulative experimental studies on plots in the m^2 size range to the km^2 scale at which management decisions are made and applied is a straightforward process of multiplication. By contrast, on the complex and dynamic landscapes of the Appalachian plateaus, such scaling up is more complex. GIS-based, integrative tools such as the IMI make the conversion from small plot-based studies to broad-scale prediction possible in a way that is only now being realized by ecologists and forest managers.

Table 2.—Variations in soil properties among landscape positions within Ohio mixed-oak forests in the absence of fire (except Boerner and others (2000b) in which fire had no significant effect on soil properties). The Integrated Moisture Index (IMI) classes of Iverson and others (1997) are used to denote landscape positions. Upper case letters reflect relative ranks, with A=highest and C=lowest; landscape positions with different upper case letters were significantly different, based on the anova or ancova used in the indicated source.

Sources/site(s)	Parameter	Integrated Moisture Index Class		
		Xeric	Median	Mesic
Boerner *and others* (2003)	Soil pH	B	A	A
Four southern Ohio	Inorganic N	C	B	A
sites pooled	PO_4^{3-}	B	A	A
	extractable Ca^{2+}	B	A	A
	extractable Mg^{2+}	B	A	A
	extractable Al^{3+}	A	B	C
	Ca:Al molar ratio	B	A	A
Morris and Boerner (1998)	Soil pH	B	A	A
Watch Rock	Organic C	A	B	B
	N mineralization	B	B	A
	Nitrification	B	B	A
Decker *and others* (1999)	Acid phosphatase	A	B	C
Arch Rock and Young's	Chitinase	A	B	B
Branch pooled	Phenol oxidase	A	A	A
	β-glucosidase	A	A	A
	Organic C	A	A	A
Dress and Boerner (2001)	Live root mass	A	A	A
Arch Rock	Dead root mass	A	A	A
	Root production	A	B	B
Boerner *and others* (2000b)	Organic C	A	C	B
Arch Rock	N mineralization	C	B	A
	Nitrification	B	B	A
Boerner *and others* (2000b)	Organic C	A	A	A
Young's Branch	N mineralization	C	B	A
	Nitrification	B	A	A
Knorr *and others* (2003)	Glomalin content	C	B	A
Arch Rock				
Dress and Boerner (2004)	Oribatid Mites	A	B	B
Arch Rock	Collembola	A	A	A

Table 3.—Fire-induced variations in soil properties at the watershed scale and among landscape positions within Ohio mixed-oak forests. The Integrated Moisture Index (IMI) classes of Iverson and others (1997) are used to denote landscape positions, and the column labeled "Pooled" represents the full watershed. Effects of fire are indicated as 0=no significant effect, ↓=significant decrease, and ↑=significant increase, based on the anova or ancova used in the indicated source.

Sources/site(s)	Parameter	Integrated Moisture Index Class			
		Pooled	Xeric	Median	Mesic
Boerner *and others* (2000b)	Organic C	0	0	0	0
Arch Rock	Total Inorganic N	0	0	0	0
Boerner *and others* (2000b)	Organic C	0	↑	↑	0
Young's Branch	Total Inorganic N	0	0	↑	0
Boerner *and others* (2000a)	Acid phosphatase	↓	0	↓	0
Arch Rock	Chitinase	0	0	0	0
	Phenol oxidase	↑	↑	↑	↑
Boerner *and others* (2000a)	Acid phosphatase	↓	↓	↓	↑
Young's Branch	Chitinase	↓	↓	0	↑
	Phenol oxidase	↑	0	0	↑
Dress and Boerner (2001)	April Root Mass	↓	0	0	↓
Arch Rock	May Root Mass	0	0	↑	0
	June Root Mass	↓	0	0	↓

CONCLUSIONS

The last century has seen literally thousands of individual studies of the effects of fire on the forest floor and soils of American forests, shrublands, and grasslands. Despite this rich literature, the available database for the eastern oak forest ecosystems is surprisingly sparse, particularly in studies that have continued past the first post-fire year.

Most fires in oak ecosystems for which there are data generally have intensities that fall in the lower end of the range for North American ecosystem fires, though occasional higher intensity, stand-replacing fires have been reported. Most of the unconsolidated litter and a portion of the humus layer are combusted during typical oak forest fires, yet significant heating of the mineral soil is limited to areas where accumulations of fuel smolder for extended periods. As a result, direct mortality of organisms in the soil is low, and mortality in the forest floor is proportional to the extent of consumption.

A considerable portion of the N in detritus and vegetation may be lost to a combination of direct volatilization and ash convection. Ash convection also can result in significant losses of P and base cations. Fire behavior, intensity, and weather are important regulators of these processes. The nutrients contained in ash typically are dissolved by the first several precipitation events following fire, and the proportion of those solubilized nutrients that remain in the soil for subsequent plant use depends on the exchange capacity of the soil, landscape geomorphology, and weather. In shrublands and coniferous forests, development of hydrophobicity in soil during and just after fire is often an important factor in determining how much of the post-fire precipitation and solubilized nutrient load are lost to runoff. However, the literature does not suggest that this is an important process in eastern oak forests.

The availability of N, a potentially limiting nutrient for post-fire plant growth, and the total amount of organic matter in the soil are typically greater after fire, though this may be a more transient phenomenon in oak forests than in coniferous forests and shrublands where fire intensities are considerably greater. The possible importance of the charcoal (black carbon) remaining after partial combustion of woody material to post-fire biological activity is anl area of current fire research in coniferous forests and grasslands, though little information on black carbon in oak forests is available.

Microbial activity, biomass, and community structure in the forest floor often is affected in proportion to the consumption of this organic-rich layer. By contrast, short-term effects on microbes in the mineral soil seem to be primarily the result of progressive soil heating under accumulations of organic materials that smolder for longer periods. Recovery of the microbial assemblage from fire is relatively rapid and is the result both of propagules in situ and colonization from surrounding, unburned areas. Similar spatial and temporal patterns have been reported for microarthropods in the forest floor and soil, but the database on other soil invertebrates groups is so sparse as to warrant no generalization.

Much of the oak forest region of Eastern North America is complex in geomorphology, and this complexity is a challenge when experimental studies must be scaled up to land areas as large as those for which management decisions are made. The use of GIS-derived measures of landscape structure and heterogeneity to stratify sampling designs, inform statistical analyses, and integrate plot results to the landscape scale has emerged as a viable tool for generating management-scale information.

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RESPONSES OF OAK AND OTHER HARDWOOD REGENERATION TO PRESCRIBED FIRE: WHAT WE KNOW AS OF 2005

Patrick H. Brose[1†], Thomas M. Schuler[2], and Jeffrey S. Ward[3]

Abstract.—An obstacle to using prescribed fire to manage mixed oak forests is the varied results of previous fire studies. It has been reported that fires enhanced, hindered, or had no effect on the competitive position of oak in the regeneration pool. We review a portion of the published literature and identify key factors that led to the relative competitiveness of oak reproduction benefiting from or being harmed by prescribed fires. These key factors are synthesized into general guidelines to help practitioners understand how fire can be a positive force in the oak regeneration process. We also point out some situations where fire hinders the competitive position of the oak regeneration, and provide suggestions for researchers studying fire in mixed oak forests.

INTRODUCTION

There is developing consensus that fire has had a close association with and has shaped the species composition of mixed-oak forests in the eastern United States for the last several thousand years (Crow 1988; Abrams 1992; Brose and others 2001). Scars on stem cross sections are evidence that prior to the fire control era that began in the 1920s, fires in Ohio, Maryland, and West Virginia occurred at intervals of 5 to 15 years during the previous four centuries (Sutherland 1997; Shumway and others 2001; Schuler and McClain 2003). Other fire-scar studies have documented fire return intervals ranging from 2 to 24 years from New Jersey to the western portion of the mixed-oak forest in Missouri (Buell 1953; Dey and others 2004). Evidence of reoccurring fire also predates European contact in the eastern United States. In a charcoal and pollen accumulation study in the southern Appalachians, Delcourt and Delcourt (1997) found evidence of landscape-level fires in a forest dominated by oak, American chestnut, and pine throughout the past 3900 years.

The heretofore unrecognized close association of fire and oak forests in the eastern United States has led researchers to investigate the use of prescribed fire to sustain and regenerate oak forests. Eastern fire studies cover a wide range of stand characteristics, species mixes, fire intensities, associated silvicultural practices, and number of fires, and include an equally wide range of results. The competitive position of oak reproduction in the regeneration pool can be enhanced or hindered or not be affected. In this paper we review a portion of the fire/oak literature, grouping studies during the past four decades by stand structure (fully stocked mature and less than fully stocked mature), fire intensity (low and high), number of burns (single, twice, three to five, and six or more), and response of oak regeneration. We synthesize the common denominators for when the competitive position of oak reproduction is enhanced or harmed by prescribed fire and provide guidelines on using fire to help regenerate mixed oak forests.

Single Fires

Mature Stand, Low-Intensity Fire

This is by far the most common stand and fire type in the literature (Fig. 1). In this group, mature stands are in the understory reinitiation stage of development (Oliver and Larson 1996), usually are at least 75 years old and are near the end of a typical timber rotation for oak sawlogs. Low-intensity fires are defined as those with flames less than 2-feet long, which consume only the leaf litter and small woody debris, and cause little, if any, overstory mortality. The effect of these fires in this setting may be beneficial, detrimental, or neutral with respect to the relative competitiveness of oak regeneration (Fig. 1). American beech (*Fagus grandifolia*) remained the dominant species after a light surface fire in New York (Swan 1970). Nyland and others (1982) reported that the dense understory that develops after a single fire in New York could inhibit oak regeneration.

[1]USDA Forest Service, Northeastern Research Station, Irvine, PA 16329; [2]USDA Forest Service, Northeastern Research Station, Parsons, WV 26287; [3]Connecticut Agricultural Experiment Station, Department of Forestry and Horticulture, New Haven, CT 06504; [†]Corresponding author, 814-563-1040, email: pbrose@fs.fed.us.

In a study of the same burns, McGee and others (1995) found that the single fire increased the competitive status of American beech and red maple (*Acer rubrum*) while reducing the importance value of northern red oak (*Quercus rubra*). Prescribed fire reduced northern red oak survival and did not increase basal diameter growth in northern Georgia (Loftis 1990a). Low-intensity fires increased red maple and black birch (*Betula lenta*) density relative to oak in Connecticut (Ducey and others 1996). In this study there was no difference in height growth of oaks between the low-intensity and unburned sections (Moser and others 1996).

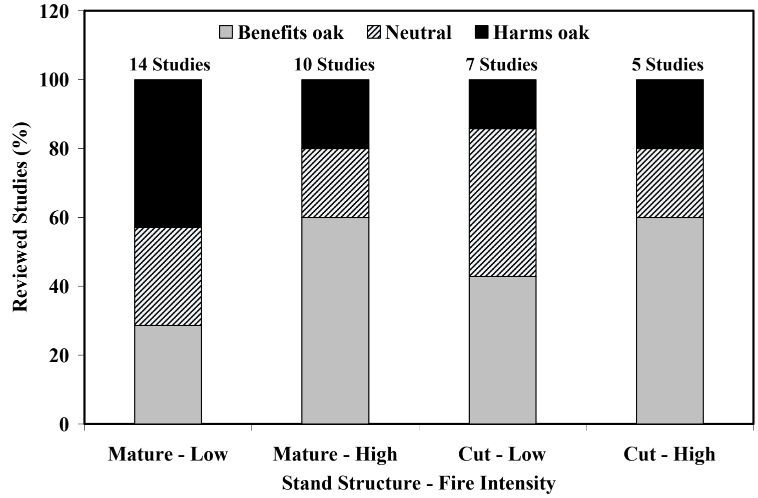

Figure 1.—Distribution of reviewed studies of the response of hardwood regeneration to single fires. The studies were divided into 12 groups based on stand structure, i.e., had the overstory been harvested to some degree, reported fire intensity (flame length or overstory mortality), and whether the fire benefited, harmed, or did not affect the competitive position of oak in the regeneration pool.

One of the more widely reported studies is the National Fire and Fire Surrogates project in southern Ohio. Low-intensity fires in unmanaged stands did not increase the number of oak seedlings after 1 year (Iverson and others 2004) or 2 years (Apsley and McCarthy 2004). In an associated study, McQuattie and others (2004) found that burning had no significant effect on seedling height of black oak (*Q. velutina*) and red maple. However, in this study, seedlings of both species were more massive on burned than unburned plots. A later report of the same study found that red maple seedlings, but not those of chestnut oak (*Q. prinus*), were taller in burned than unburned sections 2 years after the burn (Apsley and McCarthy 2004). In a related study also in southern Ohio, Hutchinson and Sutherland (2000) reported that red maple and white oak seedling frequencies were unchanged 3 years after a light surface fire. The effect of low-intensity fire on comparative height growth was equally ambiguous.

Several studies of low-intensity fire in mature stands have reported beneficial effects on oak regeneration. Barnes and Van Lear (1998) found that 2 years after a low-intensity spring fire in the southern Appalachians, regeneration (less than 2.0 inches in d.b.h.) densities of oak, hickory (*Carya* spp.), and sourwood (*Oxydendrum arboreum*) increased, while red maple density decreased. This change in species composition resulted in a slight increase in relative density of oak regeneration from 31 percent before the prescribed burn to 37 percent 2 years later. In Kentucky, Adams and Rieske (2001) found that the seedling height and diameter growth of white oak (*Q. alba*) was enhanced by a prescribed burn. A separate study on Kentucky ridgetops reported increases in density of both oak and red maple seedling densities after prescribed fire (Kuddes-Fischer and Arthur 2002). A low-intensity fire in south-central New Jersey resulted in low mortality of small oaks (Little 1946).

Mature Stand, High-Intensity Fire

In this category, high-intensity fires are defined as those with flames longer than 2 feet or killing significant numbers of overstory trees. Studies reporting on this type of fire in mature stands are less common than for the previous group (Fig. 1). Generally, these fires enhanced the competitive status of oak regeneration. Several of the researchers examined the aftereffect of wildfires by comparing burned areas and adjacent unburned stands. In New York, high-intensity fires did not change the average frequency of black oak but reduced the average frequency of red maple and white oak (Swan 1970). Swan also reported that the percentage of oaks top-killed by the fire was lower than that for red maple, sugar maple (*Acer saccharum*), and American beech.

Both absolute and relative densities of oak were increased 7 years after a high-intensity prescribed fire in North

Carolina (Maslen 1989). Although survival of oaks within a given size class was higher than for non-oak species following the fire, height growth was greater for non-oak than for oak species. Another North Carolina study reported that the density of scarlet (*Quercus coccinea*) and chestnut oak increased 1 year after a high-intensity fire while the density of red maple, northern red oak, and white oak decreased (Elliott and others 1999). In Connecticut, Ducey and others (1996) reported the density of northern red oak, eastern white pine (*Pinus strobus*), and black birch was higher 7 years after a high-intensity prescribed burn than on unburned plots while the density of red maple and white oak was decreased by the fire. As a follow-up to the Ducey paper, Moser and others (1996) reported that the height growth of oaks was greater following the high-intensity fire than that of oaks in the unburned controls.

Perhaps the primary reason for the increase in survival and growth of seedlings after high-intensity fires is that the amount of light reaching regeneration following fire-induced mortality of overstory trees was increased. Loftis (1990b) reported that the survival of northern red oak seedlings increased following a reduction in overstory density (similar to a high-intensity fire) as opposed to a reduction in understory density (similar to a low-intensity fire).

High-intensity fires are not a panacea for regenerating oaks. In addition to the resulting economic loss of timber (Hepting 1941; Little 1946; Paulsell 1957; Loomis 1973; Ward and Stephens 1989), high-intensity fires may inhibit the development of oak regeneration relative to other species. In West Virginia, the density of northern red and chestnut oak advanced regeneration decreased 5 years after a fire that reduced overstory basal area by 17 percent (Wendel and Smith 1986). In contrast, red maple density increased slightly over the same period.

Cut Stand, Low Intensity Fire

This stand type differs from the mature type in that cutting by group selection, shelterwood, clearcut, etc preceded the fire by one or more years and the regeneration layer was developing into saplings, i.e., the stand initiation stage (Oliver and Larson 1996). There were fewer studies of fire for this stand than for mature stands (Fig. 1).

In West Virginia, Collins and Carson (2003) reported that prescribed burning of recently created gaps reduced the abundance of northern red oak while the density of competitors was increased or not affected. Dolan and Parker (2004) found that a combination of burning and thinning did not result in the establishment of new oak seedlings on mesic or xeric oak sites in Indiana, but did result in greater consistency with respect to the number of shade-tolerant and intolerant seedlings in the understory. Jackson and Buckley (2004) noted an increase in small oak seedlings but attributed this to the sprouting of large oak seedlings that had been top-killed by fire a year earlier.

Fire studies in cut stands with a longer interval between the harvest and fire have shown substantial benefits to the oak regeneration. Brose and Van Lear (1998) and Brose and others (1999a) reported increases in the relative dominance of oak when burning was done 4 years after a heavy shelterwood cut. Kruger and Reich (1997) reported comparable results from fires conducted in 4-year-old oak shelterwoods in Wisconsin. In Connecticut, burning several years after a heavy shelterwood cut or a complete overstory removal also has increased the competitive position of oak in the regeneration pool (Ward and Gluck 1999; Ward and Brose 2004).

Cut Stand, High Intensity Fire

This grouping of papers was the smallest (Fig. 1) but included the oldest fire study in the eastern United States. A high-intensity fire in 1932 topkilled half of the stems less than 6 inches d.b.h. in a 30-year-old sapling stand in Connecticut. Fifty-five years after that fire, Ward and Stephens (1989) reported that oak densities were twice as high in burned than in unburned areas. The fire had no long-term effect on the density of other species groups. Comparable results were reported for Rhode Island (Brown 1960).

One of the first attempts to use fire along with a partial reduction in overstory stocking to increase the importance of oak in the understory was by Wendel and Smith (1986) in the Ridge and Valley region of eastern West Virginia. The effects of the fire were evaluated on the overstory and understory. Two years before the fire, the stand was thinned from below, e.g., the undesirable trees less than 6 inches d.b.h. plus cull trees were felled and left on the site. Strip head fires in the spring of 1980 consumed about 56 percent of the litter fuels and 18 percent of the woody fuels. Fire intensity varied with strip width, fuel loading, and wind patterns, though it likely was increased by the addition of fuels from the thinning 2 years earlier. Average fire scar height was about 5 feet, indicating a high-intensity fire for hardwood leaf litter. The fire reduced the number of stems more than 5 inches d.b.h. from 115 to 93 per acre and reduced basal area from 90 to 73 feet/acre. In the understory, red maple and black locust seedlings and sprouts increased in number while the number of northern red and chestnut oak decreased.

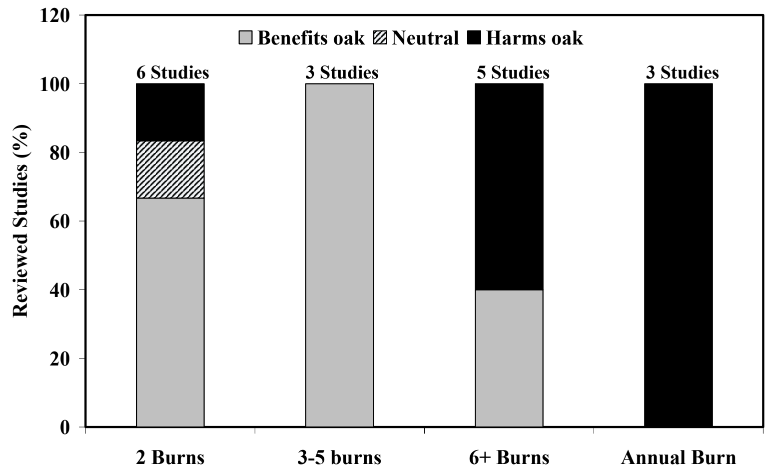

Figure 2.—Distribution of reviewed studies of the response of hardwood regeneration to multiple fires. The studies were divided into 12 groups based on the number of periodic burns and whether the fire benefited, harmed, or did not affect the competitive position of oak in the regeneration pool.

In a fire study conducted in the Virginia Piedmont, Brose and Van Lear (1998) and Brose and others (1999a) found that high-intensity fire was especially beneficial to the relative competitiveness of oak regeneration if the fire occurred during the spring versus summer or winter. High-fire intensity also was more beneficial to oak than low-intensity fire in studies by Ward and Gluck (1999) and Ward and Brose (2004).

Multiple Fires

Multiple prescribed burns at interval of 2 to 3 years were suggested by Sander (1988) to control unwanted woody competition. However, the impact of multiple burns may vary by the number of burns and their frequency (Fig. 2). Some researchers reported that periodic burns increase the competitive status of oak regeneration. A prescribed fire in Kentucky followed 2 years later by a wildfire increased the number of oaks in the shrub stratum (20 inches tall to 1 inch d.b.h.) more than six fold (Arthur and others 1998). This combination of fires also doubled the number of red maple in the shrub stratum versus the unburned plots. However, the plots that had been burned only once had higher red maple densities than the plots that had been burned twice. The second fire reduced the density of red maple sprouts and increased the number of chestnut oak sprouts.

In New York, both the relative and absolute densities of northern red oak reproduction more than 4.5 feet were greater on plots that had burned twice than on plots that had been burned once in New York (McGee and others 1995). This study also reported that northern red oak densities were higher following two high-intensity than two low-intensity fires and that the two hot burns increased the competitive status of American beech. Periodic burns at intervals of 5 years or less have nearly eliminated red maple from red pine (*Pinus resinosa*) stands in Michigan (Henning and Dickmann 1996).

Barnes and Van Lear (1998) reported that after three winter burns in South Carolina, there were fewer oaks than after a single spring fire. However, the winter-burned stands contained more than 5,000 stems/acre of oak regeneration and the height of oak sprouts following three burns was more than twice that of sprouts after a single spring burn. Following three periodic prescribed fires in Tennessee, oak regeneration was abundant, averaging more than 1,500 stems/acre (Clatterbuck 1998). Basal diameter and relative height growth of white oak seedlings were increased by two prescribed burns in Kentucky (Adams and Rieske 2001). However, relative height growth was greatest for seedlings in plots burned only once.

Other researchers maintain that periodic burning should be used with caution if the management objective is to regenerate oak. In a study in Tennessee, Thor and Nichols (1973) found that two periodic burns separated by 5 years increased the density of oak, sumac (*Rhus* spp.), and sassafras (*Sassafras albidum*) regeneration and eliminated red maple reproduction. However on the same plots, four additional periodic burns at intervals of 4 to 5 years the same plots increased the absolute and relative densities of red maple (DeSelm and others 1991). Concurrently, the density of post (*Q. stellata*), scarlet, and southern red oak (*Q. falcata*) decreased. In Missouri, periodic burning reduced the density of small oak saplings (Sasseen and Muzika 2004) while oak mortality increased with burning frequency (Dey and Hartmann 2004). For example, a single fire only resulted in 10-percent mortality among black oaks with a basal diameter of 1 inch, but three or more fires killed 25 percent of stems. In New Jersey, the mortality of oak saplings (one inch d.b.h) increased as the intervals between fires decreased. Mortality was 100 percent on annually burned plots, 62 percent on plots burned at 3-year intervals, and 11 percent mortality on plots burned at intervals of 10 years or longer (Somes and Moorhead 1950). In Kentucky, Green and others (2004) reported that red maple seedlings were tallest on plots burned three times than on those burned twice or not at all. They also noted that oak seedlings were tallest on unburned plots. Although red maple seedlings were taller than oak seedlings, mortality was higher in red maple.

In most studies researchers reported that annual burning reduces all regeneration density, including oak seedlings and saplings (Little and Moore 1949; Chaiken 1952; Paulsell 1957; Grano 1970). Although 6 years of annual burning in Tennessee increased total regeneration density and the density of oak regeneration (Thor and Nichols 1973), an additional 21 years of annual burns on these same plots eliminated all regeneration except for winged sumac (*R. copallina*) and oak (DeSelm and others 1991). Although some oak remain (primarily post oak) total density has declined by 97 percent or more. In the Missouri Ozarks, four annual burns greatly reduced or, in one case, eliminated small oak saplings less than 1.5 inches d.b.h. (Sasseen and Muzika 2004) and 14 years of annual burning eliminated large sapling stems in Minnesota (White 1983). The frequency of red maple seedling decreased after three annual burns in southern Ohio (Hutchinson and Sutherland 2000); surprisingly the frequency of white oak seedlings was not changed by the annual burns.

Synthesis

Every possible response of oak regeneration to fire is found in the literature, though commonalities among studies are apparent when the competitive position of the oak reproduction is enhanced or hindered. Results were most varied for single, low-intensity prescribed fires in mature stands. Where oak regeneration was harmed in this setting (Loftis 1990a; Nyland 1982; McGee and others 1995; Moser and others 1996), the studies were conducted on high-quality sites. The oak stems were small and they had been growing in dense shade. These stems probably had small root systems and minimal root carbohydrate reserves, and likely were of low vigor at the time of the fire. Several researchers have reported that such stems, regardless of species, usually do not survive fire of any intensity (Johnson 1974; Loftis 1990a; Brose and Van Lear 2004). Also, the fires did not significantly increase the amount of sunlight reaching the forest floor, so oak stems sprouting after the fires were forced to grow new tissue in sub optimal light conditions.

When single low-intensity surface fires benefited oak regeneration in mature stands (Little 1946; Barnes and Van Lear 1998; Adams and Rieske 2001; Kuddes-Fischer and Arthur 2002), the studies were conducted on dry, hot, sandy soils or xeric upper slopes. Oak regeneration tends to accumulate on such sites because they have sparser canopy cover and more sunlight reaches the forest floor (Johnson and others 2002). Thus, the regeneration has larger root systems and greater root carbohydrate reserves, and is more vigorous. Brose and Van Lear (2004) found that such oak regeneration generally sprouts after a low-intensity fire.

Single low-intensity fires in cut stands also produced varied results. The major differences between studies reporting a positive effect on oak regeneration and those reporting no effect or a negative impact were related to the intensity of cutting and the length of time between the cutting and the fire. The positive studies (Kruger and Reich 1997; Brose and Van Lear 1998; Ward and Gluck 1999; Ward and Brose 2004) were conducted in heavily cut shelterwoods or clearcuts and at least 4 years had passed between the cutting and the fires. In the neutral and negative studies (Collins and Carson 2003; Dolan and Parker 2004; Jackson and Buckley 2004), the degree of cutting was much lighter e.g., small gaps and low thinning, and only 1 or 2 years elapsed between the cutting and the fire. There is a clear relationship among the amount of light, number of growing seasons, and size/vigor of oak seedlings (Miller and others 2004). In the positive studies, the oak reproduction at the time of the cutting had sufficient light and time to build large root systems so it was more likely to sprout after the fires. In the neutral and negative studies, the regeneration reacted like small, low-vigor seedlings experiencing fire in a mature stand: it died or responded poorly.

The high-intensity fires in mature and cut stands generally benefited oak by being more detrimental to the survival and growth of non-oak species than to the oaks. This disparity is attributed to inherent differences in germination strategy between oaks and many of their competitors. Acorns have hypogeal germination, i.e., cotyledons remain in the shell and serve as a below-ground energy source for seedling development. Seeds of common competitors, e.g., red maple, have epigeal germination, i.e., cotyledons emerge and rise above the shell to form the first photosynthetic leaves. This difference in germination strategy places oak seedlings' root collar, and the accompanying dormant buds, lower than that of red maple. This difference in germination strategy is accentuated by wildlife. Acorns are routinely buried an inch or more into the forest floor by birds and small mammals while seeds from red maple and other competitors typically are not cached. Thus, an oak seedling generally has a deeper root collar than a red maple seedling due to seed burial and hypogeal germination. Consequently, hotter fires are more likely to kill the dormant buds of red maple than those of oak.

Another important silvical difference between oak regeneration and the reproduction of many competitors is the developmental rate of the root system. Upon germinating, oaks send a strong radicle deep into the soil to establish a taproot and emphasize root development over stem growth (Kelty 1988; Kolb and others 1990). Most competitors take the opposite approach; root development is sacrificed to promote rapid stem growth. Thus, oak regeneration usually has shorter stems than those of the competition but has larger root systems. Because of these silvical characteristics – hypogeal germination, emphasis on root development, and seed burial by wildlife – oak regeneration is less detrimentally affected by high intensity fires than its competitors.

The adverse effect of high-intensity fire on oak regeneration in previously cut stands (Wendel and Smith 1986) likely was caused by the short interval between cutting and the fire. Also, a high deer population may have preferentially browsed oak sprouts more than those of competitors during the 5 years between the fires and data collection.

Generally, two to five fires spread over a decade or more have benefited the competitive position of oak in the regeneration pool, likely due to selecting against species with inherently smaller root systems and creating/maintaining an understory light regime more favorable to the development of oak rootstocks. Multiple periodic burns may be a reasonable mimic of single, high-intensity fires in that they both create more open understories (but at different time scales).

However, multiple periodic fires do not always benefit oak. Merritt and Pope (1991) compared single burns conducted in the fall with a second burn conducted in the spring two growing seasons later. Consistent with other studies, both the first and second prescribed fires significantly reduced the number of competing shade-tolerant saplings 10 to 20 feet tall; however, neither the first or second prescribed burn aided in the establishment of new oak seedlings. Merritt and Pope concluded that it was not advisable to use fire alone, including multiple fires, to aid in the establishment of new oak seedlings in stands described as "recently mature."

Although the root mass/size of oak regeneration dwarfs that of most competitors of similar-size, it can be exhausted by multiple resprouting. This is evident in studies in which multiple periodic fires or annual burning were examined. There seems to be a point at which repeated fire does not continue to benefit the oak component of the regeneration pool. Where that point is and how rapidly it is reached likely is a function of several stand characteristics.

None of the reviewed fire/oak papers focused exclusively on oak and all contained information on other eastern species. From these papers, species can be grouped into three broad classes: susceptible, intermediate, and resistant to fire. Susceptible species, which are poor sprouters following fire, include black birch, eastern hemlock (*Tsuga canadensis*), eastern white pine (*P. strobus*), hophornbeam (*Ostrya virginiana*), sugar maple, and yellow-poplar (Marquis 1975; Kruger and Reich 1997; Brose and Van Lear 1998; Ward and Brose 2004). Fire-resistant species include the oaks, sympatric species such as the hickories (*Carya* spp), and those capable of consistent root sprouting e.g., aspen (*Populus* spp.), black locust *(Robinia pseudoacacia*), and sassafras, as long as there is sufficient light for sprouts to survive (Kruger and Reich 1997; Brose and Van Lear 1998; Waldrop and Brose 1999). Intermediate species differ from resistant species in how their sprouting ability responds to changes in fire intensity, fire seasonality, and seedling size. This is particularly true of red maple (Huddle and Pallardy 1996, 1999). Small red maple is readily killed by fire, even low intensity burns (Reich and others 1990) but the sprouting ability of this species develops as size increases (Brose and Van Lear 2004; Ward and Brose 2004). In most dormant-season burns and low-intensity fires, large advance red maple regeneration equals that of oaks in sprouting ability (Brose and Van Lear 1998). However, red maple is a markedly poor sprouter as fire intensity increases, especially during the growing season. Other intermediates that exhibit a wide range of root sprouting ability include American beech, blackgum (*Nyssa sylvatica*), and pin cherry (*Prunus pensylvanica*).

Guidelines for Foresters and Researchers

The following guidelines can aid foresters who plan to use fire to regenerate oak forests:

1. Prescribed fire is best used after significant reductions in overstory stocking to release existing oak seedlings, sometimes referred to as advanced regeneration, from competition. Prescribed fires should occur after released oak seedlings have had time to develop robust root systems (root collar diameter > 0.75 inch) that can sprout vigorously. The recommended time is 3 to 5 years after a shelterwood harvest that reduces stocking by about 50 percent in a fully stocked stand (Brose and others 1999a, b). Measures to protect valuable residual trees likely will be needed (Brose and Van Lear 1999).

2. Prescribed fires in late spring or summer are the most lethal to oak competitors because their root reserves are at their lowest levels. In some locations, the window of opportunity for such burns may be too short for planning purposes, so the focus should be on spring burns just before or during leaf expansion.

3. In the absence of adequate numbers of existing oak seedlings, the use of fire to establish new oak seedlings generally has not been successful. Some factors related to understories with inadequate densities of advanced oak regeneration are not related to the fire and oak relationship. Fire can create conditions suitable for oak establishment by reducing litter layers and reducing understory and midstory competition, but in the absence of large acorn crops, the newly available growing space will be used by other species in 1 to 2 years. Seedling establishment is largely dependent on large acorn crops that can overwhelm seed and seedling consumers, for example insects, rodents, and white-tailed deer. Avoid burning when an acorn crop has just occurred (Auchmoody and Smith 1993) or when oak seedlings are small, of low vigor, or recently established (Loftis 1990a; Brose and Van Lear 2003).

4. The use of fire following clearcuts has produced encouraging results in some locations. When oaks are present in the regeneration but other species are at much higher densities, a moderately intense prescribed fire can increase the relative density of oak by taking advantage

of this species' superior sprouting characteristics. Like shelterwood harvests, prescribed fires after clearcuts should be scheduled to allow oaks time to develop robust root systems that can produce vigorous sprouts. The time needed may be less than the 3 to 5 years recommended for the shelterwood regeneration process, but allow sufficient time for seeds of other species to germinate. Fires in clearcuts do not injure valuable residual trees unless some are retained for other management objectives.

5. Deer browsing is important in many areas and must be addressed should densities exceed a critical threshold before the regeneration process. High deer densities drive forest succession and discriminate against sprouting oaks after a fire (Collins and Carson 2003; Horsley and others 2003). This is an added obstacle on productive mesic sites where competition from non-oaks is greatest. In such cases, the use of fencing or other drastic means is necessary to mitigate the impact of deer.

6. A successful oak regeneration plan likely will ensure a diversity of woody species in the new stand. None of the silvicultural studies we reviewed reported the elimination of common oak competitors though many documented the virtual loss of oak when the regeneration plan favors other species intentionally or unintentionally.

The following are reminders concerning the essentials of high-quality research as well as points to consider in conducting fire research in oak forests:

1. Classify or measure fire intensity and/or severity at the same scale used to monitor regeneration response. This was by far the most common shortcoming in the studies we reviewed. The authors of several early fire/oak papers did not describe fire behavior while others included only general descriptions (cool, hot) at the stand level. Fortunately in more recent papers, fire behavior is described in more appropriate terms, for example, flame length, rate of spread, and char height, to help the reader gain a mental picture of the fire. The best papers report fire behavior and fuel consumption at the same scale that responses are measured.

2. Document preburn forest conditions. The response of a plant community to fire is influenced by several preexisting factors, especially the size/vigor of the regeneration and understory light levels. The more the preburn conditions are documented, the more useful the study. For oak reproduction this almost always requires measuring the diameter of the root collar

3. Use a control treatment. This should seem obvious, but, we reviewed studies that did not include unburned controls. Statistical inference, subsequent publication, and value to the scientific community of fire studies will be greatly enhanced with a control.

4. Do not be hasty in reporting results. Fire effects may take three years or more to be fully manifested. Early results need to be reported as preliminary rather than as definitive.

5. Publication of fire/oak studies need to be in high-quality journals so they reach the scientific community and technical outlets for the practicing forester.

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FIRE AND THE HERBACEOUS LAYER OF EASTERN OAK FORESTS

Todd Hutchinson[1]

Abstract.—Across oak forest landscapes, the herbaceous layer supports the great majority of plant diversity. As the use of prescribed fire increases, it is important to better understand its effects on biodiversity. This paper reviews the current "state of the knowledge" regarding fire effects on herbaceous layer vegetation. In typical dormant-season fires, direct heating effects are minimal on most herbaceous plants (forbs, grasses, sedges). Although woody plants are topkilled, nearly all resprout. Fire indirectly affects the herb layer by altering the forest floor and soil environments. The consumption of leaf litter during fire stimulates germination for a number of seedbanking species. Three case studies (oak forests in Missouri and Ohio, oak barrens in Illinois) of herb-layer response to fire are reviewed. These and other studies show that species richness and the cover of herbaceous plants usually increase after fire. Fire can have unique effects on herbaceous communities that are not realized with mechanical treatments (e.g., partial harvesting) alone. Although prescribed fire is commonly applied to maintain open-structured habitats that often contain rare plants, it also could be a useful management tool for sustaining and enhancing rare plant populations in upland oak forests. What is lacking most from our knowledge of how fire regimes affect the herbaceous layer of oak forests is: 1) the long-term effects of fire suppression, and 2) the long-term effects of periodic application of prescribed fire.

INTRODUCTION

The objective of this paper is to review what is known about the response of herbaceous layer vegetation to fire in eastern oak landscapes. While the focus is on fire effects in oak forests in the central hardwoods region, more open canopied communities within oak forest landscapes are included, as is research from other ecosystems, particularly when information is lacking from oak forests. In reviewing the literature, two major generalizations became evident: 1) although there have been a number of studies documenting herb-layer response to fire, nearly all report short-term effects (<10 years) of one to several fires, and 2) although the effects of fire vary among studies because of differences in vegetation and fire intensity, similar general responses were reported from many of the study sites.

THE HERBACEOUS LAYER

The herbaceous layer, also referred to as understory or groundlayer vegetation, generally is defined as all plants (woody and herbaceous) <1 m in height, though taller woody vegetation may be included (Gilliam and Roberts 2003). Across oak forest landscapes, the herbaceous layer harbors the great majority of plant diversity, including most rare plant species. The herb layer also provides habitat and food for numerous species ranging from arthropods to large mammals.

Broadly defined, herbaceous life forms are forbs (broad-leaved plants) and the graminoids (grasses and sedges); woody life forms are trees (seedlings and sprouts) and shrubs (including woody vines). Perennial forbs comprise the majority of plant diversity: in Ohio oak forests, perennial forbs made up 60 percent of species, followed by graminoids (14 percent), trees (11), shrubs and woody vines (9) and annual forbs (6) (Hutchinson et al. 2005a). However, woody plants, often have greater cover and biomass than herbaceous plants, particularly on dry sites (e.g., Hartman and Heuman 2003). Another characteristic of herb-layer vegetation is that most of the species occur infrequently across the landscape (Keddy 2005).

Many factors affect the composition and diversity of herb-layer communities in oak forest landscapes. At the broad scale, plant assemblages vary with climate and landforms while at the local scale (e.g., a watershed), species abundances are strongly associated with topographic gradients of soil moisture and fertility (Hutchinson et al. 1999). Also within oak forest landscapes, areas where tree growth is restricted (e.g.,

[1]USDA Forest Service, Northern Research Station, 359 Main Road, Delaware, OH 43015, 740-368-0090, email: thutchinson@fs.fed.us.

shallow soils) support woodlands as well as barrens communities usually containing unique species assemblages (Anderson et al. 1999). The intensity and frequency of past disturbances is a major factor related to current vegetation. For example, among secondary forests, those regenerated after agricultural conversion (old-field succession) have less diverse native herbaceous layers than continually forested stands (Flinn and Velland 2005). In many areas of the Eastern United States, deer browsing also has a major impact on herb-layer vegetation (Côté et al. 2004).

Evidence indicates that periodic fire has occurred in forests for millennia (Patterson, this volume; Guyette, this volume; Ruffner, this volume). It has become clear that fire control has played an important role in the changing species composition of trees in oak forests, facilitating the abundant establishment of fire-sensitive and/or shade-tolerant tree and shrub species. Although it seems likely that more than 75 years of fire control also has had a large effect on herbaceous communities, long-term data to evaluate this hypothesis are lacking. However, a better understanding of fire effects on herbaceous vegetation is available from recent studies in which prescribed fire has been reintroduced to long-unburned sites.

DIRECT FIRE EFFECTS ON PLANTS AND SEEDS

Most wildfires and prescribed fires in oak forests occur during the dormant seasons in late-winter/early spring and autumn (Haines and Johnson 1975) when most perennial herbs are dormant. Although the below-ground rhizomes of most herbs are shallow (<10 cm), little heat is transferred into the mineral soil during typical prescribed fires (Boerner, this volume). In Ohio oak forests, the peak soil temperature during fires, at 1-cm depth, averaged only 18°C (64°F) (Iverson and Hutchinson 2002). Flinn and Pringle (1983) found that rhizome death of several forest understory species occurred only as temperatures approached 60°C (140°F). The fact that few perennial herbs decrease in abundance after fire, reported from a number of studies, suggests that their rhizomes are largely unaffected. However, the direct effects of fire on the rhizomes of oak forest herbs have not been investigated.

If spring fires are conducted later in the season when plants are emerging, direct heat damage to foliage occurs. For example, Primack et al. (1994) found that a late-spring fire reduced leaf area in a population of *Cyprepedium acaule* (pink lady's slipper orchid) by 50 percent; however, for plants with 10- to 50-percent leaf-area damage, reproductive output returned to normal levels by the second year after fire.

In contrast to the perennial herbs, fire topkills the above-ground portions of nearly all small woody stems (shrubs, tree seedlings, vines). However, topkill induces sprouting (often vigorous) by hormonal signals, from dormant buds located below ground or on the base of stems. For example, Matlack et al. (1993) showed that fire increased stem density and shoot-growth of the shrub *Gaylussacia baccata* (black huckleberry).

Similar to fire effects on belowground rhizomes, fires likely cause little direct damage to seeds in the mineral soil. However, seeds within the litter and humus layers, which receive far more heat than mineral soil (Boerner, this volume) can be damaged by fire. Fire killed nearly 50 percent of red oak acorns located in the litter layer (Auchmoody and Smith 1993).

In the early 1990s, it was first reported that smoke could directly stimulate seed germination (Brown and van Staden 1997). Since then, it has been shown to be a widespread phenomena, at least in fire-prone habitats. Of 215 species tested in the Cape Region of South Africa, 101 have exhibited increased germination when treated with smoke (Brown and van Staden 1997). In western Australia, smoke increased the germination of 45 of 94 species (Dixon et al. 1995). A butenolide compound derived from cellulose combustion (common to all vegetation fires) was responsible for the stimulation of germination by smoke (Flematti et al. 2004). The effects of smoke on germination have not been tested on plants native to the Eastern United States.

INDIRECT FIRE EFFECTS ON PLANTS AND SEEDS

Herb-layer vegetation is affected indirectly when fire alters the forest floor and soil environments. When fires are of low to moderate intensity and conducted in the dormant season, these indirect effects arguably have a much greater effect on herb-layer vegetation relative to direct fire effects.

Leaf litter plays an important role in herb-layer plant communities (Facelli and Pickett 1991). In addition to its role in soil moisture and nutrient dynamics, leaf litter also serves as a mechanical barrier to seed germination and establishment. In the short term, the consumption of leaf litter and humus during fires probably causes the greatest changes in herb-layer vegetation by reducing the mechanical barrier and increasing light levels to seeds in the humus and mineral soil. Seeds of many herbaceous species in the eastern deciduous forest region have higher germination rates in light vs. dark conditions (Baskin and Baskin 1988). Throughout the growing season following spring fires, soil temperatures are elevated as the blackened litter absorbs more radiation (Iverson and Hutchinson 2002). Elevated soil temperatures likely affect the germination, establishment, and growth of herb-layer vegetation. The consumption of leaf litter in fires also releases nutrients that are then incorporated into the mineral soil, altering soil chemistry and likely affecting plant productivity, particularly on nutrient-poor sites (Gilliam and Christensen 1986; Boerner, this volume). If fire increases the amount of nitrate in soils, this could stimulate the germination of some seedbanking species (Auchmoody 1979).

As fire alters forest structure by topkill of saplings and some trees, light levels increase to the forest floor. However, in typical low-intensity fires, these changes are moderate and thus may have relatively little effect on plant performance. By contrast, high-intensity fires can open the canopy to a greater extent, increasing the competitive abilities of shade-intolerant herbs and shrubs. For example, in southern Ohio, all fires stimulate the germination of seedbanking, shade-intolerant species such as *Erechtites hieracifolia* (fireweed) and *Rubus* spp. (brambles) (Hutchinson et al. 2005a). After low-intensity fires when the canopy remains closed, these species rarely grow to more than several centimeters above the forest floor. However, when a portion of the canopy is removed by high-intensity fire or a combination of fire and harvesting, these species can rapidly grow to >1 m height in a single growing season (personal observation).

Fire has been shown to increase the reproduction of herbaceous species in several studies. In southeastern longleaf pine savannas, fire increased flowering (Platt et al. 1988) and clonal growth (Brewer and Platt 1994). For the rare plant *Liatris scariosa* var. *novae-angliae* (northern blazing star), which occurs in grasslands in New England, Vickery (2002) found that prescribed fire reduced seed predation rates, from 90 percent before fire to 16 percent after fire. Little is known about the effects of fire on the flowering and seed production of herbaceous species in oaks forests.

FIRE EFFECTS ON HERB-LAYER COMMUNITIES: THREE CASE STUDIES

It is difficult to determine the effects of fire on herb layer communities because of differences in vegetation and fires among studies. Therefore, to illustrate specific fire effects, I will first summarize the results of three studies. Each study: 1) had a duration of at least 5 years, 2) involved repeated dormant-season fires that reduced understory and midstory tree density but had little affect on overstory trees, and 3) included both pretreatment and multiple years of postreatment plant community data.

Missouri Ozarks Dry Oak Forest (Hartmann and Heumann 2003)

The study was conducted in the dissected 1,012-ha Chilton Creek basin, owned and managed The Nature Conservancy. The presettlement landscape consisted of open woodlands dominated by shortleaf pine and white oak. Early descriptions suggest an understory that was dominated by grasses and forbs. Postsettlement logging, grazing, and fire suppression have greatly altered the landscape, which now consists of closed-canopy oak forests averaging 13 m^2/ha of basal area. Although the area contains more than 500 herbaceous species, woody plants and hardwood leaf litter dominate the herb-layer cover. The authors state that fire is being applied to create a "mosaic of high quality native habitats." Five separate units were burned 1 to 4 times (1998-2001). Herb-layer data were collected in 250 plots (4,000 1-m^2 quadrats) prior to fires, and again in 1998 and 2001. In all, 486 species (99 percent native) were recorded.

By 2001, forest structure had become somewhat more open on the burned units (all five combined) as understory (1 to 4 cm d.b.h.) and midstory (4 to 11 cm d.b.h.) stems were reduced by 47 and 28 percent, respectively. Fires increased the cover and frequency of herbaceous plants relative to woody plants. Among herbs, the cover of legumes more than doubled to 8 percent, and the cover of other forbs, grasses, and sedges all increased over pretreatment levels, but each remained at less than 5 percent total cover. Comon herbaceous species that increased in relative importance included *Panicum boscii* (Bosc's panic grass), *Carex nigromarginata* (black-margined sedge), *Brachelytrum erectum* (long-awned wood grass), *Helianthus hirsutus* (woodland sunflower), and *Lespedeza intermedia* (wand lespedeza). Total vegetation cover in the herb layer increased from 15 to 24 percent on burned sites while hardwood litter cover was reduced by 30 percent. Over 5 years, small-scale species richness increased slightly on burned sites (11.5 to 12.2 species/1 m^2) as did the total number of species recorded each year (465 to 482).

The authors concluded that the prescribed fires had created a "landscape in transition" as the understory was more open but the herb-layer response had been moderate. They hypothesized that the longer-term application of periodic fire (20+ years) seems necessary to restore the area to a "more open and biologically diverse landscape."

Ohio Dry-Mesic Oak Forest (Hutchinson et al. 2005a)

Our study was conducted on four 75-ha sites in the dissected Allegheny Plateau of southern Ohio. Presettlement forests were dominated by oak. Currently, second-growth forests that established prior to fire suppression (ca. 1850 to 1900) remain oak-dominated, but shade-tolerant trees (e.g., red maple) are abundant in the midstory and understory. Tree basal area averaged 28 m^2/ha. Fire treatments from 1996-99 were no fire, burned 2X, and burned 4X. Species' frequencies were recorded annually in 108 plots (1,728 2-m^2 quadrats) over a 5-year period (1995-99). Plots were stratified by an integrated moisture index into xeric, intermediate, and mesic classes. Overall, 452 species (97 percent native) were recorded.

Fires reduced the density of saplings (<10 cm d.b.h.) by more than 80 percent. The 2X and 4X burn treatments produced similar results. Species composition was significantly affected by fire but differences between unburned and burned sites were relatively minor compared to compositional differences between dry and mesic sites. Species composition shifted on all moisture classes after fire, but to a greater degree on dry plots. Total species richness on burned sites increased from 17.1/2 m^2 before fires (1995) to 18.5/2 m^2, averaged across post-burn years. On unburned plots, richness decreased slightly from 15.3 to 15.1/2 m^2.

Among species groups, richness increased significantly for annual forbs, summer-season perennial forbs, grasses, and woody seed-banking species, while tree-seedling richness decreased. Perhaps the most striking result was that the vast majority of perennial forb species exhibited little change in frequency on burned sites over the 5-year study (Appendix A). Only 7 of 49 common forb species exhibited a change in frequency of ≥ 5 percent; only one of those, *Viola* spp. (Violets), changed more than 10 percent, increasing on burned units from a mean of 30.5 percent before fire to 49.2 percent on the burn sites after 5 years.

In a separate posttreatment sample of 480 1-m^2 quadrats located in six 2-ha stands within the same study sites (3 years after the last fire), herbaceous cover averaged 17.1 percent in burned sites vs. 4.7 percent in unburned sites (Hutchinson 2004). For each of the major herbaceous species groups (grasses, sedges, legumes, composits), cover was higher on the burned sites (Fig. 1); the combined cover of the these four groups averaged 12.3 and 3.3 percent in burned and unburned stands, respectively.

Southern Illinois Oak Barrens (Taft 2003)

This study was conducted on a 3-ha dry sandstone barrens owned by The Nature Conservancy, and also on a nearby unburned barrens in the Shawnee National Forest. Both barrens were dominated by post oak (*Quercus stellata*). Tree basal area averaged 18 m^2/ha. Herb-layer cover was high at both sites (>90 percent) prior to treatment. Fire was applied to the barrens

Figure 1.—Photograph (2005) of a ridgetop site in southern Ohio that was burned annually 1996-1999 and again in 2004. Here the overstory is dominated by white oak, the sapling layer has largely been eliminated, and common species in the herbaceous layer include *Panicum commutatum*, *Panicum boscii*, and seedlings of sassafras and oak.

to prevent its conversion to dry forest and sustain and enhance the herbaceous flora of this rare natural community. Fires were conducted in 1989 (fall) and 1994 (spring). Herb-layer vegetation was sampled in 1989 (pretreatment) and four posttreatment years. Twenty-three plots (276 0.25-m^2 quadrats) were sampled each year.

Density of stems in the shrub/sapling layer (50 cm height to < 6 cm d.b.h.) was reduced by 55 percent on the burned site after fire. Herbaceous cover increased from 90 to 117 percent on the burned sites and concurrently decreased from 98 to 61 percent on the unburned site. Small-scale species richness more than doubled on the burned site, to > 8 species/0.25 m^2 while remaining static on the unburned site. The total number of species sampled increased from 94 to 121 on the burned site and decreased from 74 to 68 species on the unburned site.

Most of the species that increased on the burned site were perennial forbs, grasses (e.g., several *Panicum* species, the most abundant of which was *P. laxiflorum,* pale green panic grass), and sedges typical of dry woodland habitats. The most common mode of establishment after fire was germination of seed stored in the soil. The only two herbaceous species to decrease after fire were, surprisingly, the prairie grasses *Schizachyrium scoparium* (little bluestem) and *Sorgastrum nutans* (indian grass), presumably because light remained limiting to their productivity.

The author concluded that periodic fire (every 3 to 4 years) is necessary to maintain the composition and diversity of the barrens flora. Without fire to stimulate the seedbank, some less common species likely would become locally extinct as woody succession proceeds.

FIRE EFFECTS ON HERB COMMUNITIES: GENERALIZATIONS

When observing a typical oak forest that has been burned recently, the most noticeable effects of fire are: 1) a more open structure caused by topkill of the sapling and shrub layer, 2) the prolific sprouting of most woody plants, and 3) the greater cover of herbaceous plants, including forbs, grasses, and sedges. Nearly all studies of fire effects, in addition to those described earlier, have revealed that the cover and/or abundance of herbaceous plants increased after fire (e.g., Swan 1970; McGee et al. 1995; Nuzzo et al. 1996; Arthur et al. 1998). For woody plant cover, the fire response may be more variable because of the variation in the sprouting response in different vegetation types (e.g., Nuzzo et al. 1996; Elliot et al. 1999; Kuddes-Fischer and Arthur 2002). The abundance of woody stems often increases because of sprouting (e.g., Ducey et al. 1996; Arthur et al. 1998), and foliage cover is shifted from the midstory/understory to the herb layer.

As with the three case studies, fire often has been shown to increase species richness and/or diversity in the herbaceous layer (e.g., Arthur et al. 1998; Nuzzo et al. 1996; Elliot et al. 1999), though results vary among studies. By contrast, Ducey et al. (1996) showed that diversity was reduced after fire in some areas, likely due to abundant sprouting from *Kalmia latifolia* (mountain laurel). Several studies have shown no significant effects of fire on herb-layer diversity (Luken and Shea 2000; Kuddes-Fischer and Arthur 2002; Franklin et al. 2003).

The increases in small-scale species richness after fire largely result from species already present on the site that increase via germination after fire. Species groups that commonly respond to fire include woodland grasses (e.g., *Panicum* spp.), sedges, composites, legumes (e.g., *Desmodium spp*, *Lespedeza* spp.), and woody seedbanking species (e.g., *Liriodendron tulipifera*, *Rubus* spp.).

In southern Ohio we have documented the establishment of several species on burned sites that were absent or rare before fire, including *Ceanothus americanus* (New Jersey Tea), *Phaseolus polystachios* (wild kidney bean), *Clitoria mariana* (butterfly pea), *Rhus glabra* (smooth sumac), *Chamaecrista nictitans* (wild sensitive-plant) *Rhus copallinum* (winged sumac), *Desmodium cuspidatum* (large bract tick trefoil), *Eupatorium sessilifolium* (upland boneset), *Eupatorium serotinum* (late-flowering thoroughwort), *Phytolacca americana* (pokeweed), *Hackelia virginiana* (common stickseed), *Lobelia inflata* (indian-tobacco), *Sphenopholis nitida* (shining wedge grass), and *Helianthus microcephalus* (small wood sunflower). Although there is the potential for invasive exotic plants to establish or to increase in abundance after fire (Huebner, this volume), most studies have reported only minor changes in the abundance exotic plants.

FIRE EFFECTS ON HERB COMMUNITIES: CAUSES OF VARIATION

Fire intensity can vary dramatically both among different fires and across the landscape within a fire. Although most prescribed fires have relatively little affect on overstory trees, intense surface fires can cause significant overstory mortality (Regelbrugge et al. 1994; Ducey et al. 1996; Elliot et al. 1999). In Connecticut oak forests with abundant *Kalmia latifolia* (mountain laurel), Ducey et al. (1996) found that postburn plant diversity was higher in intensely burned patches, where the overstory had been killed, than moderately burned patches, where *Kalmia* was the most dominant after sprouting.

In landscapes with significant topographic heterogeneity, vegetation also varies across the landscape. In the southern Appalachians, Elliot et al. (1999) found that prescribed fire intensity and effects on the overstory and understory increased along an upslope moisture gradient from a mesic cove community to dry midslope oak community to a xeric ridge pine-oak community. In southern Ohio, where total elevation changes are less pronounced than in the southern Appalachians, we found that fire intensity and effects were less pronounced from dry to mesic sites (Hutchinson et al. 2005a). In other ecosystems, fire has reduceed differences in vegetation across topographic gradients (e.g., Gibson and Hulbert 1987; tallgrass prairie) or reinforced differences (Liu et al. 1997, longleaf pine forest).

Fire-season effects on herb-layer vegetation seldom have been compared in oak forests. In a dry-mesic "degraded" forest in northern Illinois, Schwartz and Heim (1996) found that the abundance and richness of native herbs was reduced for several years by a May growing-season fire but not by a March dormant-season fire. In restored shortleaf pine grassland communities in Arkansas, Sparks et al. (1998) compared late dormant-season burns (March-April) to late growing-season burns (September-October) and found that differences were relatively minor. However, March and July burns produced different vegetation responses for some early-season and late-season species in planted tallgrass prairies (Howe 1995). In longleaf pine savannas, growing-season burns are more effective at suppressing midstory deciduous trees (*Quercus* spp.) and understory shrubs than dormant-season fires (Glitzenstein et al. 1995; Drewa et al. 2002).

Not surprisingly, different fire frequencies applied over a short period have produced largely similar herb layer vegetation responses in oak forests (e.g., Hutchinson et al. 2005a). However, in other vegetation types where different fire frequencies have been applied over several decades, vegetation response can be significantly different. For example, long-term differences in fire frequency at Konza Prairie in Kansas indicate that frequent burning (annual or biennial) tends to homogenize vegetation by favoring the dominant warm-season grass *Andropogon gerardii* (big bluestem) to a much greater extent than infrequent burning (Collins 1992).

When prescribed fire has been applied to more open-structured communities within oak forest landscapes in which woodland and/or prairie species have persisted, fire effects usually are more pronounced compared to fire effects in closed-canopy oak forests. In addition to the oak barrens study described (Taft 2003), Nuzzo (1996) reported a 50-percent increase in herb-layer richness after prescribed fires in a low-density Illinois "sand forest." In a 25-year study of fire effects in a Tennessee oak barrens, unburned plots showed a sharp decline in herbaceous cover as woody succession proceeded, while burned plots maintained high coverage of both forbs and graminoids throughout (DeSelm and Clebsch 1991).

EFFECTS OF SILVICULTURE AND FIRE ON THE HERBACEOUS LAYER

Although prescribed fire is being applied more often to sustain oak forests, research has shown that fire alone, at least in the short term, often does not improve the competitive status of oak regeneration because the canopy remains closed and competing species also sprout readily (Brose, this volume; Hutchinson et al. 2005b). The combined use of silvicultural treatments that remove a portion of the canopy followed by prescribed fire has improveed oak regeneration in some cases (Kruger and Reich 1997; Brose and Van Lear 1998).

The effects of timber harvesting on herb communities has received much attention, some of which has been controversial. Although results differ substantially among studies (Roberts and Gilliam 2003), some of the more thorough studies have shown no reduction in herb layer diversity following timber harvest, either in the short term (e.g., Grabner and Zenner 2002) or in the longer term (e.g., Elliot et al. 1997).

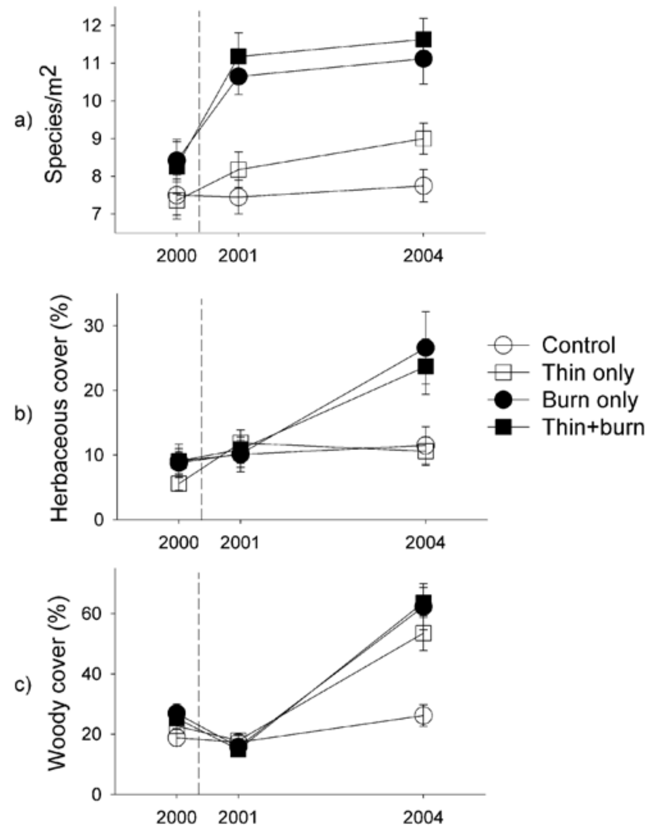

Figure 2.—Response of herbaceous layer vegetation to thinning, burning, and thinning+burning treatments at two replicate sites (Raccoon Ecological Management Area and Zaleski State Forest) of the Ohio Hills Fire and Fire Surrogate study site. Mean annual values (± 1 S.E.) for a) species richness per 1 m^2 quadrat, b) the combined foliar cover (percent) of all herbaceous species <1 m above the forest floor, and c) combined foliar cover (percent) of all woody species <1 m above the forest floor. In each graph, the dashed line shows the timing of both thinning and burning treatments: thinning occurred from autumn 2000 to spring 2001; fires were conducted in March and April 2001. Herb-layer data were collected in the summers (June-August) of 2000, 2001, and 2004.

Few studies have examined the effects of fire, harvesting, and combinations of the two on herb-layer vegetation in oak forests. The national Fire and Fire Surrogates Study is quantifying the effects of fire and other treatments that alter forest structure (e.g., mechanical thinning) at 13 sites across the country. At the Ohio Hills Site, we are monitoring the response of herb layer communities to fire only, thinning only, and thinning followed by fire. Preliminary data indicate clear differences in herb-layer response to thinning and fire treatments (Fig. 2). Species richness and herbaceous cover have increased much more after fire (and thinning+fire) than after thinning alone. By contrast, the total cover of woody plants (shrubs and tree seedlings+sprouts), has increased substantially on all treatments, relative to untreated stands. These results suggest that fire as a process is a unique disturbance that may be required to stimulate the seedbank and increase the diversity and productivity of the oak-forest herbaceous layer. Two other short-term studies of thinning and fire have shown less substantial responses of herb-layer vegetation (Franklin et al. 2003; Dolan and Parker 2004).

RARE PLANTS AND FIRE

Owen and Brown (2005) divided the 186 federally listed plant species into four categories of fire adaptation and tolerance: 47 species require fire, 65 tolerate fire, 70 occur in habitats where fire does not occur (e.g., aquatic plants), and are adversely affected by fire.

Throughout the central hardwoods region, prescribed fire is used to sustain and restore open-structured plant communities (e.g., barrens, woodlands) that usually support rare species. However, maintaining rare plant populations is seldom the primary objective of fire use in oak forests. Some rare plants that occur in upland oak forests, particularly those threatened by shading, could potentially benefit from prescribed fire through increased germination, establishment, and reproduction. In Ohio, I compiled a list of 125 state-listed rare plant species that occur primarily in the "hill country" of southern Ohio. I excluded species that occur in habitats where fire does not occur. For each species, the Ohio Department of Natural Resources, Division of Natural Areas lists 13 primary threats to the populations (http://www.dnr.state.oh.us/dnap/heritage/Rare_Species2004.html). By far the most common threat among the 125 species is "shading as a result of woody plant succession," which applies to 71 species. Prescribed fire is a potential tool for maintaining the open habitats required by these species. However, for some of these species, other methods (e.g., mechanical removal of trees, herbicide application, mowing) could be more effective or achieved more readily than burning.

For nearly all rare plants that occur in oak forest landscapes, the effects of fire are unknown. In southern Ohio, I quantified the response of the state-endangered *Calamagrostis porterii* subsp. *insperata* (Bartley's bent reed grass) that occurs as distinct clonal patches, primarily on dry-mesic ridgetops in oak forests (Hutchinson 2004). Tiller density increased in patches burned annually from 1996 to 1999 compared with unburned patches, though cover and patch area were unaffected. Annual fires also stimulated flowering, which has been observed only infrequently in natural settings. From 1995 to 2001, 125 flowering stems were documented on the seven annual burn patches (total area of patches = 229 m^2) compared to only 13 flowering stems on the 33 unburned patches (area = 999 m^2).

MANAGEMENT IMPLICATIONS

The use of prescribed fire to restore oak-forest structure, and to improve oak regeneration and wildlife habitat is a critically important management issue, as evidenced by the nearly 400 attendees at this conference. As the use of prescribed fire in oak forests is expanding rapidly, it is important to better understand its effects on other ecosystem components such as the diverse herbaceous layer. Research indicates that typical prescribed fires, those conducted during the dormant season and of low to moderate intensity, can be applied to oak forests without having a major impact on the herbaceous layer vegetation. The increase in herbaceous diversity and cover, documented after fire in most studies, is desirable to many land managers. While the implications of fire management are perhaps less obvious to the herbaceous layer of oak forests than in more open communities, such as barrens and woodlands, the fire-induced germination of some species and the more open conditions promoting their growth and reproduction, likely are important for the long-term maintenance of species diversity. Local populations of some rarer species may be threatened by the continued exclusion of fire.

Although research has shown general patterns of herb-layer response to fire, the implications of fire management likely will vary substantially across the central hardwoods region and also within local landscapes. To promote landscape-scale plant diversity, it may be desirable to apply periodic fire to dry upland sites while continuing to exclude fire from mesic sites. This approach is being used in the southern Appalachians where ridgetop ignitions of large units result in fires that burn completely and at moderate to high intensity in xeric uplands, contrasting with a patchy mosaic of low-intensity and unburned areas in mesic low slopes (Hugh Irwin, personal communication).

Studies of fire in oak forests have not shown significant establishment or increased abundance of invasive exotic plant species (Huebner, this volume). However, because many invasives are adapted to disturbed conditions for germination and growth, fire-induced alterations to the forest floor and canopy could facilitate their establishment, particularly if invasives are common and fires are of high intensity and/or combined with harvesting. Prior to burning, invasives should be treated to reduce the likelihood of postburn establishment. The abundant establishment of invasive species after fire almost surely would have a negative impact on native herbaceous communities.

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APPENDIX

Mean frequency of common species over 5 years on burn units on four study sites in southern Ohio. Each site contained a unit burned 2X (1996 and 1999) and a unit burned 4X (1996 to 1999); here the burn treatments have been combined. Each year sixteen 2-m^2 quadrats were sampled within each of nine 625-m^2 plots per burn unit (total of 72 plots).

		Frequency (%)				
Scientific name	Common name	1995 (preburn)	1996	1997	1998	1999
Graminoids						
Brachyelytrum erectum	long-awned wood grass	21.3	22.2	25.3	22.4	27.2
Bromus pubescens	woodland brome	6.5	3.8	4.8	4.8	5.4
Carex digitalis	narrow-leaved wood sedge	9.9	6.1	6.3	10.0	6.7
Carex gracilescens	slender wood sedge	10.9	9.6	13.6	12.0	15.0
Carex rosea	stellate sedge	4.5	4.0	3.4	4.2	4.6
Carex spp.	sedge spp.	6.9	27.4	17.6	22.1	37.6
Carex wildenovii	Willdenow's sedge	12.8	5.7	10.2	14.0	10.9
Danthonia spicata	poverty oat grass	4.6	2.3	3.6	5.4	5.7
Diarrhena americana	beak grass	2.7	3.3	2.8	3.0	3.8
Panicum boscii	Bosc's panic grass	15.1	14.9	21.6	25.0	29.8
Panicum commutatum	variable panic grass	0.7	7.0	20.7	22.0	23.1
Panicum dichotomum	forked panic grass	6.2	7.9	10.2	12.5	13.3
Poa cuspidata	cuspidate spear grass	16.8	13.3	16.4	12.8	12.5
Annual forbs						
Acalypha virginica	three-seeded mercury	1.5	10.9	10.9	11.7	18.2
Amphicarpaea bracteata	hog-peanut	20.9	23.1	26.2	25.9	21.5
Erechtites hieracifolia	fireweed	5.9	72.7	35.9	30.5	34.8
Galium aparine	cleavers	12.3	3.1	10.6	8.3	0.7
Pilea pumila	common clearweed	7.9	10.1	9.8	11.2	10.9
Perennial forbs						
Agrimonia spp.	agrimony spp.	5.9	5.1	7.6	6.5	5.7
Anemonella thalictroides	rue anenome	24.7	24.3	25.7	25.7	25.6
Arisaema triphyllum	jack-in-the-pulpit	19.1	17.7	17.4	19.0	18.6
Aristolochia serpentaria	Virginia-snakeroot	10.3	12.5	12.2	11.3	11.9
Asarum canadense	wild ginger	13.5	10.5	11.5	11.9	10.1
Aster divaricatus	white wood aster	10.3	9.7	12.3	13.9	10.1
Aureolaria laevigata	entire-leaved false foxglove	2.2	2.3	2.5	3.7	4.7
Cardamine angustata	slender toothwort	6.8	3.0	7.0	6.2	6.3
Chimaphila maculata	spotted pipsissewa	4.9	4.2	4.1	4.6	0.7
Cimicifuga racemosa	black cohash	14.7	15.7	16.0	15.0	16.2
Circaea lutetiana	c. enchanter's nightshade	11.1	8.9	9.9	10.7	10.6
Collinsonia canadensis	richweed	3.2	5.6	5.2	5.6	5.4
Conopholis americana	squaw-root	3.6	3.9	2.9	4.4	4.5
Cunila oreganoides	common dittany	4.1	3.6	4.2	4.4	5.1
Desmodium glutinosum	cluster-leaved tick-trefoil	11.5	12.2	10.6	12.4	11.4
Desmodium nudiflorum	naked tick-trefoil	43.8	43.1	42.4	46.3	38.7
Dioscorea quaternata	wild yam	12.5	11.9	12.0	13.3	12.2
Eupatorium rugosum	white snakeroot	14.8	19.8	23.9	24.9	22.7
Galium circazans	wild licorice	27.8	21.3	28.6	31.3	31.9
Galium concinnum	shining bedstraw	6.4	6.5	7.2	6.9	7.6
Galium triflorum	sweet-scented bedstaw	30.3	42.4	43.8	44.0	37.4
Geranium maculatum	wild geranium	35.8	33.9	31.1	31.3	29.8
Geum spp.	avens spp.	3.1	3.1	4.6	3.4	3.3

APPENDIX—continued

Helianthus divaricatus	woodland sunfower	5.5	10.8	8.0	7.4	6.8
Hieracium venosum	veined hawkweed	4.3	3.1	3.2	4.3	3.7
Hydrophyllum macrophyllum	large-leaved waterleaf	5.3	4.2	4.6	4.6	5.0
Lespedeza spp.	lespedeza spp.	1.5	4.8	11.6	7.9	11.2
Lysimachia quadriflora	whorled loosestrife	7.0	7.8	8.6	12.1	10.3
Medeola virginiana	indian cucumber-root	3.2	2.8	3.6	3.9	3.3
Monarda fistulosa	wild bergamot	5.1	4.2	5.6	5.8	4.4
Osmorhiza claytonii	wooly sweet cicely	9.1	6.2	6.1	6.2	4.4
Oxalis violacea	violet wood-sorrel	2.6	3.9	2.9	4.6	4.5
Phlox divaricata	blue phlox	6.2	7.3	6.3	4.9	4.2
Podophyllum peltatum	mayapple	9.7	8.7	8.9	8.7	8.1
Polygonatum biflorum	smooth solomon's seal	13.6	6.0	13.5	9.9	12.9
Polygonum virginianum	jumpseed	6.5	10.2	8.2	7.1	8.6
Potentilla spp.	cinquefoil	14.4	13.7	18.8	19.5	19.5
Prenanthes spp.	rattlesnake root	14.6	15.5	15.5	14.3	11.5
Sanguinaria canadensis	bloodroot	6.6	5.8	7.3	8.2	6.9
Sanicula spp.	snakeroot	16.1	12.4	18.8	16.9	15.9
Scutellaria spp.	skullcap	10.2	10.8	10.5	11.7	11.8
Smilacinia racemosa	false solomon's seal	31.4	15.6	29.7	26.7	28.0
Solidago caesia	blue-stemmed goldenrod	13.5	16.8	14.1	17.4	16.3
Solidago flexicaulis	zigzap goldenrod	3.2	5.4	4.9	5.0	3.7
Tiarella cordifolia	foamflower	11.3	10.5	11.6	10.9	10.2
Trillium grandiflorum	large white trillium	16.5	7.5	16.8	15.1	16.7
Uvularia perfoliata	peroliate bellwort	38.3	28.9	34.5	35.8	36.1
Vicia caroliniana	pale vetch	3.0	5.0	5.1	4.3	2.6
Viola spp.	violets	30.5	52.2	55.2	55.1	49.2
Pteridophytes						
Adiantum pedatum	maidenhair fern	3.3	3.3	3.9	3.3	3.3
Botrychium virginianum	rattlesnake fern	10.3	4.9	10.5	8.4	9.2
Osmunda claytonia	interrupted fern	1.5	1.6	2.3	2.0	1.9
Polystichum acrosticoides	Christmas fern	19.5	19.0	21.1	20.4	21.4
Thelypteris hexagonoptera	broad beech fern	3.6	4.3	4.5	4.8	4.5
Shrubs						
Corylus americana	American hazel	4.6	5.8	4.2	4.7	3.0
Hamamelis virginiana	witch-hazel	4.8	4.3	4.8	3.8	3.6
Hydrangea arborescens	wild hydrangea	7.5	7.1	8.0	8.9	7.6
Lindera benzoin	spicebush	17.3	15.7	16.5	16.2	12.8
Rhus glabra	smooth-sumac	0.0	12.0	4.1	6.4	13.9
Rosa carolina	pasture rose	14.1	13.9	12.5	13.9	13.1
Rubus spp.	brambles	22.1	28.3	41.1	40.2	43.5
Smilax glauca	sawbrier	27.6	28.0	27.3	29.1	30.9
Smilax hispida	bristly greenbriar	5.7	6.7	5.2	6.4	5.6
Smilax rotundifolia	common greenbriar	41.0	39.4	37.2	36.5	34.5
Vaccinium palidum	low blueberry	21.0	22.0	21.8	21.3	21.3
Vaccinium stamineum	deerberry	7.1	3.0	4.0	3.1	2.7
Viburnum acerifolium	maple-leaved viburnum	23.3	19.8	19.6	19.5	17.8
Vines						
Parthenocissus quinquifolius	Virginia creeper	38.5	36.9	32.9	33.1	25.7
Toxicodendron radicans	poison ivy	15.7	22.2	9.5	10.9	8.2
Vitis sp.	wild grapvine	21.3	60.3	47.8	42.4	37.4

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Fire - Fauna Interactions

DO FIRE AND INSECTS INTERACT IN EASTERN FORESTS?

Lynne K. Rieske-Kinney[1]

Abstract.—The increasing use of prescribed fire as a management strategy for manipulating forest-species composition generates questions regarding the effects on the arthropod community and the underlying processes in which arthropods play a dominant role, as well as its potential as a pest suppression strategy. Despite the apparent benefits of prescribed burning for manipulating stand composition and enhancing tree vigor, relatively little is known about how fire interacts with arthropod-dependent processes in eastern forest ecosystems. This paper reviews the evidence of direct and indirect interactions between forest arthropods and fire, and addresses the following questions: 1) are soil- and litter-dwelling arthropods irreparably harmed by burning? 2) does prescription burning alter plant susceptibility to insect herbivores? 3) can fire be used as a management strategy to suppress forest arthropod pests?

Although soil- and litter-dwelling arthropod abundance is affected by prescribed burning, arthropod diversity and richness are not. Litter arthropod evenness increases in response to burning, most likely due to reductions in mites and collembolans, the two dominant taxa. Fire-induced changes in foliar chemistry often are transient and may be species-specific. These changes are not fully predictable but could alter patterns of insect herbivory. Use of prescribed fire for pest suppression in managed forests has lagged behind that of other managed systems, and it has had limited use for pest suppression in deciduous forests of the Eastern United States. The highly clustered spatial distribution of acorn predators makes effective suppression through prescription burning problematic.

INTRODUCTION

Prescribed fire is increasingly used as an intermittent disturbance agent to mimic pre-settlement disturbance regimes. This management strategy is designed to encourage regeneration and stand development (Lorimer 1993). Prescribed fire combusts the leaf litter and can reduce total microbial and fungal biomass (Fritze et al. 1994). Burning can lead to increased soil pH and greater fluctuations in temperature and moisture, with subsequent loss of vegetation (Haimi et al. 2000). Burning suppresses vegetative competition and enhances light penetration to the forest floor. In oak-dominated forests of the Eastern United States, fire helps reduce invasive fire-sensitive maples (Arthur et al. 1998), and makes conditions more favorable for development and growth of oak seedlings (Reich et al. 1990; Adams and Rieske 2001). The increasing use of prescribed fire as a management strategy for manipulating forest-species composition generates questions regarding the effects on the arthropod community and on the underlying processes in which arthropods play a vital role.

Insects and related arthropods are essential to forest ecosystem processes. Arthropods dominate forest soils and leaf litter, and play vital roles in litter decomposition, nutrient dynamics, soil development, and soil stability (Wood 1995). Arthropods contribute to decomposition of organic matter and soil development by reducing the size of organic particles, consequently accelerating fungal and bacterial decomposition (Metz and Dindal 1975). Arthropod fungivores also stimulate fungal growth by hyphal grazing, thereby influencing the balance between fungi and bacteria (Hanlon and Anderson 1979).

As herbivores, arthropods also play critical roles in ecosystem processes. Herbivores can function as seed predators, causing extensive, localized loss of regeneration in eastern oak forests (Drooz 1985). Acorn predators such as acorn weevils, sap beetles, and the acorn moth are primary and/or secondary pests (feeding and breeding in intact versus damaged seed), consuming acorn cotyledons and damaging radicles, or directly damaging germinated seedlings (Van Leeuwen 1952; Gibson 1964; Galford and Weiss-Cottrill 1991; Galford et al. 1988, 1995). Acorn predators are closely associated with soil and litter and as such may vector

[1]University of Kentucky, Department of Entomology, Lexington, KY 40546-0091, 859-257-1167, email: lrieske@uky.edu.

and/or create infection courts for potentially pathogenic microorganisms. Additionally, these arthropods are strongly affected by factors such as stand composition, litter quality and depth, and soil type and depth, which influence their local distribution and impact (Dindal 1990).

Defoliating herbivores alter net primary productivity, increasing nitrogen input and light penetration. Indirectly, defoliators can influence watershed characteristics and, through changes in resource availability, affect wildlife distribution patterns. Herbivore population outbreaks cause widespread mortality that can shift forest-stand composition and influence ecological succession (Drooz 1985). Insect herbivory is one of a suite of disturbance factors that have contributed to the development of today's forests. In addition to herbivore pressure, intermittent disturbances consisting of pathogens, anthropogenic activities, climactic pressures, and fire have influenced the formation and maintenance of forests of Eastern North America (Abrams 1992).

To mimic historic disturbance patterns, fire is being used as a management strategy in eastern deciduous forests. Despite the apparent benefits of prescribed burning, relatively little is known about how fire interacts with arthropod-dependent processes in eastern forest ecosystems. In this paper I review the evidence of direct and indirect interactions between forest arthropods and fire, and addresses the following questions: 1) are soil- and litter-dwelling arthropod populations irreparably harmed by burning? 2) does prescription burning alter plant susceptibility to insect herbivores? 3) can fire be used as a management strategy to suppress forest arthropod pests?

Are soil- and litter-dwelling arthropods irreparably harmed by burning?

Relatively large, highly mobile arthropods are unlikely to be negatively impacted by prescribed fire. Oak savanna arthropod communities in the Upper Midwest were unaffected by burning (Siemann et al. 1997). In central Ohio, prescribed burning had no negative effect on large predatory carabid beetles (Smith and Horn 2000), or on scavenging scarab beetles (Stanton et al. 2000). Similarly, in south-central Kentucky, prescribed fire had little effect on ground-dwelling arthropod abundance, richness, and diversity, but arthropod community evenness increased in response to a single-burn disturbance (Coleman and Rieske, unpublished).

For less mobile arthropods functioning on the forest floor, the direct effects of fire include immediate mortality and habitat destruction. Loss of these faunal groups to fire could potentially disrupt essential ecosystem functions. One year following a prescribed fire in hardwood forests of eastern Kentucky there was a reduction in total dry mass of soil- and litter-dwelling invertebrates (Kalisz and Powell 2000). The loss was attributed primarily to direct mortality of Coleopteran larvae, which serve several critical roles on the forest floor as predators, herbivores, and detritivores (Dindal 1990).

To examine the period of recovery needed to obtain pre-burn levels of arthropods following a prescribed fire regime using single- and multiple-burns, Coleman and Rieske (unpublished) monitored plots for two post-burn growing seasons. Mites (Acari) and springtails (Collembola) dominate at the soil/litter interface, and populations were devastated by both fire regimes. Mite abundance did not recover over the course of the study. Arthropod abundance was affected by the prescribed burns, but arthropod diversity and richness were not. Litter arthropod evenness increased as a consequence of burning, probably due to reductions in the two dominant taxa.

In addition to the direct effects of habitat loss and mortality, burning reduces the available resource base, resulting in bottom-up regulation of soil/ litter community dynamics. Fire also may indirectly affect arthropod communities by changing plant species composition and foliar accessibility (Mitchell 1990), and by altering plant phenology. Changes in flowering phenology could affect insect pollinators, pollination rates, and subsequent seed set, though there are no studies addressing this.

Does prescription burning alter plant susceptibility to insect herbivores?

Fire suppresses vegetative competition and increases soil nutrients (Reich et al. 1990), and enhances seedling vigor (Adams and Rieske 2001), which may reduce the negative effects of herbivory. Fire also may indirectly affect arthropod communities by changing plant-species composition and foliar characteristics. Combustion of the litter layer provides an influx of nutrients to the forest floor, which can influence plant foliar chemistry through changes in nutrient availability and light intensity (Reich et al. 1990; Kruger and Reich 1997; Arthur et al. 1998), potentially altering herbivore feeding patterns (Rieske 2002; Rieske et al. 2002; Adams and Rieske 2003).

Fire-induced changes in foliar chemistry are not fully predictable and may be species-specific. Reich et al. (1990) and Kruger and Reich (1997) found enhanced, though transient, leaf nitrogen levels in northern red oak seedlings. Similarly, chestnut oak seedlings sampled in the post-burn growing season had higher foliar nitrogen and water content following a wildfire than seedlings sampled from unburned sites (Rieske 2002). By contrast, Kruger and Reich (1997) found a transient elevation in red oak seedling foliar carbohydrates following an early spring, low-intensity surface fire, whereas chestnut oak seedlings from burned sites had transient declines in foliar carbohydrates, and higher initial tannin levels (Rieske 2002). Although it is possible that chestnut oak and northern red oak seedlings respond differently to fire with respect to foliar carbon levels, the differences in seedling response may be attributed to the timing and intensity of the fires. Regardless, fire-induced changes may not be detectable, due to the buffering capacity of mature trees (Rieske et al. 2002), spotty fire coverage over irregular terrain, or to the relatively cool nature of some prescribed burns (Adams and Rieske 2003).

To assess the extent to which fire interacts with seedling herbivory, Adams and Rieske (2001) assessed mammalian and arthropod herbivore pressure on white oak seedling growth and vigor in burned forests in Kentucky. They found that herbivory is a measurable force impacting white oak seedlings but that prescribed fire did not affect herbivore pressure. They also found that the mammalian component of the herbivore complex had a greater impact on white oak seedling growth than the arthropod component.

Can fire be used as a management strategy to suppress forest arthropod pests?

Fire has been used extensively in agricultural, prairie, and rangeland ecosystems to manipulate vegetative composition and alter host plant quantity and quality, thereby directly or indirectly manipulating arthropod pest populations (Miller 1979). Successful suppression using fire requires that the pest be spatially and temporally vulnerable to fire-induced mortality (i.e., in the soil at the appropriate time), and that the fire itself does minimal damage to the standing crop of trees. Use of prescribed fire for pest suppression in managed forests has lagged behind that of other managed systems and has focused primarily on boreal forests (Mitchell 1990; Brennan and Hermann 1994; McCullough et al. 1998). The intricate link between fire and bark beetles as episodic disturbance agents in coniferous systems is now widely accepted, and recognition of this relationship gives land managers the opportunity to minimize harmful outcomes of these interactions.

Prescription burning has had limited use as a pest suppression strategy in deciduous forests of the Eastern United States (Brennan and Hermann 1994; McCullough et al. 1998). Historically, forest tracts in Massachusetts were burned to kill overwintering egg masses and clusters of feeding caterpillars as part of early attempts to eradicate the gypsy moth, *Lymantria dispar*, with mixed success (Doane and McManus 1981). More recently, prescribed fire was used to suppress populations of pear thrips, *Taeniothrips inconsequens*, in northern sugar maple stands. In addition to the direct mortality to overwintering thrips populations, fall burning seems to disrupt the phenological synchrony between emerging thrips and sugar maple bud expansion that is critical to the success of these insects (Brose and McCormick 1992). Attempts to reduce acorn mortality caused by the acorn weevil, *Conotrachelus posticatus*, and associated acorn predators using prescribed fire in eastern oak forests have met with some success (Wright 1986; Roccardi et al. 2004; Rieske, unpublished). The highly clustered spatial distribution of these seed feeders makes effective suppression and effective sampling problematic.

CONCLUSIONS

Although large, highly mobile arthropods appear to escape the negative effects of fire, the smaller, less mobile arthropods that are critical to decomposition processes are unable to escape. Arthropod abundance at the soil/litter interface can be severely reduced by burning, while effects on diversity, richness, and evenness are less marked. The use of repetitive prescribed fire to encourage regeneration and enhance stand development should be based on a schedule to allow appropriate time for leaf litter habitats to return to pre-burn conditions, and allow for resurgence of the arthropod community.

Fire-induced changes in oak foliar chemistry include increases in foliar nitrogen, carbohydrates, and defensive compounds, which can influence herbivore feeding patterns. Many of these changes are transient and not fully predictable; they may be species-specific and depend on the timing and intensity of the fire. Herbivory is a measurable force on oak seedlings, and mammalian herbivory has a greater impact on seedling vigor than arthropod herbivory.

Prescribed fire has limited use for pest suppression in deciduous forests of the Eastern United States. Pest vulnerability, both spatially and temporally, and the highly clustered spatial distribution of many pests makes effective suppression through prescription burning challenging and unpredictable. Although direct suppression of selected herbivores with prescribed fire is possible, prescription burning may be more viable as a means of enhancing forest health and redirecting succession, thereby reducing stand susceptibility to herbivore outbreaks.

Additional research is needed to fully understand the dynamics of insect-fire interactions in eastern oak forests. We lack a complete understanding of the effects of fire on the critical arthropod-driven processes in the soil/litter interface. More work is needed on fire-induced changes in herbivore pressure and herbivore susceptibility of important forest tree species.

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DOES FIRE AFFECT AMPHIBIANS AND REPTILES IN EASTERN U.S. OAK FORESTS?

Rochelle B. Renken[1]

Abstract.—Current information about the effect of fire on amphibians and reptiles in oak forests of the Eastern and Central United States is reviewed. Current data suggest that fire results in little direct mortality of amphibians and reptiles. Fire has no effect on overall amphibian abundance, diversity, and number of species in comparisons of burned and unburned plots, though salamander numbers tend to be greater in unburned plots. Current research also suggests that fire has no effect on reptile abundance, diversity, or number of species except in several studies in which lizard abundance was or tended to be greater in burned plots. The season of burn seems to make no difference in amphibian and reptile response. Although fire generally has little effect on amphibians and reptiles in oak forests, managers need to continue to consider the potential effect of fire on amphibians and reptiles associated with streams and forest pools, and on endangered threatened species and/or those of special concern. Managers can monitor the effects of fire on their own or with help from biologists, or can consult references to surmise how management affects these animals.

INTRODUCTION

Within the last 20 years, the effects of forest management on amphibians and reptiles has received increased attention. Typically, attention has focused on the effects of even-age and uneven-age forest management (see de Maynadier and Hunter 1995 for a review), but as fire has been gaining acceptance as a forest management tool, questions have been raised about the impact of fire (Russell and others 1999; Bury and others 2000; Pilliod and others 2003; Russell and others 2004). "Altered fire regime" was one of the reasons given for the problems experienced by 7 percent of 19 frog and toad species and about 17 percent of 49 salamander species classified as "net extirpations" in the United States (Bradford 2005). Amphibians and reptiles are some of the least visible animals in the forest, yet with increasing focus on amphibian declines (Phillips 1990; Wake 1991; Lannoo 2005), forest managers need to know the effects of their practices on these animals.

There are several reasons why amphibians and reptiles should be considered when making forest management decisions. First, they comprise a significant amount of biomass in forest systems. For instance, the biomass of salamanders alone within the Hubbard Brook Experimental Forest in New Hampshire was 2.6 times that of birds and approximately equal to the biomass of mice and shrews (Burton and Likens 1975). Reported densities for just the eastern red-backed salamander (*Plethodon cinereus*), one of the most common salamanders within oak forests, have reached 11,452 per acre (Mathis 1991). Second, amphibians require moisture in the environment for breeding and respiration. Amphibians lay their eggs within water or in moist places (e.g., moss or logs) where eggs can remain moist or be covered by water. Amphibians also have permeable skin (Bury and others 2000) and are in a much greater danger of desiccation than birds or mammals. There are forest-dwelling salamanders (salamanders of the family Plethodontidae) that lack lungs and respire primarily through the skin. These species require moist environments for respiration. Third, amphibians and reptiles move relatively short distances (Bury and others 2000) such that they must deal with the changed conditions of their landscape and generally cannot escape or relocate to more favorable conditions (Szaro 1988). Amphibians and reptiles typically have small home ranges. For instance, eastern red-backed salamanders remain within an area of about 15.5 square yards (Kleeberger and Werner 1983) of the forest floor within their lifetime; by contrast, the little brown skink (*Scincella lateralis*) has a home range of 62 square yards (Brooks 1967).

Fire within a forest can alter the environment for amphibians and reptiles in several ways. Forest floor litter and coarse woody debris, which are used by amphibians

[1]Missouri Department of Conservation, 1110 S. College Ave., Columbia, MO 65201, 573-882-9909 ext. 3242, email: Rochelle.Renken@mdc.mo.gov.

as a moist environment (Kleeberger and Werner 1983; Herbeck and Semlitsch 2000) and by reptiles as cover and perches (James and M'Closkey 2003), could be eliminated (Bury and others 2000; Pilliod and others 2003; Vandermast and others 2004). Changes in canopy cover and ground level vegetation could result in a drier environment on the forest floor (Pilliod and others 2003). The reduction or elimination of trees around temporary or permanent pools and streams could change the water temperature and chemistry, the amount of time water remains in pools, and the pattern of stream flow, thus changing the breeding environment for amphibians (Gresswell 1999; Skelly and others 1999; Pilliod and others 2003).

In the section that follows I summarize current information with respect to the effects of fire on forest amphibians and reptiles in oak forests of the Eastern and Central United States, and suggest issues that forest managers might to consider before using fire to obtain desired forest conditions.

CURRENT KNOWLEDGE ABOUT FIRE EFFECTS ON AMPHIBIANS AND REPTILES

Research on the effects of fire on amphibians and reptiles within oak forests of the Eastern and central United States has been limited, though studies to date indicate that fire results in little direct animal mortality, has little effect on amphibian abundance and diversity, and may result in increases in abundance for some reptile species. Palis (1995) in Indiana and Floyd and others (2002) in South Carolina did not find evidence of direct mortality of amphibians and reptiles due to fire. Direct mortality may not have occurred because amphibians tend to reside in moist environments, such as heavy litter and duff, that cannot sustain fire, or in underground refuges, such as small mammal tunnels (Gordon 1968), that would insulate them from fire (Russell and others 1999; Pilliod and others 2003). Reptiles also likely sought refuge in tunnels or under cover objects such as rocks, or climbed trees with the advance of fire. Although unburned sites tended to have more salamanders than sites that recently burned (the results between burned and unburned plots were not significantly different in statistical tests; Floyd and others 2002; Moseley and others 2003), investigators in Georgia (Moseley and others 2003), South Carolina (Floyd and others 2002), and Virginia (Keyser and others 2004) did not detect an effect of fire on overall amphibian abundance, diversity, or the number of species (Table 1). Nor did Ford and others (1999) observe an effect of fire on salamander numbers in North Carolina (Table 1). The only reported effect on amphibians was by Kirkland and others (1996), who observed more American toads (*Bufo americanus*) on burned than on unburned tracts in Pennsylvania (Table 1). These authors did not have an explanation for this difference.

As for reptiles, some investigators have reported a greater abundance and diversity of lizards following burning (Table 1; Moseley and others 2003; Keyser and others 2004). These authors believed that the reduced litter, increased amount of bare ground, and removal of the midstory within burned forest resulted in more advantageous thermoregulatory conditions for lizards. However, others did not detect an impact of burning on lizard, snake, and turtle abundance and diversity, though lizards tended to be more abundant on burned plots (Table 1; Floyd and others 2002). Also, in preliminary analyses of more recent work, there were no effects of fire on amphibians in oak-hickory-poplar forests in South Carolina (K.R. Russell, University of Wisconsin-Stevens Point, pers. commun.) or on amphibian abundance (but perhaps on reptile abundance) in North Carolina oak-hickory forests (C. Greenberg, USDA Forest Service, pers. commun.). The season of burning also had no effect on amphibians and reptiles. Although spring and summer burns resulted in less forest shrub cover than winter burns, winter, spring, and summer burns did not differ in their effect on amphibian and reptile abundance (Keyser and others 2004). No investigators reported local disappearances of species or the appearance of new species on study plots following burning (Kirkland and others 1996; Ford and others 1999; Floyd and others 2002; Moseley and others 2003).

In nearly all of the studies reporting the effects of fire on amphibians and reptiles in eastern oak forests, few discussed the intensity and type of fire (Table 1). The fires ranged from intense wildfires (Kirkland and others 1996) and light prescribed burns (Floyd and others

Table 1.—Details of fires and amphibian and reptile responses to those fires in the eastern oak forests of the US.

Location and reference	Type of fire treatments	Fire intensity	Fuel consumption	Patchiness of consumption	Species evaluated	Animal r80esponse
Indiana (Palis 1995)	April burn; Prescribed burn set at 1615 hrs	Nighttime fire	NA[a]	80 percent of 1,000 acres burned	Amphibians and reptiles of that region	No direct mortality observed; resulting amphibian and reptile numbers and assemblage appeared typical
Pennsylvania (Kirkland and others 1996)	Burned in November	Charred tree trunks; killed many saplings; charred vegetation in ground and shrub layers	NA	NA	American toad	More toads on burned than unburned sites
North Carolina (Ford and others 1999)	Burned in April	High intensity prescribed fire	NA	NA	Woodland salamanders	No difference in salamander abundance detected between burned and unburned sites
South Carolina Piedmont (Floyd and others 2002)	First round of fires in February - March 1999; strip head fires set 15 to 30 feet apart; second round of fires in April 2000; strip head fires set 10 to 20 feet apart;	First round average flame length was >1 foot; second round average flame length < 1 foot	NA	NA	Total herpetofauna, frogs, toads, salamanders, newts, lizards, turtles, snakes	No difference in abundance, diversity, and eveness in total herpetofauna or by taxonomic groups

Table 1.—continued.

Georgia upper Coastal Plain (Moseley and others 2003)	Winter burns; stands had been burned every 2-3 years in the previous 9 years	Low intensity	More litter in unburned; more bare ground in burned; no difference in coarse woody debris volume between burned and unburned	NA	Total herpetofauna; amphibians, anurans, salamanders, reptiles	No difference in overall amphibian and reptile, salamander, or anuran abundance, richness, and diversity between burned and unburned; reptile abundance and diversity were greater in burned than unburned sites
Virginia Piedmont (Keyser and others 2004)	Seasonal prescribed burns (winter, spring, summer) applied to first-stage shelterwood-harvested stands (to 48 ft^2 basal area per acre) cut 3 to 5 years prior to burns	Average fire temperature 39 inches above the ground was 526 °F in winter burn, 648 °F in spring burn, and 485 °F in summer burn	All fires reduced fine, medium, coarse fuel loads (fine fuels were eliminated; coarse fuels were partially reduced), yet duff layer remained intact	Spring and summer burns caused a shift from shrub dominated to herbaceous dominated understory	All amphibians and reptiles, especially eastern red-backed salamanders, American toads, northern fence lizards (*Sceloporus undulatus*), little brown skinks, and southeastern five-lined skinks (*Eumeces inexpectatus*)	No difference in abundance of salamanders, toads, and all amphibians between controls and different seasonal burns; lizard abundance was greater in burned than unburned stands

a Not Available

Figure 1.—Results of a prescribed burn where not all the litter and woody debris was eliminated.

2002; Moseley and others 2003) to high-intensity prescribed burns (Ford and others 1999; Keyser and others 2004). The effects of fire on amphibians and reptiles generally were similar among these studies.

Researchers may not have detected an effect of fire on amphibians and many reptiles due to short-term changes in the physical features important to these animals or the effects were not significant. Unless conditions are extremely dry, some coarse woody debris remains after a fire (Palis 1995; Van Lear and Harlow 2000; Moseley and others 2003; Trammell and others 2004) and can serve as cover for both amphibians and reptiles. Fire causes a temporary reduction in ground flora cover and litter depending on the frequency and intensity of fire, but ground flora cover rebounds quickly, sometimes within the next growing season (Kirkland and others 1996; Vandermast and others 2004). Not all litter or duff is consumed (Fig. 1; Kirkland and others 1996; Floyd and others 2002; Keyser and others 2004; Trammell and others 2004) and burns may create a mosaic of vegetation on the forest floor (Ford and others 1999). Litter seems to accumulate to unburned plot levels within 3 years after fire (Gagan 2002). Also, unless the objective of burning is to dramatically reduce the density of trees and greatly open the canopy, trees remaining following infrequent fires provide shade to reduce the amount of temperature variation at the forest floor, and the canopy helps soil and any remaining litter retain moisture (Ford and others 1999). Moisture within soil, litter, and coarse woody debris allows salamanders to remain active at the forest floor surface. In dry environments, amphibians, particularly salamanders, retreat underground and remain there until conditions on the forest floor are more suitable (Heatwole 1960; Heatwole 1962; Jaeger 1980; Feder 1983). For lizard species that experience an increase in abundance with fire, the temporary benefits of more bare ground and more basking sites probably boost survival rates (Parker 1994). However, populations probably return to pre-burn levels as post-fire forest conditions return to conditions of unburned forest.

OTHER CONSIDERATIONS CONCERNING THE EFFECTS OF FIRE

Do we need to be aware of other considerations concerning the use of fire as a tool and its effect on amphibians and reptiles? One is the possible indirect effects of fire on streams and stream-associated amphibians. There have been no studies on the indirect effects of fire on streams in eastern oak forests, but it is possible that streams could be affected by increased rates of sedimentation, increases in temperature due to reduced canopy cover, increases in amount of woody debris into the stream, which could change the flow and course of the stream, and increased concentrations of chemicals from the leaching of ash and the diffusion of smoke (Gresswell 1999; Pilliod and others 2003). Increased rates of sedimentation, increased concentrations of chemicals such as potassium and nitrogen, and increases in woody debris may change stream productivity and habitats, thus changing the flow and quality of streams. Stream-associated amphibians may be sensitive to such changes (Bury and others 2000; Pilliod and others 2003), which also can affect the foods and reproduction of stream-side dwelling amphibians, such as the northern dusky salamander (*Demognathus fuscus*). A reduced canopy cover might result in increased water temperature, which also could affect foods (e.g., macroinvertebrates) and reproduction of these animals (Pilliod and others 2003).

Another consideration is the direct effect of fire on vegetation structure around amphibian breeding ponds. Frequent fires open the canopy around pools and the increased light can change amphibian use of the pools. Depending on how isolated the pool is from other pools and the amount of time water is in the basin in a year, amphibian species more characteristic of open pools may start using formerly closed-canopy pools. Amphibian species such as spring peepers (*Pseudacris crucifer*) prefer open habitats and more open canopies around breeding pools (Skelly and others 1999; Halverson and others 2003), and may invade formerly closed-canopy pools if other spring peeper populations are nearby. Pools in closed canopies typically have lower oxygen levels and water temperatures, and shorter periods when water is in the basin than open-canopy pools of the same category (Skelly and others 1999, 2002; Pilliod and others 2003). The reason for this is that trees shade the water and use the water within the basin (Skelly and others 1999). Lower water temperatures reduce tadpole growth rates for some amphibian species (Skelly and others 2002; Halverson and others 2003). Reduced growth rates and faster drying pools in closed canopies make it difficult for tadpoles to develop into terrestrial-living juveniles before the pools dry. In a study examining amphibians using ponds in Michigan between 1988 and 1992, Skelly and others (1999) noted that tree growth and maturation around pools that were open-canopy pools from 1967 to 1974 resulted in fewer amphibian species using the pools. The composition of the amphibian community was less diverse and species that preferred more open habitats stopped using the closed-canopy pools. The remaining amphibians were considered forest species and generally tolerated of a wider range of canopy conditions than open-habitat species.

A third consideration is the use of fire for forest management in the presence of endangered or threatened species and/or those of special concern. Although direct mortality does not seem to be an issue, managers may want to be aware of these species' habitat needs and seasonal and daily activity patterns so that fire does not result in direct mortality (e.g., burning outside the months of the year when that species is active), and that required habitat remains or is improved following burning.

A fourth consideration is that while current information suggests that infrequent fires in eastern oak forests result in little direct mortality or in insignificant impacts to amphibians and limited changes to reptile communities, future data may point to impacts that have not yet been measured. I found few published studies on the effects of fire on amphibians and reptiles in eastern oak forests, yet I am aware that several other studies that are in progress or that recently were completed (C. Greenberg, USDA Forest Service, pers. commun.; D. Miles, Ohio University, pers. commun. part of the National Fire and Fire Surrogate Study; K. Russell, University of Wisconsin-Stevens Point, pers. commun.).

CONCLUSIONS

Currently available information indicates that fire has little effect on amphibians and reptiles in oak forests of the Eastern and Central United States. Some researchers have noted fewer salamanders in burned areas than unburned areas, though significant changes in amphibian abundance, diversity, and number of species have not been detected. Researchers have also reported that fire generally does not affect reptile abundance and diversity but that there is evidence that lizard abundance increases following burning. The lack of significant effects likely is due in part to the incomplete elimination of moisture-holding litter, duff, and coarse woody debris, the continued canopy cover provided by remaining trees, and the quick regrowth of ground vegetation (Palis 1995; Moseley and others 2003; Keyser and others 2004).

Forest managers must consider many factors when contemplating management goals on properties under their care. In an ideal situation, a manager would monitor the impacts of management actions on amphibians and reptiles along with such monitoring as necessary to determine whether the desired forest conditions are obtained. Few managers have the luxury of time and money to monitor amphibian and reptile communities. The next best option is to work with a biologist who can monitor amphibian and reptile responses or provide information to help guide decision making. If this option is not available, the manager can consult general references related to the biology and habitat needs of nontarget animals. Examples include "A field guide to reptiles and amphibians of eastern and central North America" (Conant and Collins 1991), "A guide to amphibians and reptiles" (Tyning 1990), and "Salamanders of the United States and Canada" (Petranka 1998). "Amphibian declines: the conservation status of United States species" (Lannoo 2005) provides information on the life history, natural history, and conservation status of all amphibians in the United States. Other helpful references include state or regional publications such as "The amphibians and reptiles of Missouri" (Johnson 2000)

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CONSERVATION IMPLICATIONS FOR NEOTROPICAL MIGRATORY AND GAME BIRDS IN OAK-HARDWOOD STANDS MANAGED WITH SHELTERWOOD HARVESTS AND PRESCRIBED FIRE

J. Drew Lanham[1†], Patrick H. Brose[2] and Patrick D. Keyser[3]

Abstract.—Prescribed fire in conjunction with shelterwood cutting is a novel way to regenerate oak-dominated stands on certain upland sites while minimizing the intrusion of hardwoods. We describe three options (complete or partial canopy retention, postharvest prescribed burning, and complete canopy removal) within a shelterwood-prescribed fire regime that will create two-age stands that are likely to harbor a diverse mixture of mature forest and early successional birds; parklike woodlands with open woodland species; or early successional habitats with shrubland species. This system is a viable option for managing both avian and timber resources where oak-dominated stands on upland sites are the desired goal.

INTRODUCTION

In Eastern North America, shelterwood silviculture is a common technique for managing oak-dominated stands on upland sites (Sander et al. 1983). Partial harvests reduce the dense shade that suppresses root development of existing oak regeneration (Loftis 1990) and helps retard the rapid height growth of less-desirable species such as yellow poplar (*Liriodendron tulipifera* L.*)* and red maple (*Acer rubrum* L.). Additionally, soil disturbance from the harvesting operations prepares seedbeds for acorns produced by the retention stand, thereby encouraging oak seedling establishment (Cook et al. 1998).

On some upland sites, preharvest treatments such as herbicide application (Loftis 1990; Lorimer et al. 1994), low-intensity burning (Barnes and Van Lear 1998; McGill et al. 1999), tree sheltering (Potter 1988), and implantation of high-quality nursery stock (Bowersox 1993; Gordon et al. 1995; Schlarbaum et al. 1997) may precede the initial shelterwood harvest to encourage oak regeneration. However, these preharvest treatments are expensive and often ineffective and must precede the initial shelterwood cut by 5 to 15 years to allow sufficient root development of the oak regeneration.

The constraints of time and money that often affect private owners of small- to medium acreage require a more efficient means of regenerating oaks in shelterwood systems. Keyser et al. (1996) burned two oak-dominated shelterwood stands after an initial harvest and found that regeneration of yellow-poplar, red maple, and sweetgum (*Liquidambar styraciflua* L.) was reduced by as much as 90 percent while oak reproduction was reduced only by 11 percent. Followup investigations of fire effects in oak-dominated shelterwood stands reported similar results in high fire resistance among oak and low tolerance for burning among less desirable hardwood competitors. This illustrates the vital role of postharvest burning coupled with growing-season burns in creating an oak-dominated seedling cohort (Brose and Van Lear 1998a,b; Brose et al. 1999; Brose and Van Lear 1999; Van Lear and Brose 1999).

We suggest that this technique may be applicable elsewhere in upland oak-dominated stands (oak-hickory) of Eastern North America (Ward and Gluck 1999). Application of this technique (Fig. 1a-h) entails three-steps. First, an initial harvest leaves 50 to 60 dominant oaks per ha (11 to 12 m^2 of basal area/ha). The remnant stand should contain the best oak stock to encourage a vigorous regeneration cohort. The stand is then left undisturbed for 3 to 5 years to allow the development of a regeneration layer. After 3 to 5 years, a hot (flame length > 1.0 m) growing-season fire is applied to the stand, resulting in an oak-dominated regeneration cohort.

Although this technique was originally implemented to improve the viability of oak regeneration and the

[1]Clemson University, Department of Forestry and Natural Resources, 261 Lehotsky Hall, Clemson, SC 29634; [2]USDA Forest Service, Northern Research Station, Irvine PA 16329; [3]Forestry Division, MeadWestvaco Corporation, P.O. Box 577, Rupert, WV 25984; [†]Corresponding author, 864-656-7294, email: lanhamj@clemson.edu.

production of hardwood timber in uplands, the conservation of biodiversity also is a goal of many forest management activities. Birds comprise significant ecological components of eastern forest systems. Songbird population declines in many forests of the Eastern United States have focused attention on the effects of various silvicultural practices on conservation efforts. Such efforts have been particularly focused on many species of neotropical migrants (Askins et al. 1990). Game birds have long been a focus of forest management because of their economic and recreational importance. While numerous investigators have reported the effects of various silvicultural treatments on game and nongame birds, there is a dearth of information addressing the effects (real or potential) of prescribed fire on avifauna in hardwood systems. We discuss how various management options implemented in oak- shelterwood burn systems can influence the composition of avian communities. These potential influences are related to the vegetative structure and composition that result from the three oak-shelterwood burn options *and* from inferences drawn from other bird habitat studies conducted in other hardwood or pine- hardwood systems where silvicultural treatments have created habitat conditions similar to those expected in oak-shelterwood burns. To emphasize the potential importance of these treatments from a bird conservation perspective, we have used Bird Conservation Region (BCR) breeding scores (RCS-b) from Region 24 – Central Hardwoods, Region 28 – Appalachian Mountains, and Region 29 – Piedmont (Partners in Flight Species Assessment database: www.rmbo.org/pif/scores/scores.html) as indicators of management priority.

Conservation Implications for Avian Communities

Canopy Retention

Canopy retention treatments provide two-aged stands twice during the shelterwood burn regime: 1) when "undesirable" hardwoods dominate the advance-regeneration cohort, and 2) after a satisfactory cohort of vigorous, advance oak- regeneration is achieved when a portion or all of the residual overstory trees may be retained for at least half of the next rotation.

A widely observed trend of bird -habitat relationship studies is the positive association between floristic structural diversity and bird-species diversity. Retention of some overstory during shelterwood treatments may provide sufficient canopy habitat and vertical structure for some mature forest canopy bird species to use partially harvested stands (Dickson et al. 1995). Relative to the composition of a mature forest songbird community, canopy retention treatments would be the least intensive and probably most similar to an uneven-aged mature forest. Dickson et al. (1995) concluded that the retention of a residual canopy (less than 50 percent) for several years after an initial harvest can provide habitats for some mature forest birds that otherwise would not inhabit stands managed using traditional even-age management techniques.

Wood and Nichols (1995) found that two-aged stands in West Virginia contained a greater density, richness, evenness, and overall diversity of breeding birds than early successional or mature stands. The total density for all neotropical migrants also was highest in the two-aged stands. Densities of forest-interior species did not differ statistically among clearcut, mature, and two-aged stands. The two-aged stands had densities of interior-edge species equal to or greater than the other two treatments. The co-occurrence of mature forest and early successional species within the same areas indicated that two-aged stands provided habitats for both mature and early successional species. Mature forest species reported in The Wood and Nichols study included veery (scientific names for all neotropical migrant and game bird species are in Table 1), American redstart, and scarlet tanager. Early successional species recorded in the same stands included chestnut-sided warbler, indigo bunting, and eastern towhee.

In the Missouri Ozarks, Annard and Thompson (1997) reported higher species richness for breeding birds in stands treated by the shelterwood method than in clearcuts, group selection, single-tree selection, or uncut stands. Species richness was higher in shelterwoods than in unharvested or uneven-aged stands. As in the Wood and Nichols study, these differences were attributed to the presence of both early seral stage and mature forest

Table 1.—Oak shelterwood burn occupancy potential of select neotropical migrant and game bird species[a]

Species	Option 1	Option 2	Option 3
wild turkey (*Meleagris gallopavo*)	X	X	X
northern bobwhite (*Colinus virginianus*)		X	X
ruffed grouse (*Bonassa umbellus*)	X	X	X
yellow-billed cuckoo (*Coccyzus americanus*)	X		
black-billed cuckoo (*C. erythropthalmus*)		X	
whip-poor-will (*Caprimulgus vociferus*)			X
Chuck-will's-widow (*C. carolinensis*)	X		
ruby-thr. hummingbird (*Archilocus colubris*)	X		X
Acadian flycatcher (*Empidonax virescens*)	X		
least flycatcher (*E. minimus*)		X	
willow flycatcher (*E.alnorum*)			
eastern kingbird (*Tyrannus tyrannus*)			X
great-crested flycatcher (*Myiarchus crinitus*)		X	
eastern wood-pewee (*Contupus virens*)		X	
gray catbird (*Dumatella carolinensis*)	X		X
wood thrush (*Hylocichla mustelina*)	X		
veery (*Catharus fuscescens*)	X		
blue-gray gnatcatcher (*Pilioptila caerulea*)	X		
red-eyed vireo (*Vireo olivaceous*)	X		
yellow-throated vireo (*V. flavifrons*		X	
warbling vireo (*V. gilvus*)		X	
white-eyed vireo (*Vireo griseus*)	X		
Blackburnian warbler (*Dendroica fusca*)	X		
black-throated blue warbler (*D. caerulescens*)	X		
black-throated green warbler (*D. virens*)		X	
cerulean warbler (*D. cerulea*)	X	X	
chestnut-sided warbler (*D. pensylvanica*)		X	X
yellow-throated warbler (*D. dominica*)			X
prairie warbler (*D. discolor*)			X
yellow-warbler (*D.petichia*)			X
blue-winged warbler (*Vermivora pinus*)	X		X
golden-winged warbler (*V. chrysoptera*)	X		X
American redstart (*Setophaga ruticilla*)	X		
black-and-white warbler (*Mniotilta varia*)	X	X	
common yellowthroat (*Geothlypis trichas*)			X
hooded warbler (*Wilsonia citrina*)	X		
Kentucky warbler (*Oporonis formosus*)	X		
northern parula (*Parula americana*)	X		
ovenbird (*Seiurus aurocapillus*)	X		

Continued

Table 1.—Continued.

Species	Option 1	Option 2	Option 3
Louisiana waterthrush (*S. motacilla*)	X		
worm-eating warbler (*Helmitheros vermivorus*)	X		
yellow-breasted chat (*Icteria virens*)			X
orchard oriole (*Icterus spurius*)	X	X	
Baltimore oriole (*I. galbula*)	X	X	
scarlet tanager (*Piranga olivacea)*	X		
summer tanager (*P. rubra*)		X	
indigo bunting (*Passerina cyanea*)	X	X	X
blue grosbeak (*Guiraca caerulea*)		X	X

[a]Bird habitat associations are inferred from associations in similar types and conditions of habitat that would be created by the options described in this paper. Bird habitat associations are derived from Hamel et al. (1982) and denote associations with seral stages in oak-hickory habitats as follows: Option 1 = sapling poletimber-sawtimber; Option 2 = grass-forb, sawtimber; Option 3 = seedling-sapling.

birds, including blue-winged warbler, prairie warbler, red-eyed vireo, worm-eating warbler, and Acadian flycatcher.

Nesting success must be considered in conjunction with measures of density and diversity of breeding birds. Wood and Nichols (1995) reported no differences in nest success among treatments in West Virginia. Nest parasitism by brown-headed cowbirds (*Molothrus ater*) was not a major factor in their study; only 8 of 246 nests were parasitized and there were no differences in the number of cowbirds among treatments. Annard and Thompson (1997) and Welsh and Healy (1993) reported similar results in Missouri and New Hampshire, respectively. However, one must remain aware, that patterns of predation and parasitism may vary depending on the landscape context. Overall, the impact of cowbirds and predators in extensively forested systems tends to be lower than that in agricultural and suburban landscapes (Wilcove 1985)

The findings of all of these studies are consistent with an earlier study by Crawford et al. (1981). Although they did not study shelterwood systems, the findings of Crawford et al. nonetheless are relevant because they concluded that timber management strategies altered bird communities in relationship to the degree of stand disturbance. They predicted that partial harvests would provide sufficient canopy cover to buffer complete species turnover from mature forest to early successional species observed in clearcut forests. They further surmised that partial cuts would return more quickly to site conditions conducive to mature bird species than even-age treatments. These findings have been corroborated by other studies that have shown that although populations of some forest-interior songbirds may be reduced relative to an undisturbed stand due to habitat alteration, increased nest predation, and parasitism (Webb et al. 1977; Wood and Nichols 1995), these species generally are not eliminated entirely and population recovery may occur rapidly as the new forest matures (Conner and Adkisson 1975; Askins and Philbrick 1987).

Canopy disturbance may benefit forest-interior bird species that have declined in some regions. Some birds that use early-successional gaps within mature forests may decline in areas where disturbances do not produce the regenerating ground-layer and shrub vegetation they prefer (Franzreb and Rosenberg 1997). Shelterwood harvesting would stimulate the growth of low vegetative cover. In West Virginia, Wood and Nichols (1995) found that Kentucky warblers, wood thrushes, American redstarts, and black-and-white warblers were 2 to 3 times more abundant in two-aged stands than in uncut controls. The abundance of these species in the shrubbier two-age stands lends credence to the shelterwood-shrub layer hypothesis.

The retention of the best, 11 to 12 m^2 of oak basal area/ha (50 to 60 dominant oaks/ha) in shelterwood stands also provides reliable acorn sources (Healy 1997). Acorns are one of the most important wildlife foods as they are eaten by more than 200 wildlife species throughout North America (Martin et al. 1951). Among these are a multitude of avian species (Martin et. al 1951; Beck 1993). Corvids, e.g., blue jays (*Cyanocitta cristata*) and American crows (*Corvus brachyrhynchos*), are voracious acorn predators and bear a large responsibility for the regeneration of oak stands through their caching activity. Although not a neotropical migrant or game species, red-headed woodpeckers (*Melanerpes erythrocephalus*) are short-distance migrants that are likely to benefit from overstory retention management activities. They also receive attention as species of regional importance from Partners in Flight (Panjabi et al. 2005). Several species of upland game birds including wild turkeys and ruffed grouse consume acorns (Martin et al. 1951; Dickson 2001), making canopy retention stands potentially important foraging habitat for both species.

On the basis of the floristic structure of stands expected after canopy retention treatments, Table 1 lists the diverse array of neotropical migratory and game bird species that are likely to occur in the diverse two-age structure of these areas. Conservation priority scores and management priorities for species occurring under this option are shown in Table 2. Note that the cerulean warbler, a species likely to benefit from the mature canopy retained in this silvicultural scenario (in mesic types), is characterized as a species with "Immediate Management" conservation priorities in both the Central Hardwoods and Appalachian Mountain Bird Conservation Regions BCR (Panjabi et al. 2005). Illustrating the diverse avian conservation potential for this scenario, golden-winged warblers and ruffed grouse also are Immediate Management priorities (Panjabi et al. 2005). In the Piedmont, conservation priorities are even higher as ruffed grouse are characterized as a species in need of "Critical Recovery". Ruffed grouse might occupy the complex woody/herbaceous understory resulting from the overstory retention option in the Central Hardwoods, Appalachian, and Piedmont BCR while golden-winged warblers should occupy similar habitats in the Appalachian mountains.

Understory Suppression Using Prescribed Burning

The second option in the oak-shelterwood burn scheme is the use of periodic prescribed fire in shelterwood stands. Among the three options discussed here, this method is likely to be intermediate in its effects on the songbird community. Ultimately, the shift in species composition will vary depending on the vegetative structure resulting from the season, intensity, and frequency of the prescribed fire. Dormant-season (winter) burns will produce low-growing, sprouting regeneration of shrub and trees and stimulate the production of soft mast (Stransky and Roese 1984). These responses may provide forage, cover and arthropod prey for many game and nongame birds (Dickson 1981, 2001).

Repeat dormant-season burning promotes an increased abundance of oak regeneration. Oak regeneration is limited with additional fires and released at intervals by withholding burning treatments, creating patchy stands in different successional stages. Dickson (1981) surmised that in southern pine and pine-hardwood forests, a patchwork of different successional stages within a stand (or across a landscape) could enhance bird diversity and abundance. This patchwork obviously would be dependent not only on the frequency and intensity of fires but also on the size, topography, and site capability of the area burned. In stands managed with dormant-season fires that allow the proliferation of hardwood shrubs and trees underneath an open canopy, bird communities are likely to consist of a large proportion of shrub nesting, e.g., white-eyed vireo, and midstory species, e.g., wood thrush, along with species more characteristic of open canopy forests, eg., yellow-billed cuckoo and blue-gray gnatcatcher. More so than other burning treatments, dormant-season fires in oak-shelterwoods are likely to produce bird communities more similar to two-aged canopy retention stands.

Annual or biennial prescribed burning during the growing season should create open hardwood woodlands

Table 2.—Conservation scores and management priorities[a] for selected avian species of regional importance potentially occupying oak-shelterwood burn habitats in central hardwoods (CH), Appalachian mountain (AM), and piedmont (PM) bird conservation regions.

Species	CH	AM	PM
northern bobwhite	16 MA	***15 IM***	***16 IM***
ruffed grouse	***15 IM***	***16 IM***	***14 CR***
yellow-billed cuckoo	15 MA		14 MA
black-billed cuckoo		17 MA	14 MA
whip-poor-will	17 MA	16 MA	18 MA
Chuck-will's-widow		14 MA	15 MA
Acadian flycatcher	16 PR	17 MA	
willow flycatcher	12 PR	11 PR	9 PR
eastern kingbird	15 MA		14 MA
eastern wood-pewee	15 MA	15 MA	15 MA
wood thrush	16 MA	16MA	16 MA
blue-gray gnatcatcher	14 MA		
yellow-throated vireo	16 PR	17 MA	14 PR
white-eyed vireo	15 MA		
Blackburnian warbler		14 MA	
cerulean warbler	***19 IM***	***21 IM***	16 MA
prairie warbler	18 MA	18 MA	18 MA
yellow-throated warbler	15 PR	16 PR	
blue-winged warbler	19 MA	17 PR	16 MA
golden-winged warbler		***21 IM***	
black-and-white warbler		16 MA	
hooded warbler		15 PR	
Kentucky warbler	18 MA	19 MA	15 PR
Louisiana waterthrush	15 PR	18 MA	
worm-eating warbler	18 MA	18 MA	13 PR
yellow-breasted chat	16 MA	15 MA	
orchard oriole	17 MA		
Baltimore oriole			14 MA
scarlet tanager		14 PR	
summer tanager	6 PR	16 MA	
indigo bunting	14 PR	14 PR	12 PR
blue grosbeak			14 PR
Average conservation score	18.22	16.24	14.6
Critical recovery	0	0	1
Immediate management	2	4	1

[a]Management actions in decreasing priority: CR= Critical Recovery; IM = Immediate Management; MA = Management Attention; PR = Planning and Responsibility (Panjabi et al. 2005)

and savannas by gradually eliminating much hardwood shrub and tree regeneration while stimulating production of ground-level herbaceous vegetation (Thor and Nichols 1973). Oak woodland and savanna habitats were described as common landscape features by early explorers and settlers who observed the extensive use of fire by indigenous Americans (Pyne 1982; Buckner 1983; Van Lear and Waldrop 1989). However, over time oak savannas and woodlands and some of the wildlife species associated with them have become rare. The restoration of hardwood savannas and open woodlands probably would shift bird guilds from mature forest-interior species to canopy and midstory dwelling, open woodland and grove species, e.g., great-crested flycatcher, eastern wood-pewee, orchard oriole and summer tanager. Here, even more so than in overstory retention stands, the red-headed woodpecker is likely to benefit from this management option. Such open habitats may also likely to attractive to wild turkey for a number of life requisites including foraging, resting, and brood rearing.

Although growing-season fires might benefit some bird species, others could be negatively impacted by burns initiated so late that nesting and other breeding activities are disrupted. Therefore, spring burning should be prescribed judiciously as early as possible in the season to minimize direct impacts on nesting or breeding birds.

Fire intensity (hot versus cool) also affects vegetative structure and, therefore, avian community composition. Studying the effects of fire intensity on bird communities in Alabama Piedmont pine-hardwoods, Stribling and Barron (1995) found a greater abundance and diversity of birds in less intensively burned stands with canopy, shrub, and cavity nesters the most abundant. Canopy, shrub, and bark-feeding species also were more abundant in cool burn sites than in untreated stands. These differences were attributed to the patchiness of the vegetative structure in these areas. This same study reported a higher abundance of ground-foraging and ground-nesting songbirds in stands burned with a hot, growing-season fire than in those burned by cooler growing-season fires. They surmised that the observed responses of terrestrially associated species could have occurred because of litter removal that may have provided better foraging and nesting habitat.

Some residual canopy trees such as maples (*Acer spp*.) and tulip-poplars and those with slash accumulations at their bases are susceptible to fire-kill or damage (Brose and Van Lear 1999a). These snags are important foraging sites for woodpeckers and other bark-gleaning species. Snags also provide perching/hawking sites and roosting/nesting habitats. Larger snags provide nesting habitats for both primary cavity excavators (woodpeckers) and secondary cavity nesters, including neotropical migrants such as the great-crested flycatcher (Lanham and Guynn 1996). In addition to the valuable function provided by snags, downed logs and other coarse woody debris, e.g., treetops, fallen limbs, provide habitat for forest floor-dwelling arthropods, herpetofauna, and small mammals (Hanula 1996; Loeb 1996, Whiles and Grubaugh 1996). These provide food resources for songbirds and larger bird species, e.g., raptors, wild turkey. Larger logs remaining on the forest floor also may provide drumming substrate for ruffed grouse (Dickson 2001). Because the effects of fire in forested stands can have such varied effects, a variety of bird species is possible based on fire frequency and intensity and various site characteristics. Because most natural resource mangers and private landowners are primarily concerned with the production of open, oak-dominated woodland, Table 1 lists species likely to occur in understory (growing-season) burned treatments that result in park-like, oak woodlands. Conservation priority scores for species occurring under this option are shown in Table 2. Of note within the context of an option producing woodland species, highest conservation scores and management priorities are listed for the cerulean warbler, which may benefit from the park-like conditions should they be implemented in mesic oak/hardwoods preferred by the species in the Central Hardwoods and Appalachian Mountain BCR. Ruffed Grouse retain an Immediate Management priority for this option in Central Hardwoods and Appalachian mountain habitats. In the Piedmont, this priority increases to Critical Recovery status

Overstory Removal

The third option of harvesting all of the residual overstory trees creates early successional hardwood habitats. These stands will undergo a dramatic turnover in avian composition with species such as indigo buntings and field sparrows occurring in regenerating hardwood stands during the grass-forb and seedling-sapling stages (Evans 1978). In subsequent years, as vertical structure within a regenerating stand changes with the growth of shrubs and saplings, avian diversity and abundance may surpass those of mature stands (Conner and Adkisson 1975; Thompson and Fritzell 1990; Thompson et al. 1992). In many eastern uplands, regenerating seedling-sapling hardwood habitats are preferred by shrub-scrub species such as prairie warblers, yellow-breasted chats, and chestnut-sided warblers. At this juncture in succession, northern bobwhite quail may use woody or brushy hardwood habitats that are interspersed with other cover types, e.g., pine forests, fallow fields, as escape, resting, and roosting cover in the nonbreeding season (Dickson 2001). Early successional hardwood stands may be used by wild turkey for nesting and brood-rearing habitat (Speake et al.1975). Ruffed grouse are especially dependent on dense thickets of hardwoods for drumming and brood rearing. Even-age systems including shelterwood regimes are cited by Dickson (2001) as a preferred method for creating suitable habitat for the species.

As regenerating stands mature to form closed canopy sapling-poletimber stands, species richness and abundance frequently decrease to levels below younger shrubland and older mature forest habitats (Conner and Adkisson 1975). However, some forest-interior songbirds such as black-and-white warblers and wood thrushes will begin using pole stands at this stage (Conner and Adkisson 1975; Mauer et al. 1981; Askins and Philbrick 1987). These stands continue to hold high suitability for ruffed grouse. (Gullion 1984). Table 1 lists bird species typical of regenerating, early successional hardwood stands. Conservation priority scores for species occurring under this option are shown in Table 2. Given the early successional conditions that are reproduced in this option, highest conservation priorities are given to the ruffed grouse (Central Hardwoods, Appalachian Mountains, Piedmont), northern bobwhite (Appalachian Mountains, Piedmont), and golden winged warbler (Appalachian Mountains).

DISCUSSION

The hardwood forests of Eastern North America are one of the largest, broad-leaved, deciduous ecosystems in the world (Hicks 1997). Numerous wildlife species, including a diverse assemblage of birds, inhabit oak-dominated forests. The songbird communities that depend on oak-dominated forests in the Southeast include a large number of neotropical migrants (Hamel et al. 1982; Thompson and Fritzell 1990). Since many of these species are declining, the management of their habitats has become a conservation priority. Although the prevailing songbird conservation paradigm in many eastern hardwood-dominated forests (especially large blocks of public forests) has been to limit harvests to single/group-tree selection or eliminate it entirely, thousands of acres of oak-dominated forests occur on private lands where wildlife conservation goals may be secondary to timber management priorities.

Game bird species also use oak stands in various stages of succession. The cultural and economic importance of maintaining harvestable populations of these species is a priority for many state agencies as well as private landowners. Prescribed fire in oak-shelterwood systems, when properly implemented, can provide the habitat diversity across the landscape that fosters healthy populations of game and nongame birds. The novel techniques suggested here provide an innovative management option that can satisfy the multiple goals of avian conservation and sustainable quality hardwood timber production.

Burning as a silvicultural technique in many forests traditionally has been associated with pine production. Conversely, it has been regarded as a force to keep out of hardwood forest management. The technique described here is an effective method for regenerating oak-dominated stands in the Southeastern Piedmont (Van Lear and Brose 1999). The steps involved in the process (partial harvest, burning, complete harvest) will create two-aged, open woodland, or shrubland habitats. Therefore, stands managed in different stages of the shelterwood-burn process across a landscape would offer habitats to forest-interior, edge-interior, open-woodland, and early-successional shrubland species. We believe that this technique has application beyond the Southeast to many other eastern forest uplands where oak is the desired dominant or codominant. The successful application of oak-shelterwood burn techniques may be especially critical in areas such as the Southern Appalachian Mountains where a number of species require critical recovery or immediate management efforts.

CAVEATS AND CONCLUSIONS

Where landscape context and logistics allow, managing upland oak-hardwood stands using some of the options outlined here can potentially produce a variety of habitats that favor various game and nongame birds while effectively producing quality hardwood timber. Management and conservation priorities for birds will vary regionally. The silvicultural regime suggested here offers alternatives that can accommodate a variety of different regional and management contexts.

We have offered baseline information on the conservation scoring and management priorities for neotropical migrants and gamebirds using Partners in Flight rankings and our best inference as to the habitat occupancy of the silvicultural options described herein. While these scores provide a basis for understanding the relative importance of a particular management option for various species and suites of species, they should not be used as stand-alone prescriptions for management. Researchers and natural resource managers must take regional and site specific contexts into account. We suggest that the use of conservation value indexing (Twedt 2004; Nuttle et al 2003; Bryce et al. 2002), which incorporates abundance estimates and other demographic factors, may provide a more quantitatively rigorous assessment of habitat "value" that can be used to build robust scenarios for planning and management.

While wildlife and timber production goals frequently are in opposition, the ability to reliably reproduce oak-dominated stands using a less intensive form of even-age management like the technique described here and associated options might be an effective tool for both wildlife conservation and sustainable timber production. As the demand for quality hardwood products increases, compromises regarding timber management and wildlife conservation in upland hardwood forests must be reached. While the oak-shelterwood burn system offers novel solutions for managing avifauna, the manager must always remember that he or she cannot manage every acre for every species. Rather, a knowledge of the landscape context and specific management objectives, supported by an adaptive management plan, offers opportunities for both the sustainable management of quality timber and diverse avian communities in eastern oak-dominated upland forests.

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INFLUENCE OF FIRE ON MAMMALS IN EASTERN OAK FORESTS

Patrick D. Keyser[1†] and W. Mark Ford[2]

Abstract.—With the exception of small mammals, little research has been conducted in eastern oak forests on the influence of fire on mammals. Several studies have documented little or no change in relative abundance or community measures for non-volant small mammals in eastern oak (*Quercus* spp.) forests following fires despite reductions in leaf litter, small woody debris, and changes in understory and midstory composition and structure. Other studies have documented short-term improvements in quantity and quality of forage for white-tailed deer (*Odocoileus virginianus*), but none has demonstrated changes in population parameters. For other mammal species there are no explicit studies and influences must be based on inferences from other research. We suggest that species that prefer partially open canopies, herbaceous understories, reduced midstories, or savannah habitats likely will prosper in the presence of fire. Fire has the potential to both recruit as well as eliminate den sites and cavity trees; burning regimes and fire intensity likely will determine outcomes. Species of high conservation concern, such as the Indiana bat (*Myotis sodalis*) and some subspecies of fox squirrels (*Sciurus niger*), should benefit from increased fire in these landscapes. An insidious consequence of continued preclusion of fire in oak systems is the loss of structure and the change in vegetative species composition to more fire-intolerant species.

INTRODUCTION

Fire has long been recognized as an important ecological driver and disturbance agent that shapes and maintains oak-dominated forest communities of eastern North America (Delcourt and Delcourt 1998; Frost 1998; Van Lear and Harlow 2002). Unlike other hardwood forest types, such as northern hardwoods and bottomland hardwoods, many oak-dominated types have relatively short fire return intervals (4 to 30 years) over most of the region (Frost 1998). A number of researchers have focused on concerns specifically related to the sustainability of oak forests (Clark 1993; Kellison 1993; Keyser et al. 1996; Brose et al. 1999; Brose et al. 2001), including those of the central hardwoods region (Van Lear et al. 2000; Sutherland and Hutchinson 2002).

Despite a number of studies on the effects of fire on wildlife in the Coastal Plain of the Southeastern United States, little work has been done on this subject in oak-dominated forests (Brennan et al. 1998; Lanham et al. 2002; Keyser et al. 2004), particularly as it relates to mammals (Keyser et al. 2001). Only several experiments have examined the response of non-volant small mammals, typically relative abundance and species richness at the stand level or smaller, to fire in the Eastern United States (Masters et al. 1998; Ford et al. 1999; Keyser et al. 2001; Kilpatrick et al. 2004). Incidental impacts of fire have been reported as well (Kirkland et al. 1996). Studies of economically important game species such as white-tailed deer (*Odocoileus virginianus*) have largely been confined to assessment of forage quality, availability, and use (Hallisey and Wood 1976; Wentworth 1986; Masters et al. 1993). Despite their high conservation status, fire impacts on nongame groups such as bats or charismatic mega-fauna such as black bears (*Ursus americanus*) have been examined only in conceptual, hypothetical approaches (Hamilton 1981; Carter et al. 2002). In this paper we review the literature relevant to fire effects on mammals in eastern oak forests. Our purpose is to draw meaningful inferences for species and species groups based on documented effects of prescribed fire or wildfire on habitat structure, and to discuss potential research directions to address current knowledge gaps. Fire or the lack thereof will continue to influence eastern oak forests and, in turn, the wildlife communities that depend on those forests for habitat (Kellison 1993; Brennan et al 1998). Accordingly, it is incumbent upon the natural resources community to work toward managing those outcomes to the benefit of sustainable wildlife populations.

[1]Forestry Division, MeadWestvaco Corporation, P.O. Box 577, Rupert, WV 25984; [2]USDA Forest Service, Northern Research Station, Box 404, Parsons, WV 26287; [†]Corresponding author, 304-392-1606, email: pdk2@meadwestvaco.com.

NON-VOLANT SMALL MAMMALS

Despite the paucity of data on fire impacts on most mammals in eastern oak forests of North America, non-volant small mammals have been perhaps the best studied due to their abundance, diverse assemblages, and ease of capture. In coniferous forests of Western North America and along the mixedwood-boreal transition zone where natural stand-replacing fires and intense site-preparation burns are more common, impacts of fire on non-volant small mammals can be significant via direct mortality (Tevis 1956) and through habitat modifications that alter and shape species assemblages (Crete et al. 1995; Sullivan and Boateng 1996; Monroe et al. 2004). Nonetheless, Crete et al. (1995) noted that a mosaic of stand seres created by landscape-level fires added to overall wildlife diversity, including that of non-volant small mammals.

In eastern oak forests, both natural and prescribed fires typically are of relative low intensity, are rarely stand replacing (Komarek 1974), and display site variability (Ford et al. 1999) at scales matching inherent variability of small mammal populations (Bowman et al. 2000). Consequently, negative impacts are not suspected in most cases. Orrock et al. (2000) noted that in some Appalachian habitats, small mammal species such as the southern red-backed vole (*Clethrionomys gapperi*) are specialized for mesic forest conditions and could be more fragmented and isolated by disturbance events. Results from both prescribed burning in the southern Appalachians of North Carolina (Ford et al. 1999) and a wildfire in the central Appalachians of Pennsylvania (Kirkland et al. 1996) suggest that small mammal abundance and species assemblages are not changed substantially by fire. Ford et al. (1999) noted that the abundance and assemblage patterns that existed prior to spring-burning persisted following burning with mesic-forest adapted species such as southern red-backed voles, smokey shrews (*Sorex fumeus*), and masked shrews (*S. cinereus*) more common in riparian areas and areas with abundant coarse woody debris in both burned and control stands. Moreover, data from Keyser et al. (2001) in central Virginia documented that the season of burn had little effect on non-volant small mammals with shrews (*Sorex longirostris, S. hoyi,* and *Blarina carolinensis*) and white-footed mice (*Peromyscus leucopus*) numbers not significantly different among oak shelterwood stands burned in the winter, spring, or summer, and unburned controls.

It is plausible that non-volant small mammals could be affected positively or negatively by fire in number of ways other than direct mortality. For example, on the Cumberland Plateau of Kentucky, prescribed fire in mixed mesophytic forests reduced the mass of soil invertebrates by approximately 35 percent (Kalisz and Powell 2000), which, in theory, could reduce available prey for Soricids. Conversely, Ahlgren (1966) observed that deer mice (*Peromyscus maniculatus*) responded positively to prescribed burning in Minnesota in part because fire consumption of the leaf litter and duff greatly increased seed exposure and, therefore, availability. Similar to forest floor structural changes observed in more intensive western fires, heavy reductions in leaf litter and downed coarse woody debris could negatively impact many non-volant small mammals in eastern oak forests. However, Kirkland et al. (1996) and Ford et al. (1999) observed virtually no change in coarse woody debris abundance following the low- to medium-intensity fires they studied. Additionally, substantial reductions in coarse woody debris following more intense fires in the region typically will be offset by overstory mortality and contributions to the future coarse woody debris pool (Regelbruggae and Smith 1994). Generally, though, data applicable to eastern oak forests are consistent with the findings of Kirkland et al. (1996) and Ford et al (1999) that ground-level vegetation change after one growing season following fire was less important than topo-edaphic factors of slope and aspect (Wentworth 1986, Hutchinson et al. 2005). Prescribed fire in eastern oak forests can be effective in reducing woody midstory stem density (Brose et al. 2001), creating structural conditions that do benefit both southern flying squirrels (*Glaucomys volans*; Bendel and Gates 1987) and fox squirrels (*Sciurus niger*; Edwards et al. 2003).

Although wholly unknown, behavioral changes in small mammals during or following fire might cause an increase in vulnerability to predation. Rowan et al. (2005) radio-tracked eastern chipmunks (*Tamias* *striatus*) following early spring fires in West Virginia on both burned and control plots. Despite fire intensities sufficient to consume most leaf litter, interrupt or modify the initial emergent patterns of herbaceous flora, and expose acorn caches, no difference was noted in mean home range size and habitat use between animals in the burned or control plots. Of the individuals captured and tagged within burned stands, none were observed leaving the burned areas or using adjacent unburned forest habitat that was in many cases less than 150 m away (Rowan et al. 2005). Home range overlap among conspecifics did not differ between burned and control plots, further suggesting minimal impact to eastern chipmunks.

BATS

Carter et al. (2002) provideed the most complete review of the relationship between fire and bats in eastern oak forests. Depending on the fire severity and fire susceptibility of tree species present, fire in eastern oak forest can both produce as well as consume standing snags that could be used as day-roosts by several bats species in the region (Starker 1934; Wendell and Smith 1986; Moorman et al. 1999a,b; Abrams 2000). The endangered Indiana bat (*Myotis sodalis*) and the northern bat (*M. septentrionalis*) predominately use snags as day-roosts from spring through early fall (Menzel et al. 2001; Owen et al. 2002). Whereas little brown bats do roost in snags (Fenton and Barclay 1980), that species' use is far less due in part to the availability of human structures as well as the overall change in forest condition, i.e., fewer large cavity trees and standing snags than in pre-settlement forests (Whitaker and Hamilton 1998). Although bats could be subject to direct mortality from the combustion of an occupied day-roost, especially one containing non-volant young in a maternity roost, most bats are capable of escaping a snag in advance of its consumption by fire. For bats that day-roost in foliage of live trees, e.g., hoary bats (*Lasiurus cinereus*) and eastern red bats (*L. borealis*), direct impacts probably result from temporary disturbance and displacement due to smoke (Rodrigue et al. 2001). Nonetheless, as nonhibernating migratory species, eastern red bats have been observed day-roosting in leaf and pine needle litter in upland pine-hardwood forests of South Carolina and Arkansas where they were disturbed as winter burning occurred (Saugey et al. 1998; Moorman et al. 1999a,b). It is possible that eastern red bats might be unable to flee advancing fire if in a state of torpor while ground-roosting.

Realistically, however, species that depend on snags probably benefit from a net addition of snags in burned stands in a region where it is believed that snag abundance is below that needed for many wildlife species (Carey 1983). For example, Wentworth (1986) noted that midstory tree mortality was pronounced on variable winter burns in the southern Appalachians, conditions that likely produce a net gain of day-roosts highly suitable for northern bats (Owen et al. 2002). Still, there are virtually no data on the net gain or loss or longevity and useful to bats of larger snags following fire in eastern oak forests. Lastly, Carter et al. (2002) noted that bats wintering in caves and mines might be subject to negative impacts from smoke in areas where air-flow into caves or mines occurs. However, few instances of this have been documented and none with recorded bat mortalities (D. Krusac, USDA Forest Service, pers. commun.).

Relative to all other mammalian taxa in the eastern oak forest, knowledge gaps in Indiana bat habitat management in conjunction with fire probably are the most critical to address. Maternity roost habitat of Indiana bats often is characterized by stands with numerous snags and relatively low overstory canopy cover (Kurta 2004), indicative of conditions following flood events in bottomland forest habitats in the Midwest (Carter and Feldhamer 2005) or following moderate to severe fires in upland habitats in the Appalachians (Beverly and Gumbert 2004). Although less demanding of particular thermal conditions (i.e., warm temperatures and direct solar radiation) helpful for the growth and development of juvenile bats, male Indiana bats also readily use day-roosts in snags created following prescribed burning on the Cumberland Plateau of Kentucky (J. MacGregor, Kentucky Wildlife and Fisheries Resources Commission, pers. commun.). Similarly, we have documented two male Indiana bats day-roosting in burned forest habitats in the Allegheny Mountains of West Virginia where diameter-limit harvests followed by early spring wildfire produced open conditions (20 to 50 percent canopy cover) and copious small to medium snags (< 40 cm d.b.h.) with large amounts of exfoliating bark. At that same site, Beverly and Gumbert (2004) described an Indiana bat maternity roost occurring in a small fire-killed red maple (*Acer rubrum*) snag (31 cm d.b.h.). This roost occurred at an elevation of 917 m where conditions were considered too cool during the growing season to support maternity activity (Brack et al. 2002). Because that roost was near a known winter hibernacula (< 1,000 m) and several male roosts in fire-killed sugar maple (*A. saccharum*) and red maple (unpublished), it has been hypothesized that Indiana bats in the Appalachians probably formed transitory maternity roosts within a shifting mosaic of habitats that may have followed disturbances such as fire across the landscape (T. Carter, Southern Illinois University, pers. commun.).

Fire effects on the structure of bat foraging habitat and fire impacts to bat prey resources are largely unknown (Carter et al. 2002). Menzel et al. (2005) and Ford et al. (2005) both hypothesized that forest management activities that decreased stem density and or created canopy gaps and openings would benefit several bat species in the east that are adapted to foraging in less cluttered conditions, e.g., big brown bat (*Eptesicus fuscus*) and hoary bat. If they are correct, prescribed fire could be a useful tool for enhancing foraging habitat for both species by creating a more complex canopy structure through mortality of individual canopy stems.

WHITE-TAILED DEER

Stransky and Harlow (1981) provided a thorough review of the literature regarding deer habitat and fire. Because there were few studies at that time directly examining deer and fire relationships, theirs was a thorough review of work on forage, including that for livestock, and fire effects on vegetation in order to draw inferences relating to the likely effect of fire on deer habitat. Stransky and Harlow concluded that winter burns increased forage crude protein and phosphorous (P) content; increased the number of woody plant stems; increased the coverage of grasses, forbs, and legumes; and reduced soft mast production. Most of these effects were short term (1 to 3 years). Infrequent growing season burns had similar effects except that woody stems were reduced. However, frequent summer burns would eventually eliminate woody stems and lead to domination of the site by grasses and fire-adapted forbs. These structural changes were established empirically for pine forest systems by the long-term (43 years) experiment conducted by Waldrop et al. (1992).

Wood (1988) and Wentworth (1986) each documented limited responses of vegetation to burning, albeit generally consistent with the foregoing summary of Stransky and Harlow (1981). Both studies were conducted in closed-canopy mature forests. Hurst et al. (1980) and Masters et al. (1993) conducted experiments in which they documented three- and sevenfold increases in forage production. The opening of canopies in both studies was critical to these responses with fire modifying plant communities released by canopy disturbance. Working in a scrub oak community in the Appalachians, Hallisey and Wood (1976) found that crude protein, P, and amount of forage generally were increased by fire. In another experiment, Brennan et al. (1998) precluded fire from a previously burned stand and reported a dramatic decrease in herbaceous cover largely due to encroachment by fire-intolerant woody vegetation. With the exception of Wentworth's (1986) and Hallisey and Wood's (1976) studies, all of the work cited was conducted in pine or pine-hardwood systems.

Two studies (Hobbs and Spowart 1984 and Main and Richardson 2002), neither conducted in eastern hardwoods, empirically documented behavioral changes (habitat use and foraging, respectively) of ungulates in response to fire. Main and Richardson (2002) deployed infrared triggered cameras in areas that had been burned 6, 24, and 48 months previously to document use by white-tailed deer. They reported differences between the 48- and 24-month postburn sites as well as between the 48- and 6-month postburn sites, with use by deer 3 to 4 times greater in the more recently burned sites. They concluded that deer were attracted to the availability of better forage in the more recently burned areas. Hobbs and Spowart (1984) reported only modest improvements in site-level forage quality, but substantial improvements to actual diets of both mountain sheep (*Ovis canadensis*) and mule deer (*Odocoileus hemionus*) on burned versus unburned sites in Colorado. They determined that due to selective foraging, these ruminants were able to increase both CP and in vitro digestible organic matter by 100 to 400 percent. They contend that studies that look only at plant responses underestimate the impact of burns on overall diet quality.

From the foregoing discussion it seems clear that where some canopy disturbance is present, fire alters forage quality and quantity, deer behavior, and likely deer diets. Unfortunately, no study has linked any population parameter to fire in a conclusive manner. Johnson et al. (1992) documented subtle, short-term improvements in white-tailed deer body mass and condition after a single large (18,200 ha) wildfire in the Coastal Plain of North Carolina. However, cause-and-effect data for fecundity, growth, and survival for deer on burned habitats are lacking. Given the degree to which white-tailed deer herds exceed cultural carrying capacity (society's tolerance threshold for a species) despite being below biological carrying capacity throughout much of this region (Keyser et al., in press), would suggest that ample nutrition is available to support large and healthy populations in the absence of fire-enhanced habitats.

An insidious impact of fire suppression on deer habitat in the region is the possible loss of oak mast through a marked reduction of the oak component in these forests. Mast is an important and preferred food source for deer (Harlow and Hooper 1972; Osborne et al. 1992; Wentworth et al. 1992), one that is widespread and often available in association with excellent cover. It is difficult to predict the impact of a large-scale loss of this resource on regional deer populations, but it is potentially profound. This would be especially true in landscapes that are heavily forested and lack agronomic forages that could easily offset the loss of high energy and highly palatable forage provided by acorns.

OTHER SPECIES

As is the case with deer, small mammals, and bat communities, little work has been done on fire effects on other mammal species that are associated with eastern oak forests. Both Hamilton (1981) and Weaver (2000) discussed fire influences on black bear but were confined to inferring effects on habitat because of a lack of published data. They concluded that periodic use of fire could be positive in terms of enhancing soft mast production and maintaining herbaceous plant communities. Fire can be important in both the formation and destruction of den trees, though we are unaware of any work that documents optimum regimes for recruitment and/or maintenance of cavity trees in such areas. In the Appalachians, bears frequently rely on accumulations of large woody debris for ground dens (Quince 2002), which also can be eliminated by fire. Thus, fire can have both positive and negative effects on availability of dens. Where fires provide broken canopies and more diverse midstory and understory development, particularly as it relates to soft mast and herbaceous vegetation, fire likely would result in positive changes in habitat quality. On the other hand, to the extent that fire negatively affects mast production, it may negatively effect habitat quality. We do not believe that fires intense or frequent enough to induce such changes are likely to be the norm in most eastern oak systems.

Much of the foregoing discussion about black bear denning habitat likely is applicable to raccoons (*Procyon lotor*). Owen (2003) documented use of both ground dens and tree cavities by raccoons in the central Appalachians, though maternity dens were almost exclusively in tree cavities. Thus, burning frequencies and intensities that maintain adequate snags appear to be important for this species. Although conducted in the Coastal Plain, Jones et al. (2004) documented a reduction in use of forests burned the previous year versus unburned stands by raccoons. It is difficult to infer how this might apply to upland hardwood systems. As with black bears, fire effects that enhance soft or hard mast production likely would increase raccoon use (Owen 2003).

We are aware of no literature that deals specifically with responses of gray (*Sciurus carolinensis*) or fox squirrels to fire, a circumstance that has not changed since Kirkpatrick and Mosby (1981) reviewed the literature. Although we are in agreement with them that fire is unlikely to positively affect gray squirrel habitat, it may have important implications for conservation of rare or threatened subspecies of fox squirrels. Fox squirrels prefer more open stand structure and greater understory development (Edwards et al. 2003), conditions that could be created through prescribed burning.

Elk (*Cervus elaphus*) may benefit from the increased use of fire in eastern forests due to their preference for herbaceous understories. Masters et al. (1997) reported increased use of burned areas by elk in a pine hardwood system in the Ouachita Mountains where herbaceous cover had been developed by repeated fires. Elk rely on a wide variety of forages, but grasses and forbs dominate elk diets on a year-round basis and in most habitats (Boyd 1978). Most elk restoration efforts in the Eastern United States focus on release sites with adequate herbaceous habitats (McClafferty and Parkhurst 2001). As interest in restoration of elk increases in eastern states and as elk populations expand in Pennsylvania, Kentucky, North Carolina, and Tennessee, managers should consider the role of fire in developing oak savannah habitats.

SUMMARY AND CONCLUSIONS

The state of our current understanding of fire effects on mammals in the oak forests of Eastern North America remains largely based on inference from work conducted in other regions or on indirect effects of fire on various habitat components. On the basis of these studies, it is apparent that fire can have a beneficial impact on habitat quality for a number of species of mammals. Were fire to be applied on a broad enough scale, these habitat enhancements could reasonably be expected to have a positive effect on population parameters as well. It is unclear in today's social and political climate that land managers will apply fire at such a scale. Conversely, it could be expected that on a localized scale fire could have a significant impact on local populations of mammals, either behaviorally or numerically. Thus, individual landowners or managers interested in enhancing habitat for deer could use prescribed fire as a tool.

However, the continued exclusion of fire from most landscapes in eastern oak forests may have serious implications for the continued presence of oaks as a major component of regional forests. It is this scenario that likely will have broader impacts on mammalian communities of the region going forward. Current trends in forest communities suggest that oaks will be a diminishing part of our future forests. Such a change could have a deleterious effect on numerous species of wildlife.

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Obstacles and Solutions to Using Prescribed Fire in Oak Forests

PRESCRIBED FIRE: WHAT INFLUENCES PUBLIC APPROVAL?

Sarah M. McCaffrey[1]

Abstract.—Except in remote areas, most prescribed fires will have some effect on members of the public. It is therefore important for land managers to work with the public before, during, and after a prescribed burn. To do this effectively, managers need to have an accurate idea of what people do and do not think about prescribed fire and they need to understand what shapes those opinions. This paper summarizes findings from recent research studies on the social acceptability of prescribed burns and identifies the key factors that people consider in forming their opinions of prescribed fire. Results indicate that there is a fairly high level of public acceptance for use of prescribed fire and that smoke, concerns about escape, and trust are key issues shaping that support. In addition, there is a clear link between understanding of the purpose and intended benefits of prescribed fire and approval of its use. The lesson for managers who wish to introduce prescribed fire in their communities is that they are most likely to gain public support if they: 1) increase familiarity with the practice; and 2) work to build trust between officials from the implementing agency and the public

INTRODUCTION

> *I think what happens when we have prescribed burns is the majority of the people say "Well, that's something that has to be done." And there's a minority of the people that complain about it, but they get their names in the paper.*[2]
> *(*Hamilton focus group participant)

Prescribed burning is a key tool for managers working to reduce fuel loads or restore fire adapted ecosystems. Yet it can also be a problematic practice that the public may not accept or support. Since prescribed fires in all but the most remote areas will have some effect on members of the public, it is important for managers to work with the public before, during, and after a prescribed burn. To do this well, it is useful to have an accurate idea of what people do and do not think about prescribed fire and what shapes those perceptions. As the introductory quote suggests, public views of prescribed fire are generally more sophisticated and less negative than managers might expect.

This paper will discuss findings from recent studies (most sponsored by the National Fire Plan) about the social acceptability of prescribed fire, the key variables that influence approval or disapproval, and the roles that those variables play in shaping opinion. Basic information about these studies is summarized in Table 1. Some of the studies have been completed while others are still in progress; information about them is drawn from published articles, project reports, and, in one case (See Table 1, McCaffrey 2005), directly from focus group transcripts. Although there is local variation in forest composition in the study areas[3], the findings are reasonably consistent across diverse ecosystems and different regions of the country. This paper should therefore provide managers with a sense of the basic dynamics that shape public opinions of prescribed fire to help guide development of programs that fit local circumstances.

APPROVAL

Prescribed burning is a largely acceptable practice with roughly 80-90% percent of respondents across studies finding it an appropriate management tool (Bright and Carroll 2004; Cortner et al. 1984,;McCaffrey 2002; Shelby and Speaker 1990; Shindler et al. 1996). In those surveys that explored strength of support, roughly 30 percent of respondents gave strong approval for use of

[1]USDA Forest Service, Northern Research Station, 1033 University Place, Suite 360, Evanston, IL 60201-3172, 847-866-9311 ext. 20, email: smccaffrey@fs.fed.us.

[2]All quotes are taken from participants in a series of focus groups held to examine public views of various aspects of fire management (McCaffrey 2005, Table 1,). A total of fifteen focus groups were held in five different fire prone locations in the Western United States. Focus groups were made up of randomly selected local members of the public and were transcribed verbatim.

[3]Only the California and Missouri sites of the Winter et al. 2005 study included some oak woodlands.

Table 1.

Study	Where	Who	Method
Blanchard & Ryan 2002	Within a two mile radius of Myles Standish State Forest in Massachusetts	Seasonal and year round residents	Mail survey
Bright and Carroll 2004	Front Range, Colorado, Southern Illinois, Chicago Metropolitan area	Residents near National Forests and random Chicago households	Mail survey
Brunson and Evans 2005	Wasatch, Utah, and Salt Lake counties	Residents (including 113 who had been answered the same survey two years earlier)	Longitudinal mail survey
McCaffrey 2005	Flagstaff, AZ; Boulder, CO; Hamilton, MT; Reno, NV; San Bernardino, CA	General members of the public	Focus groups
Shindler et al. 2003	Forest communities adjacent to National Forests in Wisconsin, Michigan, and Minnesota	Residents	Mail survey
Weisshaupt et al. 2005	Missoula, MT; Spokane, WA	Native Americans, urban and rural residents and an anti-smoke group	Focus groups
Winter et al. 2005, 2004, 2002	California, Florida, Michigan and Missouri	Homeowners near forested lands	Focus groups and mail survey

prescribed burning and another 50 percent gave qualified approval (Blanchard 2003; Brunson and Evans 2005, Shindler and Toman 2003, Shindler et al. 2003; Winter et al. 2005).

SMOKE

Smoke is often considered a major barrier to use of prescribed fire. This is a reasonable expectation given that smoke is a health problem for roughly 30 percent of study households (Blanchard 2003; McCaffrey 2002; Shindler et al. 1996; Winter et al. 2005). However, in their four state study, Winter et al. (2005) found that smoke was significantly related to prescribed fire attitudes in only one site, Missouri, where the belief that prescribed burning meant more smoke now and less later was positively related to approval. In general, people appear to understand that no smoke is an unrealistic option: they will be exposed to smoke, either from a wildfire or from a prescribed burn, and so long-term trade-offs will need to be made. One way to manage the health issues is to provide adequate warning of a prescribed burn allows those with health issues to make arrangements.

> *I think they would (tolerate smoke), if it is communicated ahead of time so that asthmatics could stay inside; like we do now when they broadcast something that there's a wildfire. You can plan ahead. (*Reno focus group participant)

> *Maybe if it had advance notice, they would know for how long it would take and what the purpose was. (*Boulder focus group participant)

Focus groups in Washington found that, although people often don't differentiate, the source of the smoke can influence approval (Weisshaupt et al. 2005). Members of an anti-smoke group remained opposed to smoke from agricultural burning because all benefits went to the farmer but were more open to smoke from prescribed fire because the benefits of burning accrued to all. Many participants, including those in the anti-smoke group, were willing to make trade-offs between some "managed" smoke now in return for less smoke from future wildfires.

Topography is a local variable that can influence acceptability of prescribed fire due to smoke concerns.

People who live in areas prone to inversions or in valleys that "collect" smoke from other areas are more sensitive to smoke issues and may find prescribed fire less acceptable. This dynamic is demonstrated by the following excerpt from a Reno focus group.

Ann:	*I don't think very many people would tolerate it. My Grandma, she has problems like that. When there's smoke, she can't tolerate it. A lot of people can't tolerate it.*
Barbara:	*And we are in a valley, so it all just sits here.*
Carl	*Inversion*.
Carol:	*Even when they do a controlled prescribed burn in Yosemite, it still comes here.*

But individuals in these areas also may recognize that topography means they have little control over the smoke from wildfires but some control over that of prescribed fire, as in Hamilton.

> *Come August the whole valley is going to be filled with smoke anyway. If they can do something productive and burn away some of that slash that might slow down the fire a bit, then I'm all for it.*

CONTROL

Concern over a burn getting out of control is another major issue. Early studies found that fear of a prescribed burn escaping influenced approval (Cortner 1984; Shelby and Speaker 1990). Seventy percent of the respondents in Shindler et al.'s study (2003) indicated they were moderately to greatly concerned about such a possibility. Winter et al. (2005) found that the belief that prescribed fire would lead to uncontrolled fires was negatively related to acceptance, the only outcome belief significant across all four study sites.

In Utah, Brunson and Evans (2005) compared responses from individuals surveyed in 2001 and again in 2003 after an escaped burn (Cascade II) occurred in the area. They found that, after the escape, respondents expressed significantly increased concern (from 19 to 44 percent) about a prescribed fire taking place within 10 miles of their home suggesting that the escape may have led some people to have lower confidence about the ability to keep a prescribed fire under control. Notably, although almost half of the respondents stated that they held a more negative view of prescribed fire after the escape, in reality, when compared with responses from before the escape, their judgements of the acceptability of prescribed fire had not changed significantly. What had changed was an increased concern about the health impacts of smoke and decreased belief that smoke could be managed acceptably. Even with these changes only 13 percent thought smoke made prescribed fire not worth using.

Part of the issue with concerns about escaped burns is that they are what grabs people's attention, not the successful burns—this creates a rather small sample upon which people base their opinions. In the series of focus groups throughout the West (McCaffrey 2005, Table 1), conversations about prescribed fire usually started with an instant negative reaction related to escapes but, in several cases, discussion then evolved around the question of how many problem burns there actually were in a given year compared to non problem burns. When people concluded that escapes were a very small percentage of total burns they were much more comfortable with them. They suggested that more effort needs to be made to publicize *all* prescribed burns to provide some perspective on the relative number of escaped burns.

> *I think we need to know more. Just like John said, if 90% of them are successful, we need to know about it. But we just hear about the ones that aren't. (Reno focus group participant)*

> *The only time you hear it is always the bad fires. I think that they* (prescribed burns) *are helping us a lot. I would say approximately 5% goes out of hand. (San Bernardino focus group participant)*

TRUST

Trust in the agency administering the burn is the remaining key variable that shapes public acceptance of prescribed fire. As one Reno focus group member said when asked whether people would tolerate more prescribed burning: "more if you trust the guy that starts it." Winter et al. (2005) found that trust in government was a significant predictor of intention to approve prescribed burning in all four study sites. In Missouri, which had the highest trust levels of the four states, trust also had the largest effect on attitudes. Perhaps the most important impact of the escaped burn in Utah was not its effect on public views of prescribed burning but on trust levels which decreased significantly for both the U.S. Forest Service and the Bureau of Land Management but not for state, county, or local government agencies, with the local rural residents showing lower trust levels than the metropolitan sample (Brunson and Evans 2005). For three of their study sites (California, Florida, and Michigan), Winter et al. (2004) examined what elements were associated with trust and found that the strongest association for all three sites involved views on agency competence.

OTHER VARIABLES

A variety of other issues such as past experience, wildlife concerns, and aesthetics are thought by many managers to influence approval of prescribed fire. However, results from the referenced studies do not show as strong or as consistent an effect as smoke, control, and trust. Most studies found no significant relationships with these three variables and those that did did not necessarily find a dependable effect. For instance, in terms of experience, Blanchard and Ryan (2004) found that individuals with past personal experience of wildland fire had a higher level of support for use of prescribed fire than those who had not. Conversely, Winter et al. (2005) found that past experience with wildland fire or prescribed fire was not a significant explainer of attitude.

Studies that explored how concerns about wildlife shaped acceptability also showed mixed results. Almost half of Shindler et al.'s (2003) respondents expressed at least moderate concern about loss of wildlife and fish habitat from a prescribed burn but 68 percent also thought it improved wildlife habitat. Winter et al. (2005) found that belief that prescribed fire improved wildlife conditions was positively related with approval in two of their sites: California and Michigan. In terms of aesthetics, while 42 percent of respondents in Minnesota, Wisconsin, and Michigan expressed at least moderate concern about scenic quality, only 14 percent felt prescribed fire impacts on scenic quality were unacceptable (Shindler et al. 2003). Winter et al. (2005) found a weak association between aesthetics and approval for the California site and no association for the other three sites.

Finally, one item to consider that was examined in only one of the studies (Winter et al. 2005) is the belief that use of prescribed fire reduces firefighting costs in the long run—which was positively related with attitudes in California, Michigan, and Florida.

UNDERSTANDING

The most consistently found relationship in the studies is the concept that familiarity with a practice leads to acceptance. That knowledge and familiarity with a practice is associated with increased support for fuels management practices fits with findings from earlier wildfire studies (Carpenter et al. 1986; Gardner and Cortner 1988; Loomis 2001; McCaffrey 2002). More recent studies have also found a strong link between knowledge and support for a treatment method, whether prescribed fire or thinning. Shindler et al. (2003) found that support for both treatment methods was significantly associated with the respondent's natural resource knowledge—the greater the knowledge the greater the support as well as the greater the confidence in the U.S. Forest Service. Of the three states surveyed, Minnesotans were the best informed and the most tolerant of fuel treatments while Michiganders were least informed and least supportive. In Massachusetts, Blanchard and Ryan (2004) found knowledge levels to be the most significant factor determining support for prescribed fire; with a higher level of knowledge of prescribed burning significantly associated with increased support for its use as well as lower concern with related risks. Those with some knowledge of prescribed burning were less likely to think it was too dangerous a practice to be used, to be concerned about prescribed burning near a home, and to be concerned about smoke, appearance, and the effects on animals and their habitat.

In Florida, the state in the Winter et al. study (2005) where prescribed fires were most common, respondents had the highest approval rate for prescribed burning and, as previously discussed, also had the highest level of trust in government agencies doing prescribed burning. They also were more likely to think that prescribed burns restored more natural conditions and improved conditions for wildlife. In the Washington focus groups, tolerance for prescribed burning increased, particularly amongst members of the anti-smoke group, as participants learned new information about the practice during discussion (Weisshaupt et al. 2005).

Understanding the ecological benefits of prescribed burning appears to be particularly important in shaping approval. Carpenter et al. (1986) reviewed three previous studies and found that acknowledgement of beneficial effects was the most "pervasive" influence in approving various fire management methods. More recently, Winter et al. (2005) found that belief that prescribed burning restored wildlands was positively associated with attitudes toward prescribed fire in Missouri as well as in Michigan (albeit at a weaker level), while, as indicated earlier, beliefs that prescribed fire improved wildlife conditions was positively associated with attitude toward the practice in California and Michigan.

Understanding ecological benefits can also make smoke less of a concern. In the Washington State focus groups, participants became more tolerant of smoke from a burn as they understood the beneficial effects of prescribed fire (Weisshaupt et al.2005). Shindler found that 2/3 of respondents in Oregon agreed that smoke was acceptable if it helped forest health (Shindler et al. 1996). As one Hamilton focus group participant responded when asked if people would tolerate smoke from increased use of prescribed burns: "I think most people would tolerate it, if they think, they knew it was good."

CAVEATS

Results from the referenced studies thus indicate that, at a general level, there is a fairly high level of public acceptance for use of prescribed fire and that smoke, control and trust are key issues shaping that support. However, several caveats are important to keep in mind in applying this knowledge.

Local context matters

Local context, such as history and cultural practices, needs to be taken into account as it can have a significant effect on specific attitudes. For instance, Winter et al. (2002) found two exceptions to the general pattern of around 30 percent strong approval for prescribed burning: in Florida 40 percent were extremely positive about prescribed burning whereas in Michigan a mere 10 % were extremely positive. This last is generally attributed to a 1980 prescribed fire that escaped and killed a firefighter, destroyed 44 houses, and is still discussed. Trust in government was also much lower in Michigan where only 27 percent of respondents trusted the government to make proper decisions about use of prescribed burning, as compared to 55 percent in Florida and 46 percent in California.

Avoid Preconceived Notions

It also is important to be careful of preconceived notions. Bright and Carroll's (2004) study found very few significant differences between three groups often thought to hold different views: homeowners near National Forests along the Front Range of Colorado (classic wildland-urban interface or WUI) and Southern Illinois (very rural) and residents of the Chicago Metropolitan region. For all three groups, the most important factor in positive support for prescribed fire was if there was a recent history of fire and the second most important factor was if the burn was in a remote areas. The only major difference between the locations was in how primary use of the forest shaped acceptability. Illinois residents from both locations found prescribed fire more acceptable if the primary use of the land was commercial, while Front Range residents found it more acceptable if the primary use was recreational. However, primary use was less important in shaping acceptability than current conditions and forest location. The fact that such disparate areas have largely similar views is one example of how dichotomies, such as urban-rural and WUI versus non-WUI, can be misleading in terms of understanding public beliefs on fire management.

There also may be expectations that various socio-demographic characteristics, such as income or age, will be associated with specific attitudes. Although some studies have found socio-demographic variables that were tied to attitudes, there has been no clear or consistent pattern.

Understanding is a two way street

Finally, the fact that there is a clear link between familiarity with a practice and acceptance does not mean that increasing acceptance of prescribed fire is simply a case of providing information. Shindler's survey of Wisconsin, Minnesota, and Michigan showed that the most trustworthy and most helpful methods of information dissemination were guided field trips and interaction with agency personnel. Such interactive methods are most effective at changing attitudes and behavior as they allow people to question and clarify new information (Monroe et al. 2005). Manager's in turn can learn through this process about key public concerns and issues and tailor their management efforts to account for them.

CONCLUSION

Contrary to the expectations of many managers, prescribed fire is a largely acceptable practice where objections to sensitive issues such as smoke and loss of control can potentially be overcome with dialogue as understanding of purpose and benefits increases tolerance. Thus managers introducing prescribed fire as a new tool may not have immediate acceptance but likely can look forward to increased public acceptance and support as people become more familiar with the practice. However, this increased acceptance is not automatic. Trust in agency implementation is also important. Even with good knowledge, low trust levels will likely mean low tolerance for prescribed fire. Fortunately, the very dialogue that agencies engage in to build knowledge bases can also help build relationships and trust.

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TOP TEN SMOKE MANAGEMENT QUESTIONS FOR FIRE IN EASTERN OAK FORESTS

Joseph J. Charney[1†], Ann L. Acheson[2], and Andrea Stacy[3]

INTRODUCTION

In a time when fire planners and forest supervisors in many parts of the United States are implementing plans to increase the number and frequency of prescribed burns, the issue of smoke impacts on air quality is becoming increasingly important. Smoke from prescribed fires can have a dramatic impact on air quality and visibility on local, regional, and national scales, with corresponding temporal scales ranging from hours to days to months. As new national air quality regulations are implemented, smoke management will become an increasingly important concern when planning prescribed burns in the Eastern Oak Forests.

In the Eastern Oak Forests, prescribed fire is becoming a more common practice in ecosystem management (Ward and Brose 2004). Although there are many resource benefits associated with prescribed fire, the potential for the resultant smoke to impact human health and public safety also needs to be addressed. Wildland fires release various pollutants, including, but not limited to, fine particulate matter, nitrogen oxides (which are precursors to ozone formation and, thus, a surrogate for ozone), volatile organic compounds (also precursors to ozone formation), and carbon monoxide. However, in terms of prescribed fire, the pollutants of major concern are the fine particulates, referred to as $PM_{2.5}$. About 70 percent of particulates released by wildland fires are within the fine particle size class (particles less than 2.5 microns in diameter) (Hardy et al. 2002). Elevated levels of fine particulates are dangerous because they can penetrate deep into human lungs and increase the risk of serious health problems, especially for those with respiratory illnesses. Furthermore, when smoke plumes with high particulate concentrations intersect roads and highways, visibility is reduced and the likelihood of traffic accidents increases. Ozone, which is formed through atmospheric reactions of nitrogen oxides and volatile organic compounds, is also a respiratory irritant.

As population centers surrounding eastern oak forests expand and encroach on these ecosystems, increasing the wildland/urban interface, smoke management issues will become increasingly important and increasingly complex. In some situations, smoke management could become the determining factor in “go/no go” decisions for prescribed burning.

The purpose of this paper is to outline pertinent smoke management questions that fire and smoke managers in the Central Hardwoods will face in the near future. The next section identifies and describes the “Top Ten” management questions that burn planners should address when implementing burning programs in the region. Section 3 will then describe the smoke management tools that are currently available, or that will be available in the near future, to help fire and smoke managers address these questions when preparing for prescribed burns. Section 4 includes a discussion and some concluding points.

TOP TEN SMOKE MANAGEMENT QUESTIONS

This section presents what we feel are the Top Ten smoke management questions that will impact prescribed burn decisions in eastern oak forests in the near future.

10. *Where are eastern oak forests—Where will burning occur?*

The Central Hardwoods (Fig. 1) cover an area from northern Georgia, west to Arkansas, northeast to the Great Lakes and New England, and then south along the eastern slopes of the Appalachians (Stringer and Loftis 1999). This area is predominantly oak-dominated deciduous forests in hilly-to-mountainous areas. The

[1]USDA Forest Service, Northern Research Station, Stephen S. Nisbet Bldg., 1407 S. Harrison Road, Room 220, East Lansing, MI 48823; [2]USDA Forest Service, Region 9, Fire and Aviation Management, 27750 State Route 7, Marietta, OH 45750; [3]USDA Forest Service, Monongahela National Forest, 200 Sycamore St., Elkins, WV 26241; [†]Corresponding author, 517-355-7740 ext. 105, email: jcharney@fs.fed.us.

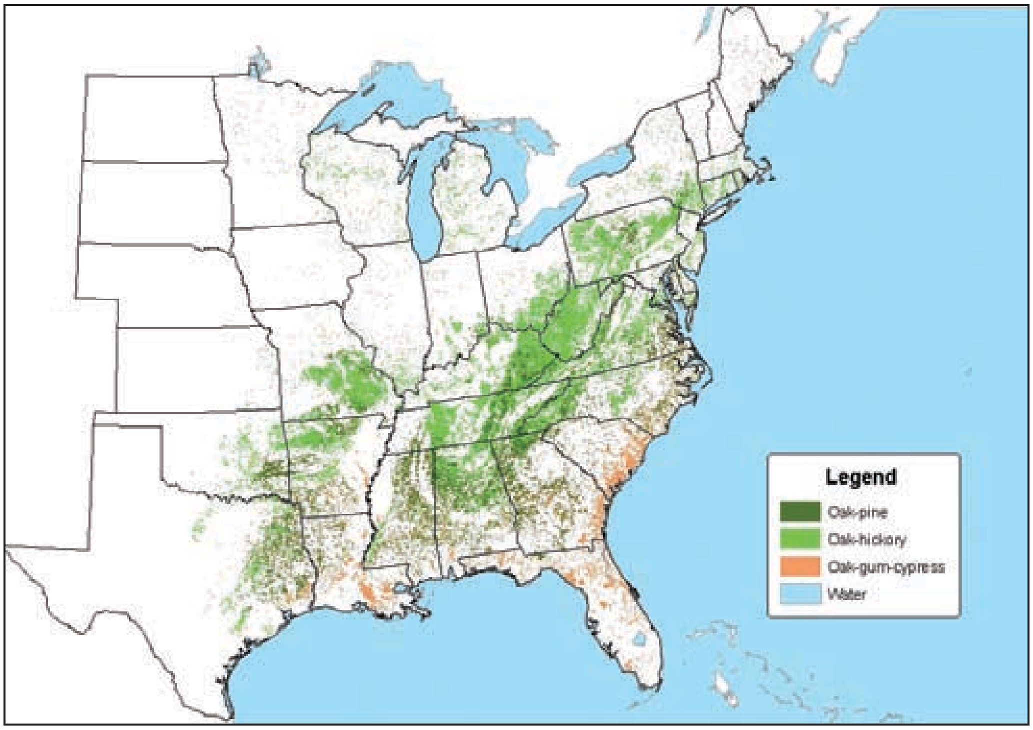

Figure 1.—Distribution of forests in the eastern United States.

Central Hardwoods cover approximately 125 million acres in eleven states, making it the most extensive temperate deciduous forest in the world.

9. *Where are the people—what percentage of people in the US live within the vicinity of eastern oak forests?*

Approximately 65 million people live in the states east of the Mississippi (U.S. Bureau of the Census 2000), with the highest population density along the Eastern Seaboard, the Great Lakes coast, and the southern tip of the Appalachians (Fig. 2). Approximately 25 percent of the population of the United States lives in the Central Hardwoods states, and over 90 percent of the Central Hardwood Forest is owned by private interests, primarily non-industrial forest owners (Hicks 1997).

8. Where are the roads?

One of the major concerns with smoke management is whether the smoke from a prescribed burn will impact a roadway. The highest density of interstate highways and primary and secondary roads in the Unites States occurs east of the Mississippi (Fig. 3). A major source of complexity in eastern smoke management and prescribed burning is the fact that a road of some sort almost invariably occurs within the immediate locale of a burn site.

7. Where are existing air pollution problem areas?

Outside of California, the eastern United States has the greatest number of areas not meeting the National Ambient Air Quality Standards (NAAQS) for a variety of criteria pollutants (U.S. EPA 2006): sulfur dioxide (SO_2), nitrogen oxide (NO_x), particulate matter less than or equal to 10 microns or 2.5 microns in diameter (PM_{10} and $PM_{2.5}$), ozone (O_3), carbon monoxide (CO), and lead (Pb) (Fig. 4). The NAAQS were developed by the Environmental Protection Agency for a specific set of "criteria" pollutants to protect public health and welfare. NAAQS are defined as the amount of a criteria pollutant above which detrimental effects to human health or welfare may result. The NAAQS are set at conservative levels, with the intent of protecting even the most sensitive members of the public including children and people with asthma and cardiovascular disease (http://www.epa.gov/air/criteria.html). Nevertheless, states can

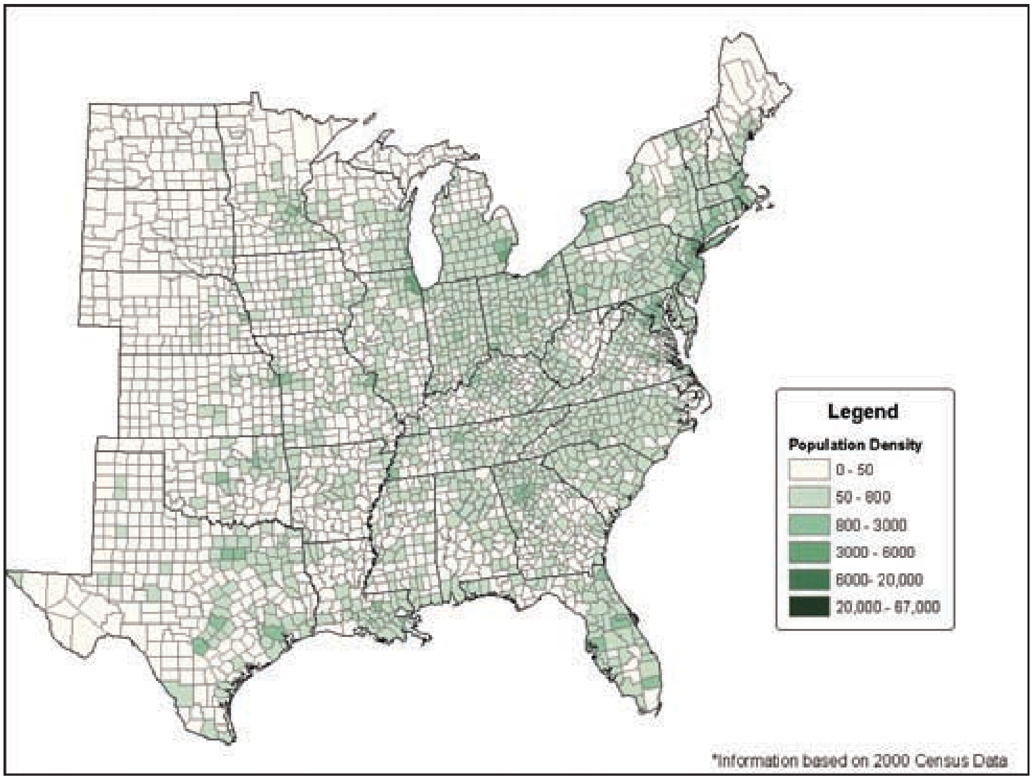

Figure 2.—Population density in the eastern United States.

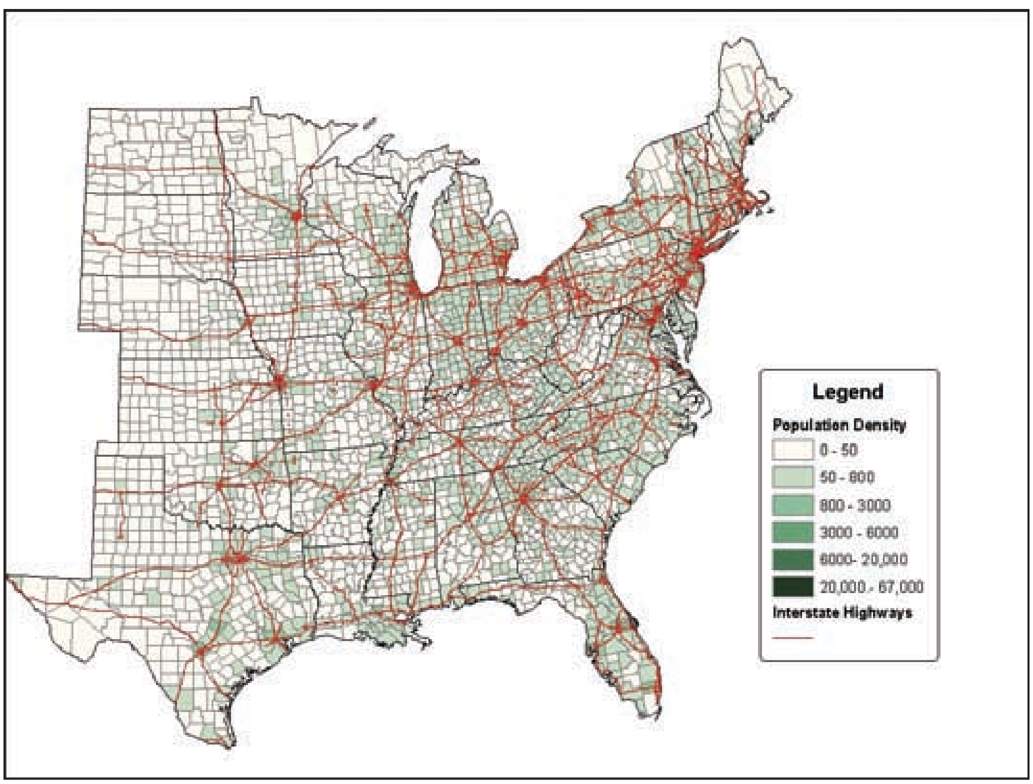

Figure 3.—Locations of Interstate Highways overlaying population density in the eastern United States.

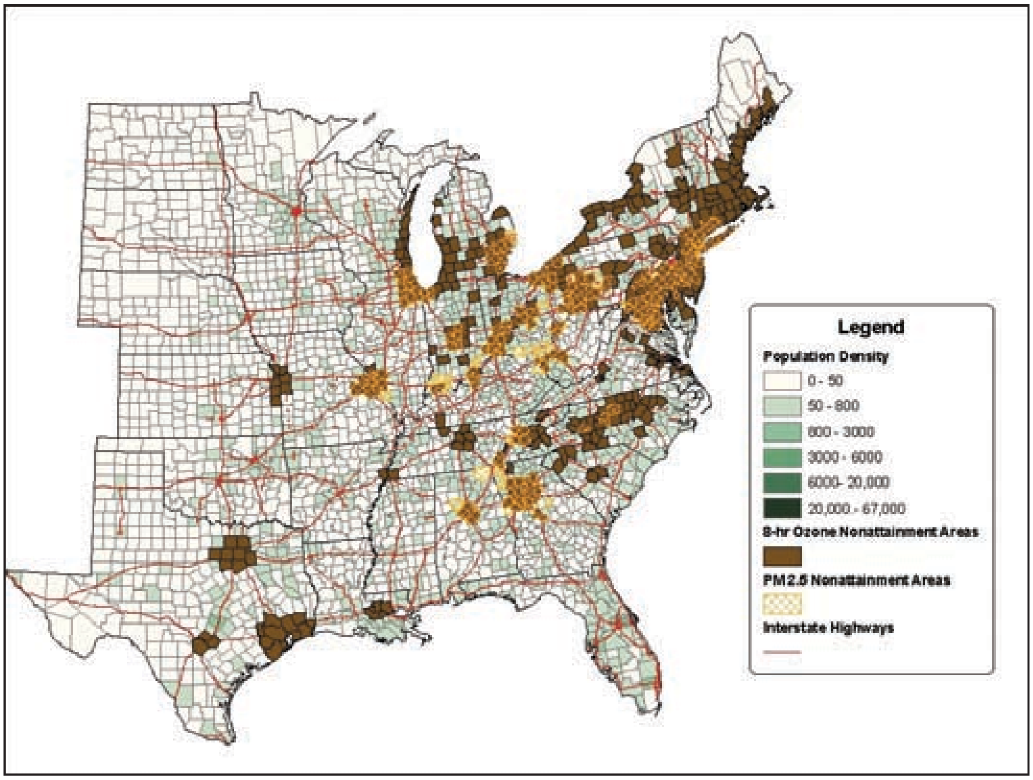

Figure 4.—Locations of federally designated Ozone and PM2.5 non-attainment areas overlaying interstate highway locations and population density in the eastern United States.

adopt standards even more stringent than the federal standards. If an area consistently violates one of the NAAQS, that area will be federally designated as a "non-attainment" area for the specific criteria pollutant(s). States are required to demonstrate to the public and the EPA how emissions will be controlled in order for the non-attainment area to again comply with the NAAQS. These demonstrations are included in a State Implementation Plan (SIP), which may include methods to control emissions from prescribed fire.

6. *Where/What are the Class I areas?*

Mandatory federal Class I Areas were designated by the Clean Air Act amendments (CAAA) of August 7, 1977, and include national parks greater than 6,000 acres and national wilderness areas and national memorial parks greater than 5,000 acres which existed at that time (42 U.S.C. 7401 et seq). Resources such as soils, streams and visibility in these Class I Areas (called Air Quality Related Values) were given an additional measure of protection under the CAAA. Although the majority of Class I areas are in the west, several are scattered across eastern oak forests (Fig. 5). The eastern Class I areas are often located in, or near, densely populated areas, and receive extensive visitor use.

Regional Haze Rules address regional emissions that affect visibility in Class I areas. Recognizing that regional haze results from many sources of pollution over a large geographic area, these regulations require states (and tribes who choose to participate) to review how emissions generated within the State affect visibility at "Class I" areas. The EPA has encouraged states and tribes across the U.S. to address visibility impairment from a regional perspective and as a result the Regional Planning Organizations (RPO's) (Fig. 6) were formed. These organizations are evaluating available technical information to better understand how the sources within their states and Tribal lands impact Class I areas across the country. This technical information can then be used by the states and tribes to develop regional strategies to reduce emissions of pollutants leading to regional haze (from: http://www.epa.gov/air/visibility/regional.html).

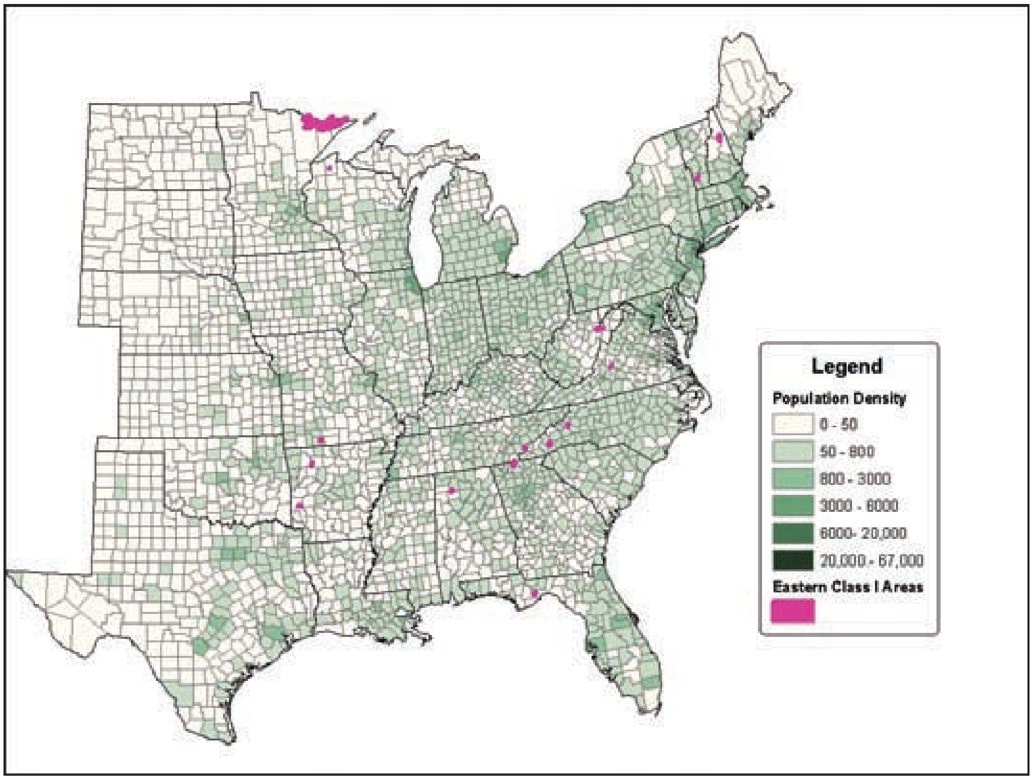

Figure 5.—Locations of Class I areas overlaying population density in the eastern United States.

Figure 6.—Names and member states of the five Regional Planning Organizations

States are required by the rule to analyze a pathway that takes the Class I areas from currently impaired conditions to "natural conditions" in 60 years. The term, "natural visibility conditions" is meant to "represent the long-term degree of visibility that is estimated to exist in a given mandatory Federal Class I area in the absence of human-caused impairment" (U.S. EPA 2003; pg. 1-1). States must show they are making "reasonable progress" toward achieving this goal. The Regional Haze Rule sets up the period of 2000 to 2004 as the baseline visibility conditions and "reasonable progress" will be tracked for that period onward. States currently are coordinating through the RPOs to determine the necessary level of reductions in haze forming emissions to achieve this. State Implementation Plans (SIPs) will need to be developed to outline strategies and schedules for emission reductions. Prescribed fire emissions are to be included in the suite of emission sources that potentially impact visibility. Most states will need to have a SIP in place by 2008 to cover a period of 10-years. While this program is aimed at Class I areas, visibility will improve throughout the country by reducing regional emissions.

Because the pollutants that lead to regional haze can originate from sources located across broad geographic areas, the EPA has encouraged the states and tribes across the U.S. to address visibility impairment from a regional perspective. Therefore, Regional Planning Organizations (RPO's) (Fig. 6) have been formed to address regional haze and related issues. These organizations are

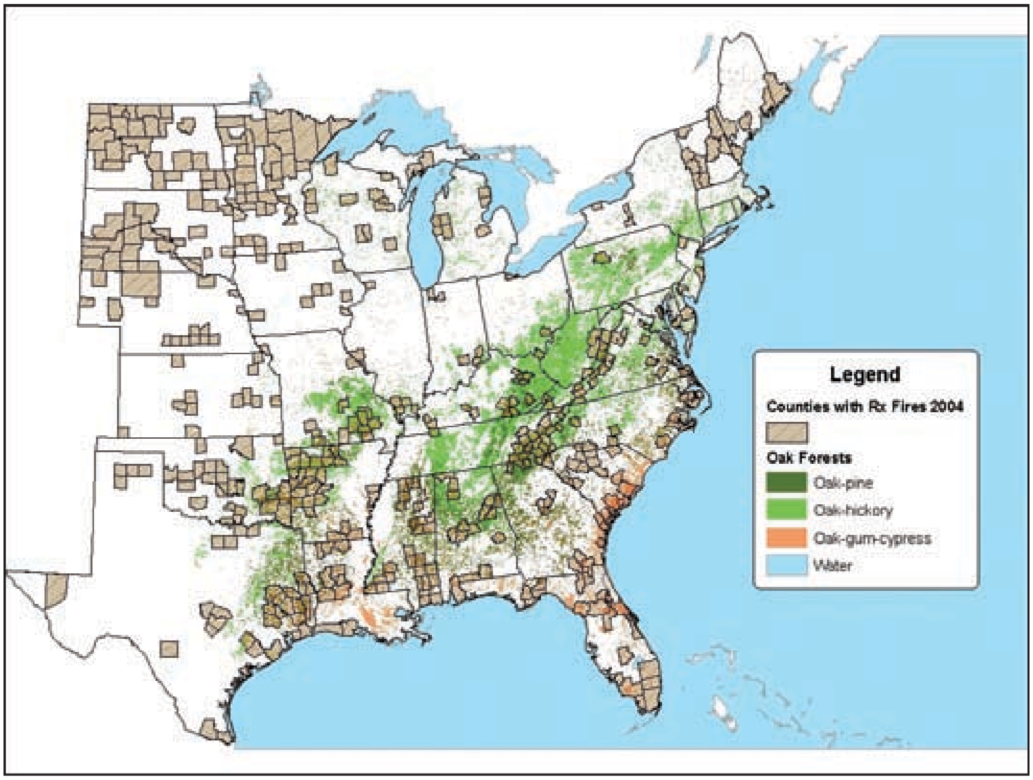

Figure 7.—Counties reporting prescribed fire activity in 2004 overlaying locations of oak forests in the eastern United States.

evaluating available technical information to better understand how the sources within their states and Tribal lands impact Class I areas across the country. This technical information can then be used by the states and tribes to develop regional strategies to reduce emissions of pollutants leading to regional haze (from: http://www.epa.gov/air/visibility/regional.html).

5. *Where and when does burning typically occur?*

Figure 7 shows the counties in the eastern United States that reported federal prescribed burn activity in 2004 (http://www.nationalatlas.gov/natlas/Natlasstart.asp). In addition to the locations indicated on this map, additional prescribed burn activity undertaken by state and local organizations occurs throughout the region.

4. *Where does smoke typically go?*

Smoke from prescribed burns has local, regional, and national impacts. Local impacts, such as those to sensitive receptors (homes, schools, businesses) directly downwind of a burn site are routinely addressed in burn plans. Regional impacts are sometimes included in burn plans, as some plans account for population centers away from the burn site. Additionally, there is evidence that smoke from prescribed burns can impact specific sites hundreds of miles away from the burn location, contributing to impacts more on a national scale. Figure 8 shows

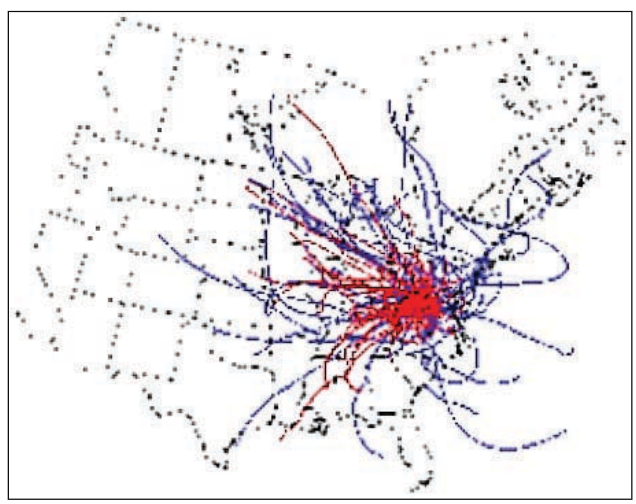

Figure 8.—48-hour back trajectories for 20% best (blue) and 20% worst (red) haze days in the Shenandoah National Park, 1997-1999 (from Regional Haze and Visibility in the Upper Midwest. Midwest Regional Planning Organization, (http://www.ladco.org/tech/monitoring/docs_gifs/rhreport.pdf).

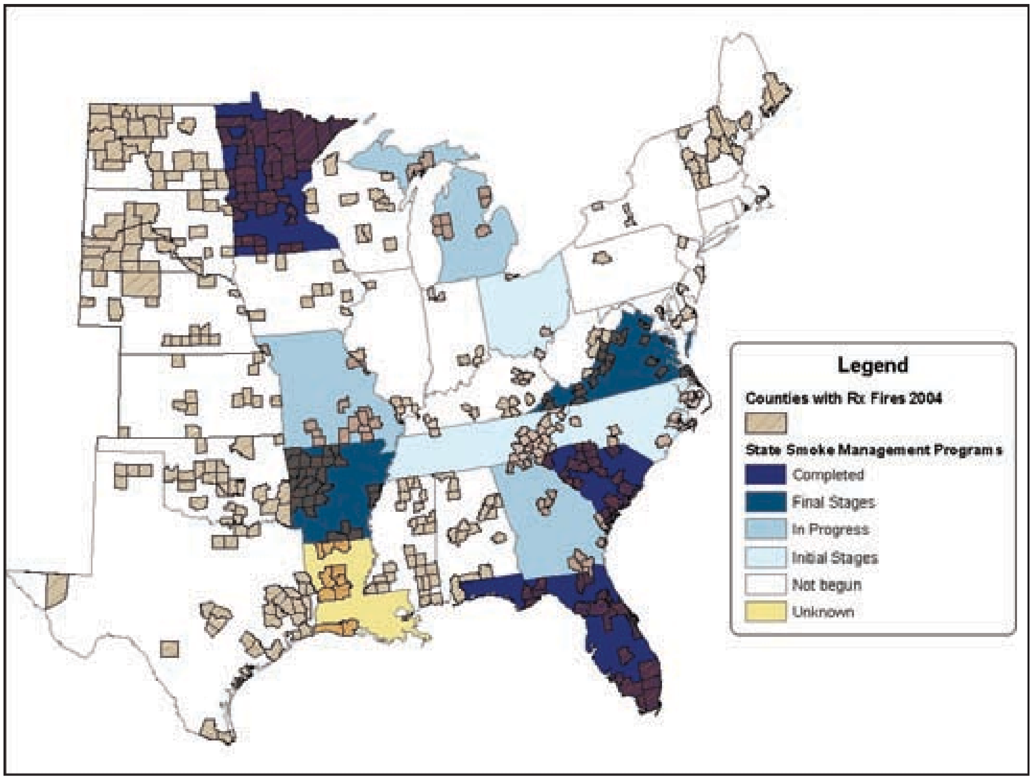

Figure 9.—Status of efforts to design and implement state Smoke Management Programs overlaying counties reporting prescribed burn activity in 2004 in the eastern United States.

a 48-hour trajectory analysis for the best and worst visibility conditions measured at Shenandoah National Park from 1997-1999 (Midwest RPO, 2001). This analysis indicates that the source air a given location, such as a sensitive receptor or a national park can come from locations all across the eastern United States. The implication here is that any prescribed burn activity has the *potential* to impact air quality and visibility conditions on a regional and national scale. As states implement smoke management programs designed to monitor and, if necessary, alter burn programs to control air quality at specific sites, prescribed burn planning will become increasingly dependent upon modeling tools that assess the potential for smoke to influence regional and national air quality.

3. *Where is burning expected to increase/decrease?*

The USDA Forest Service Region 9, which stretches from Minnesota south to Missouri and east to the Atlantic Coast, treated (using prescribed fire and other fuel treatments) approximately 96,473 acres in FY 2005 (USDA Forest Service 2006). Nationally, the USDA Forest Service in 2005 had a target of treating 2.5 million acres of hazardous fuels through a variety of prescribed fire and other vegetation treatments. In FY 2007 the target is approximately 3.2 million acres of hazardous fuels treatments. Each state also has treatment plans that include prescribed burning as an activity. Prescribed burning in mixed oak forests, and throughout the United States, is expected to increase in the near future.

2. *Where are there coordinated Smoke Management Programs?*

There is a wide variety in the type or existence of state Smoke Management Programs (SMPs). Florida has perhaps the best-established SMP in the East, with daily burning coordinated through a central clearinghouse in the Florida Department of Forestry. South Carolina and Minnesota likewise have SMPs in place. But many other states throughout the region are in the early stages of developing SMPs or have nothing at all (Fig. 9). As these programs become established and smoke management regulations are more clearly defined, the impact of smoke management on prescribed burn planning and execution will become more apparent.

1. *What does all this mean to fire and smoke managers?*

When all of the elements listed above are considered together, it becomes clear that smoke management in the Oak forests can be extraordinarily complex. This burn activity takes place in a part of the country with the greatest population (compared to the west) and in states with existing air pollution problems and an extensive transportation system. Additionally, forest managers are hoping to increase the number and size of prescribed burns in the near future. Finally, coordinated SMPs designed to account for the cumulative effects of burning are few and far between in the region.

SMOKE MANAGEMENT TOOLS

There are many smoke management tools available to fire and smoke managers to aid in planning for a burn. The tools indicate how readily smoke will disperse from the fire area, and where the smoke will go in the ensuing hours. The National Weather Service provides forecasts of the Ventilation Index (VI) (Hardy et al. 2002; and others) in routine fire weather forecasts and spot forecasts requested by fire managers. The VI provides an assessment of the extent to which smoke produced by a fire will be effectively transported (ventilated) from the fire area. However, while the VI indicates how readily smoke will disperse, it does not indicate where the smoke will go once it leaves the burn site. A smoke dispersion model is necessary to determine where smoke is likely to go and what the pollution concentrations (mainly PM 2.5) will be downwind of the burn.

The model that a fire or smoke manager employs depends strongly on what questions need to be answered. For questions concerning smoke concentrations within a few miles of a fire, dispersion models driven by single observations take at or near the fire location are often sufficient. VSMOKE (Lavdas 1996) is arguably the most advanced model that is currently in operational use in the eastern United States. VSMOKE is designed to help smoke and fire managers predict the smoke and dry weather visibility impact of a single fire at downwind locations. VSMOKE contains a plume rise model, a steady-state trajectory model, and other components that provide predictions of concentrations, visibility and other factors. Additional examples of dispersion models in use include: 1) the EPA's Industrial Source Complex (ISC3) model, which is used for predicting dispersion of pollutants from industrial facilities mainly for permitting purposes (U.S. EPA 1995); 2) the Smoke Impact Spreadsheet (SIS), a simple planning puff model for calculating particulate matter emissions and concentrations downwind of wildland fires (Wickman and Acheson, 2005), and 3) The Simple Approach Smoke Estimation Model (SASEM), which predicts ground level particulate matter and visibility impacts from single sources in relatively flat terrain (Sestak and Riebau, 1988). All of these models can be used to assess smoke concentrations immediately downwind of a fire on time scales of a few hours. Additional details about the applications and relative merits of these and other smoke dispersion models can be found in Breyfogle and Ferguson (1996).

The Smoke Management Team at the USDA Forest Service's Southern Research Station in Athens, GA is developing more advanced smoke transport models, the so-called Planned Burn (PB) models (Achtemeier, 2001). PB smoke movement and dispersion models are designed to address the following questions: 1) if I burn tomorrow, will I smoke up a road?; and 2) I just burned, where is my smoke going to be tonight? The PB models address the problem of terrain and landscape features that can cause smoke plumes to diverge, split, merge, and evolve over complex local topography. The models predict smoke movement as a mixture of independent particles similar to smoke actually flowing downwind from a burn site. Smoke that becomes trapped in a localized area gradually dissipates over time. The "bleedoff" of smoke is simulated by allowing the model to occasionally create new smoke particles from existing particles that can then disperse over time (Achtemeier, 2001). One of the major advantages of the PB modeling framework is that it is designed to incorporate highly detailed gridded weather information both at the ground and aloft. This detailed weather information allows the model to account for temporally and spatially varying

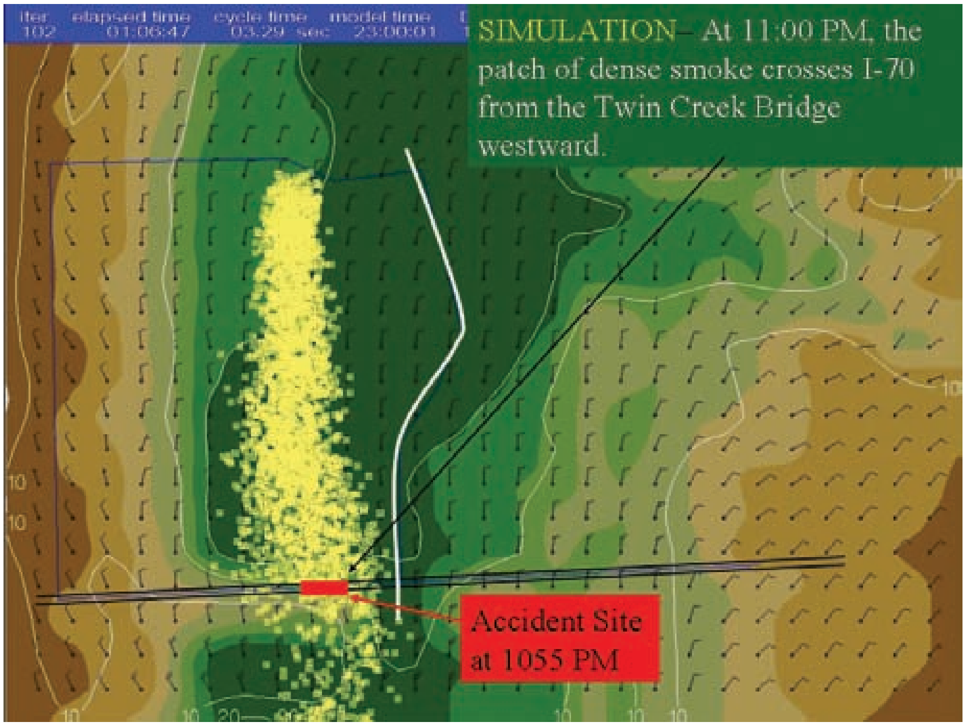

Figure 10.—PB-Piedmont simulation of smoke from burning a pile of Christmas trees at a refuse site near Lewisburg, OH in the Twin Creek Valley between 8:00 AM and 3:00 PM on 7 February 2000. Smoke from this fire, in addition to local dense fog, was implicated as a factor in a vehicle pileup on I-70 at approximately 10:55 PM on 7 February.

wind and stability conditions, and to predict how surface features (terrain, coastlines, etc) impact the smoke plumes as they evolve and disperse.

The first PB model to be developed was PB-Piedmont, which was designed to simulate smoke transport in the complex terrain of the southeastern United States Piedmont region. Two newer incarnations of the PB models have since been developed. PB-Coastal Plain incorporates land use data and land-water information, along with small variations in elevation, to model smoke movement over the lower Coastal Plain. Finally, PB-Mountain is designed to simulate smoke transport in the mountains of the Southeast. The Eastern Area Modeling Consortium (EAMC) is working to implement one or more PB models for the northeastern United States and eastern oak forests, and to make this smoke prediction tool available to fire and smoke managers across the region. Figure 10 shows an example of a one-time use of the PB-Piedmont model to simulate smoke from burning timber in Ohio.

The smoke dispersion models outlined above are generally applicable only to smoke management concerns in the immediate vicinity of a fire. None of the models are designed to address how smoke from fires could affect regional air quality. However, in Section 2 we indicated that state smoke monitoring programs designed to address regional air quality issues will soon begin to impact smoke management decisions throughout the Eastern United States. Currently, there is no well-established tool that can help fire and smoke managers make regional smoke management decisions. However, researchers in the USDA Forest Service's Pacific Northwest Research Station, in cooperation with universities and other research facilities, have developed the BlueSky Smoke Modeling Framework (O'Neill, 2003) to address this need.

BlueSky is a modeling framework that brings the latest state-of-the-science fuels, fire, smoke, and weather models into one centralized processing system. It is designed to provide fire operations and air quality management communities easy access to sophisticated emission, dispersion and weather prediction models and model output. The modeling framework predicts the cumulative impacts of smoke from prescribed fire and wildfire on regional concentrations of PM2.5 and other particulates (Fig. 11).

The BlueSky system was designed as a tool to aide land managers in making go/no-go/go-slow decisions relative to smoke management from prescribed burns. The system produces hourly PM2.5 predictions based on input from the user and 209 wildfire reports. By employing centralized collection and processing of model data, the user does not need to download data and learn to use complex modeling systems. It also allows for analysis of multiple burns and wildfires so that air quality managers can see the combined impacts of prescribed burns within shared airsheds. BlueSky output is posted to the web daily for easy access by burners, air resource specialists, and the public.

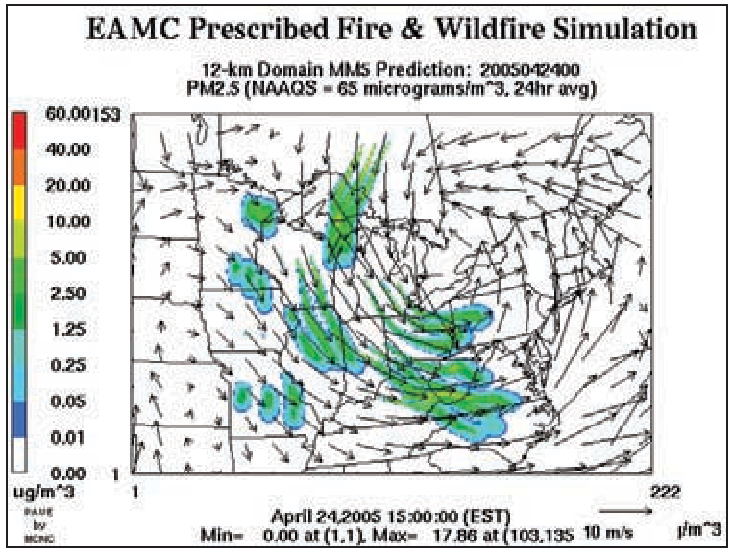

Figure 11.—BlueSky output using the weather conditions on April 24th, 2005 and a hypothetical distribution of prescribed burns across the Great Lakes, Ohio Valley, and southern Canada.

Sample BlueSky output for the Pacific Northwest United States and publications are available at http://www.fs.fed.us/bluesky. The Eastern Area Modeling Consortium (EAMC) maintains BlueSky for the northeastern United States. The user interface for submitting a prescribed burn to the EAMC implementation of BlueSky can be found at http://www.ncrs.fs.fed.us/eamc/products/bluesky/.

CONCLUSION

This paper has outlined the pertinent smoke management questions that fire and smoke managers in the Central Hardwoods will face in the near future. As prescribed fire becomes a more common practice in ecosystem management, the potential for the resultant smoke to impact human health and public safety will have a correspondingly greater impact on burn plans and go/no-go/go-slow decisions on the day of the burn. We have outlined what we see as the main questions that land managers in eastern oak forests should be aware of as they develop forest management plans, as states implement smoke management plans, and as federal regulations that could affect smoke management decisions are being established. Additionally, we have provided an introduction to the existing tools and some of the new tools that are becoming available to aid in smoke management decisions. Clearly, prescribed burn planning in eastern oak forests will continue to become more complex as additional regulations are implemented and sensitivities both in the physical and social aspects of the problem become apparent. By remaining aware of the air quality regulations affecting prescribed burning programs and utilizing the tools available to assist them in meeting those regulations, land managers can be fully prepared for the changes that are certain to come.

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RESISTANCE OF EASTERN HARDWOOD STEMS TO FIRE INJURY AND DAMAGE

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

Abstract.—This paper reviews the protective features and defensive responses of eastern hardwood species exposed to fire. Trees survive fire through protective features such as thick bark and the induced defenses of compartmentalization. Dissection of trees exposed to prescribed fire in an oak forest in southern Ohio highlights the need to distinguish between bark scorch, stem injury, and damage in evaluating the effects of fire on surviving overstory trees.

BACKGROUND

Fires ignited by humans or by lightning are fundamental influences on eastern oak forests. In the absence of suppression efforts, these fires are generally frequent, low in intensity, and patchy in extent resulting in variable exposure of trees to fire (Wendell and Smith 1986; Shumway, Abrams, and Ruffner 2001; Guyette and Spetich 2003). The eastern oak forests of today frequently contain an overstory that is comparatively fire-tolerant and a dense understory of more fire-sensitive and shade-tolerant species (Abrams 1990; Yaussy et al. 2003). In ecological terms, the current overstory is the likely result of regeneration or release of fire-tolerant trees prior to the onset of fire suppression. The current understory is dominated by fire-sensitive species that compete favorably with fire-tolerant species in the absence of fire. Under a scenario of continued fire suppression, the frequency of fire-sensitive species will likely increase in the canopy and remain high in the understory. The lower bole is potentially the most vulnerable part of a tree exposed to surface fires. How do some trees tolerate fire while others are sensitive? This review examines the basic survival mechanisms that reduce injury and damage to trees exposed to fire in eastern oak forests.

ENERGY BUDGETS AND TRADEOFFS

Although difficult to quantify, the concept of competitive advantage in the presence or absence of fire may be readily understood in terms of energy budgets. A budget reconciles outlays among various possible expenditures within the limits of income. An optimal budget strategy meets the needs of necessary expenditure with minimal surplus, leaving more resources available for other categories.

As green plants, trees capture and convert solar energy into chemical energy that is then dynamically allocated to meet the changing needs of the tree, including that of stored energy for future needs. Although strategies for allocation vary among species and settings, all trees need energy for growth, reproduction, protection, defense, and maintenance of the living system. One of the central dilemmas addressed by these strategies is the tradeoff between the allocation of energy to growth or to protection and defense (Loehle 1988; Herms and Mattson 1992). Height growth and a spreading crown architecture promotes energy capture under favorable conditions. Increased stem growth can enhance structural stability and supporter taller, broader crowns. On the other hand, investments such as thick bark and a heartwood that is enriched with wood-preserving chemicals can increase the probability of survival after injury.

One yardstick for describing the strategy for energy allocation is allometrics, the relative biomass distribution among various plant parts such as roots, foliage, wood, etc. Effective strategies promote while less effective strategies reduce the probability of survival. Tree species undergoing competition can have different strategies for energy allocation. These strategies vary among tree species and may also shift as trees increase in size (Jackson, Adams, and Jackson 1999). Trees survive fire

[1]USDA Forest Service, Northern Research Station, 271 Mast Rd., Durham, NH 03824; [2]Rocky Mountain Research Station, Forestry Sciences Laboratory, 800 East Beckwith, Missoula, MT 59801; [†]Corresponding author, 603-868-7624, email: ktsmith@fs.fed.us.

exposure through the (1) constitutive, inborn allocation of energy for protective features that prevent injury and (2) induced allocation of energy for defense that maintains and restores tree health, particularly the continuity of the vascular cambium.

FIRE BEHAVIOR AND THE POTENTIAL FOR TREE INJURY

Patterns of fire injury reflect patterns of fire behavior. Most descriptions of fire behavior and its consequences have been based on observations of western conifers (Miller 2001) although some work has been done in eastern oak forests (Bova and Dickinson 2005). Although crown involvement of overstory trees may occur, eastern oak forests are most often subjected to surface fires fueled by fine or coarse woody debris and low-growing living plants. The direction and rate of movement of a forest fire depends on fuel quality and quantity, topography, and atmospheric conditions, including the prevailing wind direction.

Following ignition, a downwind-burning fire presents a fast-moving front with flames that typically burn the most readily combustible fuel and leave behind much charred and incompletely burned fuel. The upwind or backing fire is a much more slowly moving front that burns fuel more completely. The severity of tree injury is due to both fire intensity (amount of heat generated per length of front) and the duration of the period in which the flames are close to the stem.

As the flames of the heading fire pass a tree stem, flame height increases on the leeward side of the stem (Gill 1974). This increase in height of the standing leeward flame is accompanied by increased residence time and greater stem heating (Gutsell and Johnson 1996). The same process increases heating on the upslope side of trees on uneven terrain as flames and the air entrained into the flames flows past tree stems.

CONSTITUTIVE PROTECTION

Activity by all of the meristematic tissues (shoot and root tips, vascular cambium, bark or cork cambium) results in compartmentation, the arrangement of tree cells into various tissues and organs (e.g. roots, foliage, branches, sapwood, heartwood) with specific roles in maintaining tree function. The unique characteristic of woody plants is the dominant role of the vascular cambium in tree survival and growth. The vascular cambium is the continuous sheath of small, fragile, thin-walled cells located immediately outside of the most recently formed wood and to the inside of the phloem or inner bark. The vascular cambium is the “new cell generator” for both xylem (that matures into wood) and phloem (sometimes referred to as inner bark, the transport tissue for most of the sugar and other biochemicals essential for tree life). An element of compartmentation is the annual growth layer or increment of wood in the boles, branches, and woody roots of temperate zone trees that appear in cross-section as tree rings. Essentially, a new sheath is formed every year over the core of the tree grown in previous years.

As an integrated system, all parts of the tree are essential for healthy functioning. However, we emphasize the survival of the vascular cambium in that it is more difficult to restore or replace than other plant tissues or organs (e.g. roots, foliage, branches). Also because of its anatomical position, death of the vascular cambium (sometimes referred to as “cambial necrosis”) also indicates the death of the phloem located to the outside and death of the sapwood located to the inside of the vascular cambium. In surviving trees, fire kills the vascular cambium and associated tissues by heat conducted through intact bark, rather than from direct combustion. This heat transfer can be modeled using a variety of approaches (Gutsell and Johnson 1996; Dickinson and Johnson 2001; Jones et al. 2004).

The vascular cambium of oaks is protected by a thick layer of bark. Bark is a poor conductor of heat and serves to insulate the vascular cambium from the heat of forest fire. Bark thickness has been used as the most obvious and readily measured protective feature that varies among tree species (e.g., Nelson, Sims, and Abell 1933; Harmon 1984). However, other bark qualities, such as texture and the effect of developmental stage on bark thickness, also affects the effectiveness of bark in protecting the vascular cambium (Gignoux, Clobert, and Menaut 1997).

Traditionally, 60°C (140°F) has been used as the critical threshold temperature that kills the cells of the vascular cambium. This injury is termed cambial necrosis. However, there may be differences in the rate of cambial cell death at elevated temperatures among tree species and season of exposure (Dickinson and Johnson 2004). Cambial cell death also is affected by the duration of exposure to elevated temperatures and the rate of temperature increase (Dickinson, Joliff, and Bova 2004).

INDUCED DEFENSE

Cambial necrosis provides access for the infection of wood by fungal decay pathogens and associated organisms. Injury and subsequent infection induce a cascade of responses within the constitutively compartmented portions of the tree. This process is collectively termed compartmentalization (Shigo 1984). Compartmentalization resists the spread of infection in wood. Compartmentalization is a boundary-setting process that consists of two parts. The first part of compartmentalization occurs in wood present at the time of injury. As living cells are injured, plugs form in the water-conducting cells that resist the spread of further cell death in the sapwood by aeration and desiccation. The plugging is accomplished both through the triggering of formation of specialized structures (tyloses) to block the axial water-conducting system as well as shifts in metabolism to produce waxes, gums, and resins that also restrict dessication and the entry of air into the xylem (Shigo 1984; Pearce 1996).

These shifts in metabolism of wood cells, as well as subsequent infection by fungi, tend to discolor or darken the affected wood. Because of wood anatomy, the spread of cell death and infection tends to occur in columns within the length of the stem. Boundaries are formed by living wood cells that limit the spread of wood cell death and infection to the smallest possible volume. These boundaries that resist the spread of the columns into healthy wood are referred to as reaction zones or column boundary layers. In oaks and other broadleaved trees, these layers frequently contain waterproofing materials similar to those found in bark and phenolic compounds such those found in heartwood and bark.

The second part of compartmentalization occurs in wood formed after injury. The vascular cambium produces an anatomically distinct barrier zone to both sides and above and below the injury. The barrier zone is frequently visible to the unaided eye as a darkened line within or between annual growth ring(s). The barrier zone is most obvious near the wound, but may extend around the tree circumference at some distance away from the injury. The position of the barrier zone within the growth ring allows the estimation of the timing of the fire within or between the growing seasons. In the absence of additional wounds that breach the barrier zone, wood decay will tend to occur within the compartment described by the barrier zone. The compartmentalization of infection and decay enables the vascular cambium to continue to divide and to move outward and away from the infection.

In an otherwise healthy, vigorous tree, the fire injury stimulates cell divisions of the vascular cambium at the edge of the area of cambial necrosis or fire scar, resulting in locally wide growth rings. This local growth stimulation serves to both hasten wound closure and to provide additional strength to the loadbearing structure of the tree. These thickened woundwood ribs serve to efficiently distribute the mechanical loading of the stem, branch, or root and to reduce the risk of structural failure (Mattheck 1998).

TREE DISSECTIONS AND QUALITATIVE ANALYSIS

We dissected eastern oak and associated trees previously exposed to prescribed fire at the Raccoon Ecological Management Area in southeastern Ohio (Yaussy, Hutchinson, and Sutherland 2003; Hutchinson, Sutherland, and Yaussy 2005). Our qualitative analysis (Smith and Sutherland 1999, 2001; Sutherland and Smith 2000) indicated that trees with scorched bark generally occurred in scattered clusters. Individual trees located near discrete fuel sources, such as snags and logs, were also scorched. Dissections showed that the vascular cambium of some scorched trees, particularly those with thick bark, was not injured by the prescribed fires. Thin-barked red maples were more frequently and severely injured than oaks and hickories. In oaks and hickory

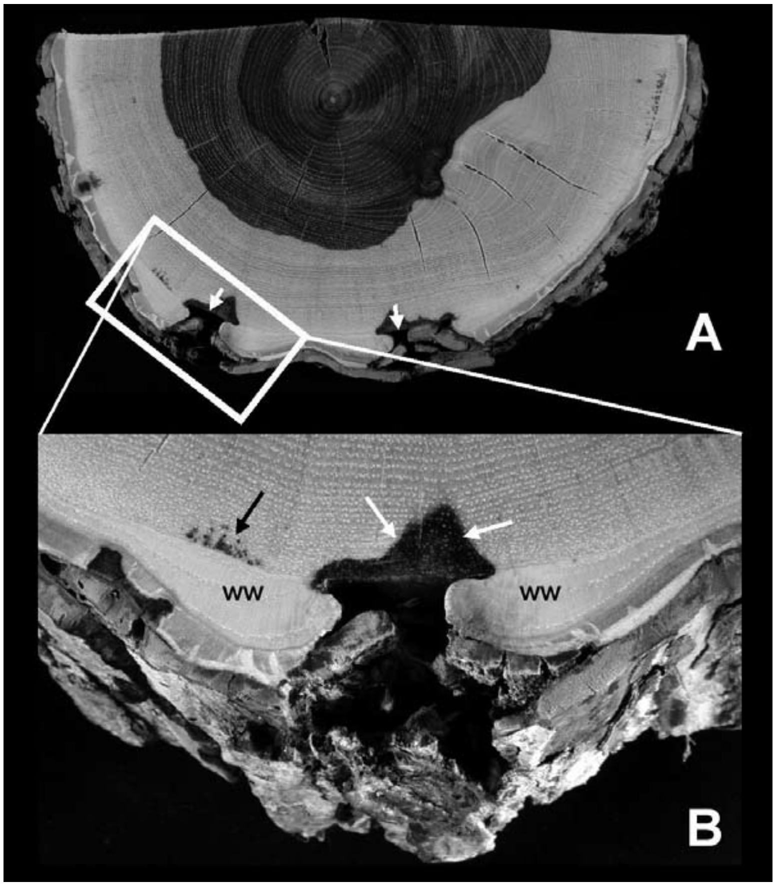

Figure 1.—Cross- section of pignut hickory (*Carya glabra*) with small fire scars covered by intact bark, dissected in the third year after fire injury. A. Cambial necrosis (white arrows) associated with bark furrows. B. The column of wound-initiated discoloration (white arrows) is small and well-defined. Woundwood (ww) appears as locally wide annual rings. The small points of discoloration (black arrow) are likely the ends of small columns of discoloration initiated by a wound several centimeters above or below the plane of the section.

that were slightly injured, cambial necrosis was localized to the areas beneath deep furrows in the bark, where the insulation provided by the bark was least.

Two years after fire injury, few if any overstory trees were killed by the fire. Most fire scars were concealed by intact bark. As with the pattern of bark scorch, these basal fire scars were triangular in shape (Smith and Sutherland 1999). This shape has been attributed to the triangular pattern of heat intensities within and around a flame (Gill 1974). However, basal wounds not caused by fire frequently form triangular scars due to increased dieback of the vascular cambium along the stem axis. Observations of trees exposed to two prescribed fires verify that the scorching or charring of wood in the open wound face resulted from exposure to a later fire and not to the fire that caused the initial scar.

Tree dissections showed that fire scars were often associated with bark fissures, particularly in tree species with thick bark (Fig. 1A, B). The relatively thin bark in the fissure provides less protective insulation to the vascular cambium beneath the fissure (Guyette and Stambaugh 2004; Smith and Sutherland 1999). The surviving vascular cambium adjacent to the fire scar generally produced xylem cells at an accelerated rate, resulting in a fold or rib of woundwood at the scar margin (Fig. 1A, B). This enhanced production of wood was seen as wider growth rings adjacent to the wound. This localized stimulation in growth tends to close over the exposed wood more quickly than at normal growth rates. Successful wound closure reduces the access of exposed wood and associated tissues to infection and allows for the potential restoration of the vascular cambium around the stem circumference.

The formation of woundwood ribs also contributes to the structural stability of the standing tree (Fig. 2A, B). In engineering terms, most of the mass of the above-ground portion of the tree is supported by the outermost portion of the roughly cylindrical stem (Mattheck 1998). Stem growth tends to uniformly distribute the loading

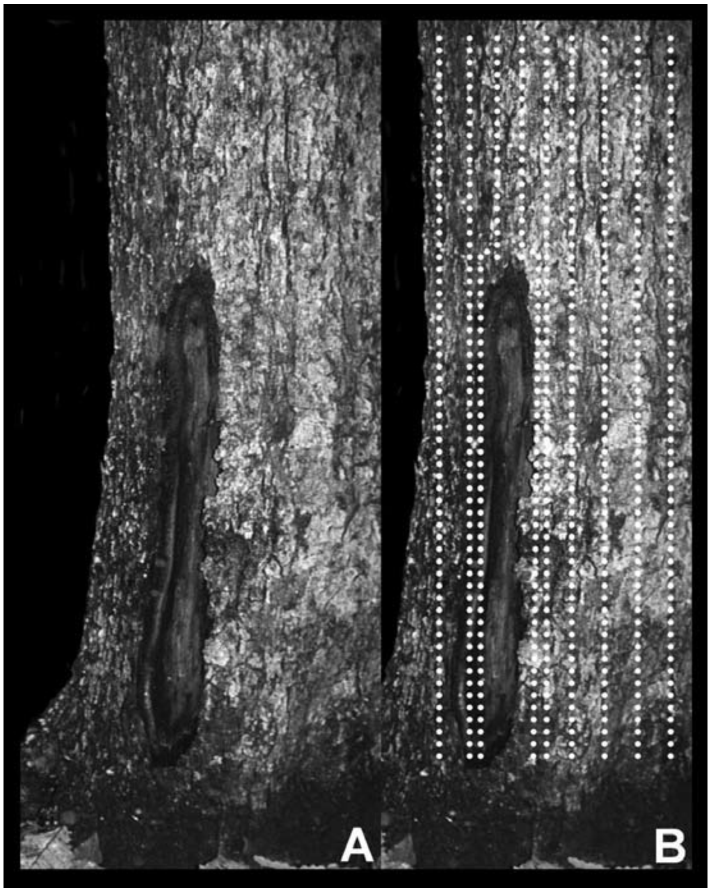

Figure 2..—Oak tree with a 4-year-old fire scar in southeastern Ohio. A. The scar is characterized by the exposed wood face that is almost completely covered by woundwood ribs. B. Simplified overlay diagram showing the mechanical loading of the stem (dotted lines). The decreased spacing of the load lines represents the increased load stress adjacent to the fire scar.

around the stem circumference. Cambial necrosis locally interrupts the formation of the wood annual increment. As the stem continues to increase in girth, the physical loading becomes less uniformly distributed with additional stress applied at the margins of the wound (Fig. 2A, B). The tree woundwood ribs add support where the stress is increased. The curved orientation of wood cells at the inward-facing edge of the woundwood resists shearing stress parallel to the wound surface (Mattheck 1998).

The long-term effect of cambial injury caused by fire illustrates the effectiveness of compartmentalization in resisting the spread of infection and wood decay and contributing to tree survival. From the perspective of compartmentalization, a tree with a well-defined hollow or cavity can indicate a successful outcome (Fig. 3). The cavity is contained within a relatively small volume and surrounded by a continuous band of healthy and structurally sound wood. The vascular cambium is intact and not compromised.

IMPLICATIONS FOR MANAGEMENT

Evaluating the effects of fire on surviving trees, particularly low-intensity prescribed fire, requires an understanding of the distinctions and linkages among signs of a fire (e.g., bark scorch), stem injury (e.g., cambial necrosis, death of associated living phloem and sapwood, and resulting infection), internal discoloration and decay (limited by compartmentalization), and damage (loss of value). The mere presence of bark scorch does not necessarily indicate injury to the vital processes of a tree. Wound-initiated discoloration and decay can, within limits, be contained to small volumes within the living tree. The tree response to extensive injury and infection depends on induced defenses. The effectiveness of compartmentalization to resist the spread of infection depends on the genetic capacity of the tree for compartmentalization and the ability of the tree to use that capacity. That ability rests on the condition of the particular tree at the time of injury (Shigo 1986). The importance of pre-existing tree conditions for survival following fire injury was observed for eastern oaks (Yaussy, Dickinson, and Bova 2004) as was previously seen for northern hardwoods exposed to storm injury (Shortle, Smith, and Dudzik 2003).

The distinction between injury and damage is especially important. "Damage" refers to a loss of value (Shigo 1986). Value is directly related to the specific goal of forest management. If prescribed fire shifts species frequency towards a more desirable stand composition, there may be significant injury to less-desirable species yet little or no damage in the sense of loss of value. Well-compartmentalized infections in the core of desirable tree species may result in a central column of wood decay or a cavity, but have little negative effect on the stem outside of that column, resulting in the potential for high wood quality for products or high habitat quality for wildlife. Depending on the severity of injury and the time required to restore continuity of the vascular cambium, fire can affect the value of overstory trees by decreasing the volume of comparatively more valuable heartwood both by increasing the volume of cull as discolored and decayed wood and by delaying the formation of heartwood (McGinness and Shigo 1975; Phelps and McGinness 1977). The degree of volume loss will depend on the extent of the injury and the effectiveness of compartmentalization.

CONCLUSIONS

Fire affects overstory trees in eastern oak forests in both obvious and hidden ways. Trees resist the occurrence of injury through constitutive protection and resist the spread of infection by induced defenses. Recognizing the patterns of fire effects can help guide prescriptions for the use of fire as a silvicultural tool and to understand the effects of wildfire. Bark scorch, tree injury, and damage are not equivalent terms. Useful evaluation of fire effects on and the implications for overstory trees requires an understanding of fire physics and tree physiology. The degree of damage sustained by fire injury depends on the specific goals of forest management.

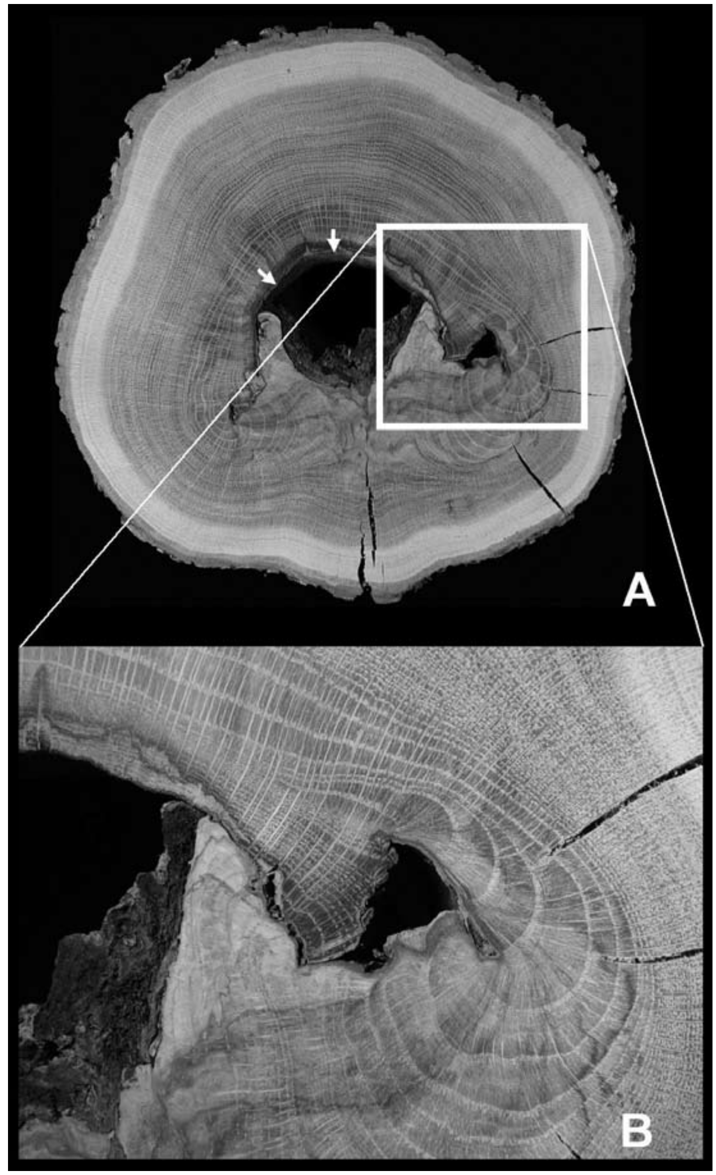

Figure 3.—Cross-section of white oak (*Quercus alba*) illustrating the long-term response to cambial wounding. A. The tree was likely injured at the size and age indicated by the outer edge of the central cavity (small white arrows). Over the next approximately 40 years, woundwood ribs grew on either side of the exposed wound face, eventually restoring the continuity of the vascular cambium. Infection by wood decay fungi and their associates resulted in the complete decay of wood present at the time of injury. However, infection did not spread into wood formed after the injury. B. Changes in orientation of wood cells in woundwood ribs. Bark that lined portions of the inside of the cavity was produced on the outer surface of the ribs as they closed over the wound.

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FIRE AND INVASIVE EXOTIC PLANT SPECIES IN EASTERN OAK COMMUNITIES: AN ASSESSMENT OF CURRENT KNOWLEDGE

Cynthia D. Huebner[1]

Abstract.—Successful regeneration of oak-dominated communities in the Eastern United States historically requires disturbance such as fire, making them vulnerable to invasion by exotic plants. Little is currently known about the effects of fire on invasive plant species and the effects of invasive plant species on fire regimes of this region. Seventeen common eastern invaders were evaluated for their response to fire and potential to change current fire regimes. Twelve species are potentially controllable with repeated growing-season burns (decreasers); five may increase in abundance in response to fire (increasers). Most of the woody decreasers are also potential resisters of fire at maturity. The presence of a seedbank or an outside seed source (evaders) for all but one species and a positive germination response to post-fire conditions (e.g., higher soil temperature, nitrogen availability, and light) make it less likely that most eastern plant invaders can be controlled by fire alone. Shifts in fire regime in eastern oak communities are undocumented but may occur due to changes in community flammability after an invasion. Current fire models are inadequate for predicting fire behavior in these oak communities due to a lack of information on eastern native and exotic plant species fuels. Consequently, fire behavior predictions are best made on a site-by-site basis, especially for sites with multiple invaders composed of increasers and decreasers as well as fire promoters and inhibitors.

INTRODUCTION

The dependence of oak ecosystems on disturbance for regeneration (Loftis 1983; Cook 1998) can result in a vulnerability to invasion by exotic plant species, successful establishment of which often is linked to disturbance (Anderson 1999; Lonsdale 1999; Debinski and Holt 2000; Knapp and Canham 2000; Mack et al. 2000; Buckley et al. 2002). Disturbances that increase understory light (e.g., fire, harvesting, herbicide treatments or combinations of these) are effective in increasing oak regeneration (Dolan and Parker 2004; Miller et al. 2004; Rebbeck et al. 2004). If disturbance were the only important factor determining invasion success, oak systems requiring greater disturbance levels for maintenance would be the most likely to be invaded by exotic plant species. Factors other than disturbance, such as high resource availability (Richardson et al. 1994; Burke and Grime 1996; Higgins et al. 1999; Lonsdale 1999; Stohlgren et al. 1999; Davis et al. 2000; Pysek et al. 2002; Thomson and Leishman 2005) and historic agricultural use (Dupouey et al. 2002, Ramovs and Roberts 2003) also can increase a community's vulnerability to invasion.

[1]USDA Forest Service, Northern Research Station, 180 Canfield Street, Morgantown, WV 26505, 304-285-1501, email: chuebner@fs.fed.us.

For this paper, the Eastern United States includes the Midwest, Great Lakes Region, Ozarks, Appalachian Mountains, Mid-Atlantic, New England, and Southeast. Within these regions are at least five community types with a dominant oak component that may be affected by fire and invasive exotic plant species: 1) mesic forests (mixed mesophytic), 2) dry forests (oak-hickory), 3) oak woodlands/savannas, 4) oak glades/barrens, and 5) oak shrublands (scrub) (Jones et al. 1984; Andreas 1989; Haney and Apfelbaum 1990; Grossman et al. 1998). Definitions for woodland/savanna and glade/barren overlap in the literature, but are separated here because glades/barrens have a more patchy distribution of trees than woodlands/savannas due to less available water or nutrients. Seventeen exotic plant species that are potential invaders of at least one or all of these five oak communities are evaluated for their response to and effects on fire (Table 1).

Fire and Exotic Plant Invasion History

Before 1850, most fires in the eastern region were surface fires set by native Americans. Fire intervals ranged from 1 to 17 years, with shorter intervals in glades/barrens and woodlands/savannas than in forests. There was an increase in frequency between 1850 and 1930 due to European conversion of wilderness areas to farms. Between ~1850 and 1930 (and up to the 1950's in the

Table 1.—Seventeen exotic invasive species organized by habit type. Nomenclature in the table and text follows Gleason and Cronquist (1993) and the Integrated Taxonomic Information System (ITIS; 2005) http://www.itis.usda.gov/servlet/.

Herbaceous Plants
Canada thistle (*Cirsium arvense* (L.) Scop.)
Cogongrass (*Imperata cylindrica* (L.) Beauv.)
Garlic mustard (*Alliaria petiolata* (M. Bieb))
Japanese stiltgrass (*Microstegium vimineum* (Trin.) A. Camus)
Leafy spurge (*Euphorbia esula* L.)
Spotted knapweed (*Centaurea biebersteinii DC*.)
Vines
Japanese honeysuckle (*Lonicera japonica* Thunb.)
Kudzu (*Pueraria montana* (Lour.) Merr. var. *lobata* (Willd.) Maesen & Almeida)
Oriental bittersweet (*Celastrus orbiculatus* Thunb.)
Shrubs
Autumn olive (*Elaeagnus umbellata* Thunb.)
(Russian olive – *E. angustifolia* was used as an example of EHOC value in Table 2 that may correlate with autumn olive)
Bush honeysuckle (*Lonicera* L. spp.)
Data in the text was presented specifically for Morrows bush honeysuckle (*L. morrowii* A. Gray), Amur honeysuckle (*L. maackii* (Rupr.)), and Maxim and Bell's bush honeysuckle (*L*. x *bella* Zabel).
Common buckthorn (*Rhamnus cathartica* L.)
Japanese barberry (*Berberis thunbergii* DC.)
Common barberry (*B. vulgaris* L.) was used as an example of EHOC value in Table 2 that may correlate with Japanese barberry.
Multiflora rose (*Rosa multiflora* Thunb.)
Privet (*Ligustrum* L.spp.)
Chinese privet (*L. sinense*) and common privet (*L. vulgare* L.) had specific research data described in the text.
Trees
Norway maple (*Acer platanoides* L.)
Tree of heaven (*Ailanthus altissima* (Mill.) Swingle)

Ozarks) catastrophic wildfires increased after logging. Before this time, fire-rotation periods generally were less than 10 years, though this varied by community type and region. After 1930-50, fire suppression increased rotation periods to as long as 6,000 years for the Eastern United States (Dey 2002). Despite these changes in fire frequency and rotation, most fires in the East were and remain human caused and often are correlated with drought (Dey 2002; Muzika et al. 2005).

Recorded invasions of exotic plants and other exotic species since 1800 have increased at an accelerated rate, presumably due to increased intercontinental mobility (Liebhold et al. 1995). Similar increases in exotic plant species at a more local scale are evident from herbarium data (Huebner 2003). Although the correlation is circumstantial and weaker for the exotics, there is a noticeable increase in all species collected after the intensive logging and burning phase (1879-1920) in

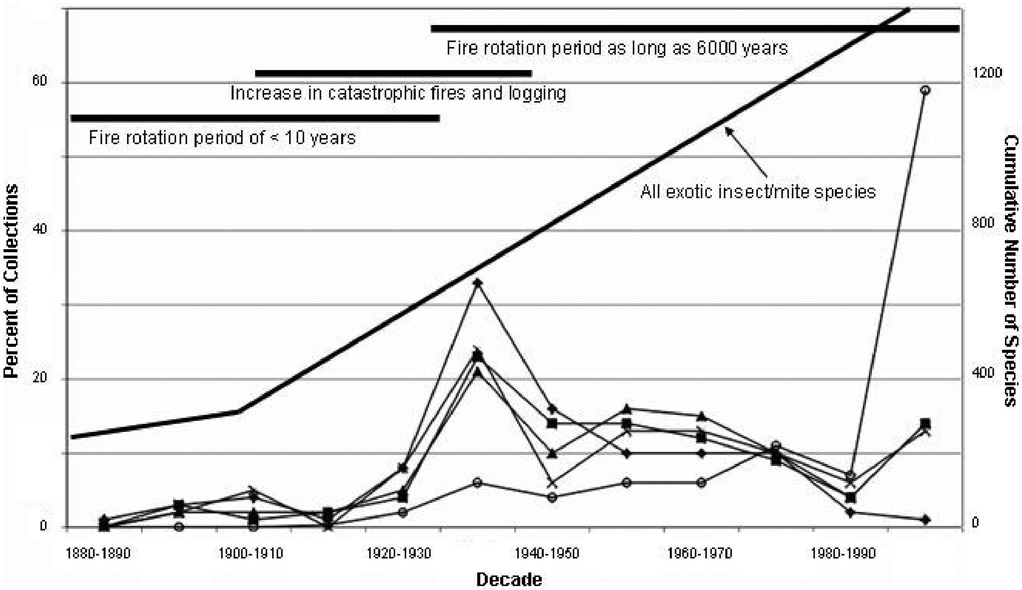

Figure 1.—Time trend in percentage of the total number of species in herbarium collections from West Virginia for nine exotics (o), pteridophytes/gymnosperms (diamond), dicots (square), Asteraceae members (triangle), *Solidago canadensis* (x) (Huebner 2003) overlaid by total number of exotic insect and mite species recorded in the US (Liebhold et al. 1995) and the estimated fire rotation periods in the eastern hardwood forests (Dey 2002). The nine species include three bush honeysuckles (amur, morrow, and tatarica), garlic mustard, Japanese honeysuckle, Japanese stiltgrass, multiflora rose, oriental bittersweet, and tree of heaven.

West Virginia (Clarkson 1964). Thus, while fires in the Eastern United States have decreased since the 1950's, exotic invasive plants have increased since the 1800's, the tenuous connection being human intervention (Fig. 1).

Native Oak Communities' Response to Fire

In general, oak species may benefit from fire because: 1) intense fires followed by low-intensity fires open areas for encroachment by oak species and/or 2) low-intensity fires in forests open the subcanopy or canopy and reduce competition from shade-tolerant species (Lorimer et al. 1994; Miller et al. 2004). Hom (2003) proposed that shrub oak barrens, oak woodlands, and oak forests require fire-return intervals of 15 to 25, 20 to 30, and 100 to 200 years, respectively, for community maintenance. These intervals are inversely proportional to each community's likeliness to burn (Streng and Harcombe 1982).

Spring burns are often more successful at reducing non-oak woody vegetation with epigeal germination (cotyledons are above ground) and slow root-system development that contrast with oak species which generally have hypogeal germination (cotyledons are below ground) and relatively rapid root growth. Fall and winter burns have less effect on woody species, which have translocated much of their energy into below-ground storage (VanLear and Brose 2002; Richburg et al. 2001). Repeated spring and fall burns may increase prairie forb cover in oak woodlands/savannas and oak glades/barrens by reducing competition of grass species, a possible management concern for threatened forbs, e.g., wild lupine (*Lupinus perennis* L.) (Pauly 1997).

Flammability and Fuels of Invasive Plant Species

Flammability has four components: 1) combustibility, 2) ignitability, 3) consumability, and 4) sustainability all of which are influenced, on an individual plant basis, by plant moisture content, percent make-up of carbon, presence of volatile compounds, leaf thickness, and overall surface area-to-volume ratio (Behm et al. 2004). A few comparisons have been made on combustibility of eastern native and exotic plants using effective heat of combustion (EHOC; quantified with a cone calorimeter) (Dibble et al. 2003b). Compared to western fuels' combustibility (14.5 to 21.6 MJ/kg), fuels of both native and nonnative species in the Eastern United States tend to have lower EHOC values (10.9 to 16.0 MJ/kg). These studies also showed that nonnative species had lower EHOC values (10.0 to 14.2 MJ/kg) for combustibility than native species (11.7 to 16.0 MJ/kg; Table 2). The available EHOC data weakly suggest that eastern invasive plant species may tend to reduce fire occurrence; flammability data, using all components, on more species are needed (White et al. 2002). The ability to predict fire behavior in oak communities also requires knowledge of the flammability of the fuel bed. Surface fuels beds are composed of a range of categories, including litter, downed wood, shrubs, grasses, and forbs and are described by their average depth and percent cover and weighted averages of the 1) loading (lbs/ft^2), 2) surface-area-to-volume ratio, 3) heat content (BTU/lb), and 4) moisture content of each sub-class of fuel. Fuel sub-classes are determined by the general fuel category and the size of particles within a category and whether the material is dead or alive (Richburg et al. 2004). Community flammability is determined by multiple, interacting species, and, consequently, is more difficult to measure than flammability of individual species.

Species and community flammability, as well as current weather conditions, are all needed to predict fire behavior at a site. Models like BEHAVE's NEWMDL program (allowing for custom fuel models) may improve predictive capabilities by incorporating data from eastern forests (Richburg et al. 2004). However, it is essential that the fuel bed characteristics of each community type be well-defined. For instance, in woodlands/savannas, the surface-to-volume ratio of fine fuels has a disproportionately large effect on fire spread, whereas in a closed hardwood stand, both the surface-to-volume ratio and the fuelbed depth of fine fuels have a disproportionately large effect on the spread rate of fire (Ducey 2003). In a study with paired invaded and noninvaded mixed hardwood and hardwood stands, the invaded stands had less nonwoody litter, more 100-hr fuels, less duff depth, more graminoid and shrub cover, and less basal area than the noninvaded stands (Dibble et al. 2003a). Thus, while invasive plants may have low individual flammability, they may change a community's flammability such that fire can burn and spread rapidly due to the presence of graminoids, or burn slower and more intensely in the presence of 100-hr fuels.

Exotic Plant Invaders' Response to Fire

Invasive plant species responding to fire can be classified as: 1) <u>evaders</u> -- species with long-lived propagules stored in the soil (or a reliable outside seed source), 2) <u>endurers</u> -- resprouters, or 3) <u>resisters</u> -- species that survive low-intensity fire due to certain adaptive characteristics. These can be classified secondarily as <u>increasers</u> -- likely to increase in abundance after a burn, <u>maintainers</u> -- likely to maintain their current population size after a burn, or <u>decreasers</u> -- likely to decrease in abundance after a burn (Harrod and Reichard 2001). Each of the 17 species evaluated has been categorized on the basis of the available literature (Table 2).

Several nonnative woody species, including autumn olive (Szafoni 1991a), common buckthorn (Solecki 1997; Richburg et al. 2004); Chinese privet (Faulkner et al. 1989); Japanese barberry (Richburg et al. 2004); Japanese honeysuckle (Barden and Matthews 1980; Solecki 1997); kudzu (Radar et al. 1999); Morrow's and Bell's bush honeysuckle (Nyboer 1992; Luken and Shea 2000; Solecki 1997; Richburg et al. 2004); multiflora rose (Szafoni 1991b; Solecki 1997); Norway maple (Simpfendorfer 1989; Maissurow, 1941, based on similarities with sugar maple); and two herbaceous plants, garlic mustard (a biennial) (Nuzzo 1996; Schwartz and Heim 1996) and spotted knapweed (a perennial), can be reduced in community importance after repeated (annually consecutive for at least 2 to 5 years) growing-season (spring to early summer) fires. The data for spotted knapweed, which can resprout after a burn like all of the above species (Sheley et al. 1998), are unpublished[2]. Japanese stiltgrass, an annual, can be expected to decrease in cover after a burn, given no seed source, though there is no direct evidence for this and resprouting is possible (Tu 2000; Gibson et al. 2002). Though all 12 of these species are technically

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[2]Zouhar, K. 2001. Centaurea maculosa. In: Fire Effects Information System. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. Available: http://www.fs.fed.us/database/feis/. Accession date: October 20, 2005.

Table 2.—EHOC values, colonizer type, response to fire, and possible effects on fire regimes documented for each exotic invasive plant species. Species are judged based on their response to repeated spring fires. na = no data; ? indicates no literature is available to support the possible categorization.

Species	EHOC (MJ/kg)	Colonizer type after fire	Population response to fire	Promote or inhibit fire
		Herbs		
Canada thistle	na	Endurer	Increaser Evader	Promote?
Cogongrass	na	Endurer	Increaser Evader (off-site seed only)	Promote?
Garlic mustard	na	Evader	Increaser	Neither? Decreaser
Japanese stiltgrass	11.14	Evader	Increaser Decreaser	Promote?
Leafy spurge	High (no value)	Endurer Evader (but hot fires kill seed)	Increaser	Promote?
Spotted knapweed	na	Evader	Increaser Decreaser	Inhibit?
		Vines		
Japanese honeysuckle	14.34	Evader (but poor seed production) Resister	Decreaser Maintainer	Promote? (as ladder fuel) Inhibitor?
Kudzu	na	Evader Resister (older individuals)	Maintainer Decreaser (younger plants)	Inhibit? Promote? (as ladder fuel)
Oriental bittersweet	11.44	Endurer Evader (off-site seed only)	Increaser	Inhibit? Promote? (as ladder fuel)
		Shrubs		
Autumn olive	na (13.42) (Russian olive)	Evader (off-site seed, possible seedbank)	Increaser Decreaser	Neither?
Bush honeysuckle	na	Evader (off-site seed only)	Increaser Decreaser	Neither?
Common buckthorn	na	Evader (off-site seed, possible seedbank) Resister (older individuals)	Increaser Decreaser Maintainer	Neither?

continued

Table 2.—continued.

Species	EHOC (MJ/kg)	Colonizer type after fire	Population response to fire	Promote or inhibit fire
Japanese barberry	na (14.04)	Evader	Increaser	Neither?
	(common barberry)	(off-site seed source, possible seedbank)	Decreaser	
Multiflora rose	12.30	Evader	Increaser	Neither?
				Decreaser
Privet	na	Evader	Increaser	Neither?
		(off-site seed only)	Decreaser	
		Trees		
Norway maple	11.11	Evader	Increaser	Inhibit?
		(off-site seed; seedbank possible but unlikely)	Maintainer	
		Resister	Decreaser	
		(older individuals)		
Tree of heaven	na	Endurer	Increaser	Neither?
		Evader		
		(off-site seed only)		

endurers, they may be considered decreasers because of the evidence that their underground reserves and ability to sprout may be insufficient to withstand repeated growing-season burns. However, this evidence is not limited to oak-dominated communities; some studies were conducted in pine plantations (Japanese honeysuckle), prairies (spotted knapweed) and pastures (multiflora rose). Also, in the case of garlic mustard, there is evidence of increased garlic mustard abundance after repeated growing season burns in oak systems (Nuzzo et al. 1996; Luken and Shea 2000). The absence of a seedbank or outside seed source of all these species would ensure their status as decreasers or maintainers (Table 2).

Conversely, fires may be ineffective against large, mature shrubs of common buckthorn (Solecki 1997) and against large, woody individuals of any of these species. Similarly, kudzu can maintain a high water content due to its deep taproot so it may not experience topkill during a growing season burn (Winberry and Jones 1973). Tissue moisture content of Japanese honeysuckle (Slezak 1976) and Norway maple (Horvitz et al. 1998) may also be resistant to some fires. Thus, depending on age and size, autumn olive, common buckthorn, Japanese barberry, Japanese honeysuckle, kudzu, the bush honeysuckles, multiflora rose, privet, and Norway maple may be classified as resisters and maintainers.

Several species with deep and/or extensive underground root reserves, including oriental bittersweet (Dreyer et al. 1987), tree of heaven (Lepart and Debussche 1991), Canada thistle (Adams et al. 1987; Solecki 1997), leafy spurge (Cole 1991), and cogongrass (King and Grace 2000; Grace et al. 2001) may respond positively to repeated growing-season burns by increasing in abundance due to an ability to root sucker or sprout, keeping in mind that the research is not directly related to fire response (oriental bittersweet and tree of heaven) or took place in a pine plantation (cogongrass) or prairie/grassland (leafy spurge and Canada thistle). These species can be considered potential endurers and increasers after a fire (Table 2). Although evidence is stronger that Canada thistle responds positively to fire, some studies show it decreasing in abundance after summer or fall burns (Kirsch and Kruse 1973; Hogenbirk and Wein 1991). Seedlings of cogongrass respond positively to a burn and this grass is likely to respond to fire like other perennial warm-season grasses. However, its growth response to multiple fires in different seasons is unclear (King and Grace 2000; Grace et al. 2001).

Seed germination of kudzu often is promoted after a fire due to scarification (Takahashi and Kikuchi 1986; Susko et al. 2001). Garlic mustard (Hintz 1996; Byers and Quinn 1998), Japanese stiltgrass (Barden 1987; Anderson and Schwegman 1991; Anderson et al. 2000; Glasgow and Matlack 2005), multiflora rose (Szafoni 1991b; Kaye et al. 1995; Leck and Leck 1998; Luginbuhl et al. 1999; Glasgow and Matlack 2005), spotted knapweed (Davis et al. 1993), Canada thistle (Kellman 1970; Turner et al. 1997), and leafy spurge (Wolters et al. 1994) all have seedbanks and may respond positively to fire (or, rather, the resulting removal of litter and increases in soil surface temperature and nutrients) with respect to on-site seed germination. Thus, these species are potential <u>evaders</u> and <u>increasers</u> (Table 2).

Seed banking for tree of heaven (Kowarik 1995, but see Kostel-Hughes and Young 1998), oriental bittersweet (Van Clef and Stiles 2001; Ellsworth et al. 2004), bush honeysuckles (*Lonicera maackii* (Rupr.) Maxim.; Ingold and Craycraft 1983), privet (*Ligustrum sinense* Lour. and *L. vulgare* L.; Panetta 2000; Shelton and Cain 2002), cogongrass (Grace et al. 2001), and garlic mustard (Baskin and Baskin 1992) is minimal and most establishment by seed after a fire is by off-site dispersal. It is uncertain whether Japanese barberry, autumn olive (Katz and Shafroth 2003), common buckthorn, and Norway maple (Hong and Ellis 1990) have seedbanks. Tree of heaven seed is dispersed as far 200 m by wind (Graves 1990; Kota 2005); Norway maple also is wind dispersed but distances are not known. Oriental bittersweet, autumn olive, and multiflora rose have been dispersed as far as 500 m by starlings (LaFleur and Rubega 2005), while Japanese barberry seeds have been dispersed at least 80 m by birds (Silander and Klepeis 1999). Bush honeysuckle and common buckthorn also are dispersed by birds but distances are not published. If outside seed sources exist, all of these species would be potential <u>evaders</u> and <u>increasers</u>. There is little evidence of Japanese honeysuckle having a seedbank and this species is a comparatively less prolific seed producer due to pollinator limitations (Larson et al. 2002) and low fruit viability (Haywood 1994). Thus, establishment by seed of Japanese honeysuckle after a fire may be unlikely.

A species' population response to fire may differ from its individual response. For instance, a patchy forest burn likely only removes portions of a large garlic mustard population. Because the population growth of this species is density dependent (Meekins and McCarthy 2002), a large population of garlic mustard likely will show a positive growth response after a patchy burn due to a decrease in intraspecific competition.

Potential to Change Current Fire Regimes

There are no specific data showing shifting fire regimes in the Eastern United States for the 17 species evaluated. Leafy spurge may alter fire intensity because of the high oil content in its leaves (Davis 1990). However, fires with this species present may become hot enough to kill leafy spurge (and native species) seedbank seeds (Wolters et al. 1994). This could halt leafy spurge regeneration if sprouting were unlikely (Brooks et al. 2004). Canada thistle, Japanese stiltgrass, and cogongrass can build up fine, continuous fuels such that a rapidly moving, even spreading fire is likely should a burn occur (Barden 1987; Hogenbirk and Wein 1995; Rice 2004). Such species may cause similar changes in fire regime as was found with the burn and regeneration cycle of cheat grass (*Bromus tectorum* L.; D'Antonio and Vitousek 1992; Brooks et al. 2004). The mesic habitats that Japanese stiltgrass tends to invade and the low flammability of this grass (compared to western species) may make such a regime change unlikely.

If a community is invaded by vines, such as oriental bittersweet, the vine may act as a ladder fuel that could result in a rapidly spreading canopy burn under ideal climatic conditions. Spotted knapweed tends to increase the patchiness of the grassland communities it invades, making it more difficult for such communities to carry a fire (Sheley et al. 1998). Also, if shrubs are invading a community that was not previously dominated by shrubs, there may be a subsequent decrease in fine fuels and a reduction in the community's ability to carry a fire, though any fire that does occur may be more intense. If the community is a savanna or barren, the latter may result in the loss of that community type over time. However, such resource-limited communities may not be invaded as easily. Japanese honeysuckle (Slezak 1976) and Norway maple (Horvitz et al. 1998) may promote an increase in shade-tolerant species by shading out intolerant species, which, in turn, may reduce the likelihood of a burn. These species would be similar to the Brazilian pepper tree (*Schinus terebinthifolius* Raddi.), which retains high moisture in its leaves and litter and has reduced the fire frequency in once pyric pine rocklands (Gordon 1998).

CONCLUSION

If an invaded site is being managed for oak regeneration, the repeated burning required to control certain invasive species, e.g., bush honeysuckle and privet, may have a detrimental effect on oak, e.g., northern red oak (*Quercus rubra* L.), and other native species in the more mesic communities. However, such burns may be required to maintain certain xeric forests, woodland/savanna and glade/barren communities, though repeated burns might promote the invasion of other nonnative species, e.g., leafy spurge and Canada thistle.

We still lack adequate data with which to discern reliable patterns. The comparatively well-studied species, garlic mustard and Canada thistle, have variable responses to fire. As additional research is conducted on these 17 and other species, it is likely that the patterns discussed here will become more complex and variable. Such variability probably will result from differences in site conditions (community flammability, species flammability, resource availability, land-use history) before and after a burn (which are often unknown or difficult to measure) as well as each invasive plant's population age structure and size just before a burn (both of which also are often unknown and difficult to measure). Moreover, many of these species occur together and joint effects on how a site will burn may differ from individual effects. Management of communities with multispecies invasions in which multiple increasers and decreasers as well as promoters and inhibiters might be present may be of such a complex nature that a site-specific approach always is required.

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Rx FIRE LAWS: TOOLS TO PROTECT FIRE: THE 'ECOLOGICAL IMPERATIVE'

Dale Wade[1†], Steven Miller[2], Johnny Stowe[3], James Brenner[4]

Abstract.—The South is the birthplace of statutes and ordinances that both advocate and protect the cultural heritage of woods burning, which has been practiced in this region uninterrupted for more than 10,000 years. We present a brief overview of fire use in the South and discuss why most southern states recognized early on that periodic fire was necessary to sustain fire adapted ecosystems and passed laws to protect and facilitate the practice of controlled burning. We also provide examples of legislation promoting this ecological imperative and share ideas we have found helpful in getting the "right" people involved to assure passage of such legislation. Finally, we discuss constraints to prescription fire that need to be mitigated, as well as items to consider including in pro-fire legislation.

INTRODUCTION

We believevthe Florida Prescribed Fire Act (Appendix A) is the best such statute in the country. If you participate in crafting a bill to safeguard the intentional use of fire in your area, we suggest you use the Florida act and accompanying rules as your starting point and modify them as necessary. That is the easy part; the real challenge is to get enough of the "right" people actively involved to sway public opinion and assure passage of your bill. The key to such an endeavor is an underlying faith that well-informed people will make well-informed choices. Because we have such faith, we weave our message and suggestions into a tapestry of fire and fire-use history in the hope that we will motivate you to action.

FIRE USE IN THE SOUTH

In the South, the ecological fire-return interval is shorter than almost anywhere else in the Nation, the result of both natural and anthropogenic fire. The incidence of cloud-to-ground lightning is higher in the South than in any other region of North America, and is responsible for many wildland fires. But anthropogenic fires are of at least equal importance. "*By about 11,000 years ago, the Paleo-Indians and their fires had traversed all of the New World from Alaska to the tip of South America*" (Stowe 2004a). Over the ensuing centuries, Native Americans continued to learn new ways to improve their standard of living through the use of fire. Henry Lewis (1973) listed 70 reasons Native Americans burned the land while Williams (2000) grouped their reasons for burning into 11 categories. The pattern of occasional higher intensity, wind-driven fires and severe-drought fires that was superimposed on chronic lightning and Native American fire regime shaped and maintained vast southern prairies, savannas, open woodlands, and canebrakes, according to early European explorers. The journals of many of these explorers also mentioned numerous smoke columns and extensive smoke and haze that often lasted for days (e.g., deLaudonniere 1587). Into this environment came European—more than 75 percent with a pastoral background—(Owsley 1945) and African settlers. Both groups brought a knowledge of fire that they merged with that of the aboriginal American residents. Thus, the existing fire regime was expanded and reinforced.

The region's endemic flora and fauna thus became even more intermingled with and dependent on fire (Martin and Sapsis 1992). Eldredge (1911) described the turn-of-the-century fire situation in north Florida: *"The turpentine operator burns his woods and all other neighboring woods during the winter months, generally in December, January, or February. The cattleman sets fire during March, April, and May to such areas as the turpentine operator has left unburned. During the summer there are almost daily severe thunderstorms, and many forest fires are started by lightning. In the dry fall months hunters set fire to such "rough" places as may harbor game. It is only by chance that any area of unenclosed land escapes burning at least once in two years."* This pattern typified

[1]The Rx Fire Doctor, LLC, PO Box 491, Hayesville, NC 28904-0491; [2]Division of Land Management, St. Johns River Water Management District, 4049 Reid Street, Palatka, FL 32177; [3]Heritage Preserves, South Carolina Department of Natural Resources,1000 Assembly Street, Columbia, SC 29201; [4]Florida Division of Forestry, 3125 Conner Boulevard, Tallahassee, FL 32399-1650; [†]Corresponding author, email: rxfire@ix.netcom.com.

the situation throughout the region, and although not everyone was in agreement with this ubiquitous use of fire, the survival of rural families depended on it (see Komarek 1981). So long as people accepted fire and its inevitability on nature's terms, the precarious balance between humans and this natural force was maintained. Rural southerners realized that fire exclusion would tip this balance, destroying both ecosystems and attendant wildlife, while significantly increasing the likelihood that they would lose their homes and even their lives to uncontrollable wildfires that follow attempted fire exclusion.

When it became obvious by the late 1920's that rural residents of the South were openly ignoring state and federal laws banning fire, the USDA Forest Service sent in the Dixie Crusaders (Schiff 1962; Jacobs 1978) to preach the benefits of fire exclusion and the evils of woods burning. Sociologists and psychologists such as John Shea (1940) also were sent with orders to "*find the inner-most reason why inhabitants of the forest lands of the South cling persistently to the custom of burning the woods*" so that a more effective fire prevention campaign could be mounted and finally put a stop to "*the annual fires that for more than a century have characterized the land and social economy of our southern states.*" Shea (1940) thought the nearly unanimous explanation given by people he interviewed "*that woods fires kill off snakes, boll weevil*(s) *and serve other economic ends are something more than mere ignorance. They are the defensive beliefs of a disadvantaged culture group."* History proved Shea wrong but the point is that it is only natural for outside experts as well as uneducated and misguided but otherwise well-meaning people to attempt to "correct" customs and actions that they do not believe are in the best interests of those who practice them. Such customs and actions may require adjustment periodically in accordance with John Dewey's "reflective morality" (Frankena and Granrose 1974; Stowe 1997), but attempted "force feeding" usually promotes only rebellion.

When you have an objective that requires the help of people with different opinions, your agenda more than likely will be advanced if you keep an open mind, state your views clearly, listen and internalize what you are told, put yourself in their shoes, and, if you still disagree, craft your rebuttal from their point of view. This process is much more likely to yield a workable solution than if you "talk down" to people, or relentlessly harangue them with your message. More extensive discussions of southern fire history can be found in Komarek (1981), Pyne (1982), Wade and others (2000), Johnson and Hale (2002), and Stowe (2004a, b). Putz (2003) is a must read for anyone attempting to understand the culture of woods burning in the South; he provides a humorous but accurate account of why historic landscapes have been maintained in many areas of the South despite federal and state laws that once banned the practice of burning the woods. Future laws that ignore or ban local traditions and culture likely also will be ignored. On its website, the Florida Division of Forestry puts it succinctly: "*Removing the option of controlled burning does not stop the burning, it just removes the control.*"

Many people questioned the southern practice of woods burning and devised experiments to study this phenomenon, which led to the science of prescribed fire. These studies and observations, many by government land managers and by Herb Stoddard of the Tall Timbers Research Station (See Komarek 1981), moved the science forward until more acres are prescribe burned in the South today than in the rest of the nation combined. Wade and others (2000) estimated that more than 3 million acres of forest land and 3 million acres of pasture in the South are treated with prescription fire each year, down about 2 million acres from the 1970's (Wade and Lunsford 1989). Haines and Busby (2001) reported that slightly more than 4 million acres of southern pine were burned annually between 1985 and 1994. The 13 states in the Sourh comprise more than 534 million acres, most of which burned periodically a couple of centuries ago, so there is an enormous gap between the acreage that burned historically and the acreage that currently burns. This disparity is a major reason why the South typically has more wildfires than all other sections of the contiguous United States combined (more than 45,000 per year). Although these fires tend to be smaller, the South often has more acreage subjected to wildfire as well, averaging nearly 1 million over the past 7 years (Gramley 2005).

THE NATURAL ROLE OF FIRE

The complexities of combustion aside, fire is nothing more than the rapid oxidation of organic matter, the same basic process as decomposition, albeit at a much faster rate. Wildland fire is a natural process that produces change—a basic premise underpinning the field of ecology. Whether these changes are viewed as desirable or not depends on their compatibility with human values, which typically have little do to with the natural scheme of things. Fire starts, facilitates, accelerates, decelerates, or stops the myriad of natural processes necessary to perpetuate fire-adapted ecosystems (see Wright and Heinselman 1973; Christensen 1977; Wade and others 1980). Sooner or later, fire is required to rejuvenate or maintain most terrestrial ecosystems and the various ancillary loops and cycles that keep them fully functional, though its return interval can range from months to centuries. In-depth discussions of the role of fire in North American ecosystems are found in DeBano and others (1998), and Brown and Smith (2000), both of which contain extensive references.

Fire has been shaping Earth's landscape for millions of years, and has been augmented by anthropogenic ignitions for a long time (See Robinson 1988). Fire not only helped shape and maintain the terrestrial biota around us but it profoundly influenced the progress of humans. Fire was the first tool we learned to use on a landscape level, and our ability to harness this natural force sets us apart from all other animals. The mystique surrounding fire, its ambivalent nature, and its importance misled ancient Greeks and Chinese into identifying this powerful force as one of the four basic elements governing our planet; even today we are still trying to unravel the mysteries of this phenomenon. Luckily, complete knowledge is not a prerequisite to its effective use. Over time, our skill and ability to manipulate the behavior and effects of wildland fire has resulted in significant economic and societal gains.

This is not the place for an in-depth discussion of the economics of prescription fire because the overriding mandatory reason we must protect and facilitate its use is that it is an "Ecological Imperative." That reason aside, prescription fire remains the least expensive method for modifying the landscape and as such always will be popular (e.g., see Dubois and others 2003). Alexander and Thomas (2006) described current costs of fuel treatment, including fire, while Wade and Moss (1999) showed that the periodic use of prescription fire results in a benefit/cost ratio of nearly 2 to 1—adding a return of more than $3 per acre per year for a landowner managing for southern pine. See Dale and others (2005) for a discussion of the need to factor the benefits of fire into wildfire suppression decisionmaking; they used a Colorado case study to show how substantial cost savings accrue from expanding the use of prescription fire. Mason and others (2006) provided an economic assessment that shows substantial net benefits from investments in fuel removals to reduce the incidence of crown fire. The literature contains numerous accounts from throughout the country of wildfires dropping to the ground, causing less damage, and becoming much easier to suppress after spreading into areas where fuelloads had been reduced by a previous fire (e.g., Moore and others 1955; Davis and Cooper 1963; Cumming 1964; Helms 1979; Wagle and Eakle 197;, Martin 1988; Ferguson 1998; Outcalt and Wade 2000).

When trying to measure the success of fire prevention, it is impossible to know how many fires your efforts prevented; the same holds true in determining what a wildfire would have cost to suppress and how much damage it would have caused had it not run into a relatively recent burn. Saveland (1987) used breakeven analysis to demonstrate that "large financial gains" would accrue from the use of prescription fire to reduce fuels in critical areas, and Jonathan Yoder is working on a dynamic economic model for prescription fire that incorporates risk, liability, and timing (e.g., Yoder 2004; Yoder and Blatner 2004).

LEARNING FROM THE PAST

We cannot look into the future to see the results of alternative actions before they are taken, but we can learn from the past. As former Florida State Forester John Bethea said: "*You can no more get to where you don't know where you're going than you got to where you think you are from where you don't know where you've been.*" With that in mind we take a brief look at past fire management and legislative attempts to exclude fire.

By the mid-19th century, steam, and early in the 20th century, the internal combustion engine, both with their tightly-harnessed fire, became commonplace and fire no longer was the only mechanism available to humans that could modify the environment on a landscape scale. Using these new tools, humankind changed the landscape at an awesome pace, but we could not distance ourselves from fire. We created an unprecedented accumulation of debris as we cleared the wilderness, clearcutting and high-grading our way across the continent. Nature's match coupled with our own carelessness with fire, including the plethora of accidental fires resulting from these new inventions (especially steam locomotives, which spewed a never ending trail of sparks from their stacks), led to devastating fires. The result was conflagrations and tragedies like the Miramichi Fire that blackened more than 3 million acres of Maine and New Brunswick in 1825 with a loss of more than 160 lives; the 1871 Peshtigo Fire in Wisconsin, which killed 1,500 people and devastated 1,000 square miles of pine forest in 8 hours; the 1903 Adirondack Fires; the postlogging fires of the Lake States in 1910 and 1918 which snuffed out 400 lives; and the 1910 Idaho and Montana fires that burned 3 million acres (USDA For. Serv. 1954).

Those fires led to a public outcry for change. The answer was obvious, or so it seemed; simply exclude the fire demon from wildlands and forests would at last be free to grow to their full potential, yielding an unprecedented bounty for human enjoyment and use. Although scientists, educators, and land managers recognized the need for suppressing unwanted fires, many also advocated the use of prescribed burning as a hazard reduction measure and fervently warned of the dangers of attempted fire exclusion. Gifford Pinchot (1898), the first Forest Service Chief and the person who implemented the Agency's fire control policy, recognized the need for intentional fire to keep hazardous fuels from accumulating. But it was not to be; once the public spoke in a unified voice, Pinchot acquiesced even though he must have realized that the plan he implemented would not be successful over time. Abraham Lincoln put it, *"Public sentiment is everything. With public sentiment, nothing can fail. Without it, nothing can succeed"* (www.brainyquote.com/quotes/authors/a/abraham/lincoln.html).

Thus began the federal fire exclusion policy and the federal government embraced it with zeal. Many forestry officials in the South recognized the folly of fire exclusion and did not actively endorse it. But despite this initial lack of support, the federal fire exclusion policy was extended to all lands with passage of the Clarke-McNary Act in 1924. This act used the carrot-and-stick approach (See Schiff 1962) that eventually forced all states to join the parade to a war that could not be won.

To make sure everyone is on the same page, fire suppression and fire entirely are completely different concepts. The objective of fire suppression is to extinguish unwanted/ illegal fires quickly in a safe, efficient manner; wildland fire suppression has been and doubtless will continue to be a high priority for all land management agencies; its long-term consequences are reduced damage to ecosystems, reduced financial losses to landowners, and increased public health and safety. Fire exclusion is the attempt to remove fire from the landscape; its long-term consequences are escalating suppression costs, decreased probability of success, increased risk to firefighters and the public, and unwanted ecosystem changes.

Most fire management agencies in the South believe that protecting the public and sustaining ecosystems entails a full-fledged fire prevention campaign, highly trained state-of-the-art fire suppression forces, and the extensive use of prescribed fire. Outside of the South, some states are slowly moving in this direction. For example, in 2005, Michigan joined Florida, Georgia, and Nevada in providing landowners with the highest level of liability protection. The small steps many states are taking to include prescription fire as an agency mandate should be applauded rather than criticized for their slow progress. Such fundamental changes in philosophy should be incorporated operationally in well-planned small steps to build confidence and develop expertise.

The application of fire on the landscape is as much an art as a science because the behavior and effects of fire are microsite-specific and change as burning conditions change throughout the day and year. Many resource managers would rather not use fire because of the risks involved, but they do so because they recognize fire is inevitable and that one is much better off using it under conditions they select as opposed to allowing nature and a temporally unpredictable ignition source to determine its timing. Bob Cooper, the first Project Leader at the Southern Forest Fire Laboratory, said "*fire makes a good servant but a poor master.*" Herbert Stoddard (1961), an early advocate of prescribed fire and the founder of Tall Timbers Research Station stated: "*Fire may well be compared to a two-edged sword which requires judgment, care, and experience to properly handle....*"

THE LEGACY OF FIRE EXCLUSION

On areas where fires had once been frequent, they were easy to extinguish due to a picturesque, low-stature, herbaceous ground cover. But as fuels accumulated on unburned areas and woody brush shaded out grasses and forbs, fires became more difficult to start, though once ignited they became increasingly difficult to suppress. When uncharacteristically high fuel accumulations inevitably did burn, rather than acting as a cleansing or rejuvenating force, fires often became high-intensity and catastrophic conflagrations. Systems that traditionally were perpetuated by fires spreading across the forest floor now faced fires that climbed into and consumed the canopy. Alterations in fire regime can lead to atypical successional pathways and extirpation of flora and fauna (Gill and Bradstock 1995; Glitzenstein and others 2003). Two early researchers, Frank Craighead during World War I in California and Harold Weaver several decades later in Oregon, documented the dramatic difference in forest health between forests burned on a regular basis where insects were endemic and those on surrounding unburned lands where insect populations were epidemic (Weaver 1959; Craighead 1977).

According to Wuerthner (1995): *"No single human modification of the environment has had more pervasive and widespread negative consequences for the ecological integrity of North America than the suppression of fire. Fire suppression has destroyed the natural balance of the land more than overgrazing, logging, or the elimination of predators."* This opinion may represent one end of the spectrum but it should be obvious to people at the other end that the primary byproduct of attempted fire exclusion was the unprecedented and unnatural buildup of fuels on the Nation's wildlands, which contributed to conflagrations capable of killing everything in their path. Vogl (1976) warned us that *"Nature strives to maintain balances. Nonliving and living systems tend to be oriented toward a balanced state. When the existing quasi-equilibrium of these ecosystems and the organisms comprising them is disturbed or upset, feedback mechanisms come into play and phenomenal forces are amassed in the recovery and restoration to stability and balance."* This is exactly the scenario that unfolded with attempted fire exclusion in many sections of the country. Dead fuels accumulated and woody species previously restricted to the ground layer grew to form a dense midstory that shaded out the herbaceous ground cover and provided a pathway for fire to reach the canopy layer. From an ecosystem standpoint, the removal of fire jeopardized the long-term perpetuation of fire-adapted ecosystems. Frances Bacon understood the folly of the sort of thinking behind fire exclusion when he observed that "*Nature is not governed except by obeying her*" (http://www.brainyquote.com/quotes/authors/f/francis_bacon.html). Removing fire also violated principles described by Aldo Leopold (1949) in his *A Sand County Almanac* such that "*The first rule of successful tinkering is not throwing away any of the pieces.*" Fire exclusion amounted to throwing away one of the critical pieces.

Over time, the American public came to understand the unwelcome but inevitable consequences of attempted fire exclusion and realized that we do not have the ability to stop catastrophic fires once they start. When faced with such fires, firefighters have to wait for a change in the weather, a change in topography, or for the fire to run out of fuel. The only practical alternative is to alter the fuels that foster these fires.

Although it became more obvious with each passing decade that the marriage between fire and wildlands could not be split asunder, it took 70 years before the public finally demanded a stop to government efforts to divorce them. Despite the outcry, which resulted in a new federal policy that now includes the use of fire, e.g., U.S. Environ. Prot. Agency (1998), U.S. Dep. Interior and others (2001), USDA For. Serv. (2003), the debate continues as we cope with catastrophic fires and the legacy of this failed exclusion policy, and ponder how enthusiastically we should embrace and regulate the intentional use of wildland fire. Several federal incentive programs promote and facilitate its use, for example, the Healthy Forest Restoration Act, Landowner Incentives Program and Wildlife Habitat Incentives Program. But this encouragement to burn lacks the same zeal or determination once used to promote fire exclusion. Also, these programs often seem to place more emphasis on fuel reduction than on the restoration of a natural process. See McCarthy (2002a) for a discussion of the politics involved.

Ten years after this watershed change in fire policy, it seems that there has been considerable vocal but relatively little operational progress, even in the South. The 13 Southern States contain about 225.6 million acres of forest land, 219 million of which are available for commercial use. This commercial forest land comprises 70 million acres of pine, 32 million acres of mixed pine/hardwood, and 117 million acres in hardwood. About 11 percent or 24.8 million acres of this is government owned (Gramley 2005). We estimate at least an additional 100 million acres, most of which is in Oklahoma and Texas, is in unimproved and improved rangeland/pasture. If Frost's (1998) estimate of presettlement fire regimes is correct, the enormity of the task before us becomes clear. According to Palmer and others (2004), only about 35 percent of state lands and 1 to 5 percent of private forestlands are being burned as often as they need to be for proper fuels management.

Few if any managers have the staff, funding, or full commitment of their superiors necessary to treat all the acres needing fire given the limited number of days with acceptable weather and fuel conditions. One impediment is the lack of incentives for a fire manager to take risks. It is much easier and professionally safer to advocate the benefits of fire but to never have the "right" conditions than it is to shoulder the risks involved with authorizing or conducting a burn. Prescribed fire requires action and with action comes responsibility. Fire exclusion is often a do-nothing-then-react approach that ensures anonymity and protection from the responsibility of taking action. There is always more responsibility associated with taking action than with reacting to outside stimuli. Although the risk of a bad outcome decreases with multiple burns on an area, it never disappears and the law of averages always is at work. Currently, there are few if any incentives for "doing the right thing" other than one's own belief that as a manager "it is my obligation and moral responsibility to future generations to try to maintain healthy, fully functional ecosystems." In the administrative climate of most organizations, including every federal land management agency, the penalties for a bad outcome let alone a mistake can be so severe that the only prudent choice for some professionals is not to burn. When fire is unleashed carelessly or with harmful intent, it can have devastating consequences. Thus, prescription fire carries with it a huge responsibility to use it wisely. Fire managers willing to shoulder this responsibility deserve the protection a statute can offer.

Nationally, we are losing fire experience to retirements faster than it can be replaced. This loss of experience places the entire prescribed fire industry at greater risk because burners with less experience are called upon to burn in a world far more complex than the one in which the "old fire dogs" gained invaluable experience. We cannot afford to exacerbate this problem by discouraging fire managers because the risks become too high for them to bear absent a good prescribed fire law.

Prescribed fire also needs to be protected from those who want to eliminate it for nonscientific personal reasons ranging from air quality issues to a misguided belief that fire is patently bad for plants and animals. If people opposed to the intentional use of fire become organized in the absence of a good law promoting/protecting prescribed fire, this valuable tool can be lost. That came close to happening in Georgia in 1989 when an escaped agricultural burn ignited peat adjacent to Interstate 75 and "smoked in" the I-75 for several weeks while the state legislature was in session. The governor demanded action and a bill was quickly introduced to ban all open burning. Although it was defeated, prescribed burners in Georgia got the message and a bill modeled after the Florida Act was introduced and passed the following year. It occurred again in Georgia 10 years later when the Environmental Protection Agency (EPA) threatened to withhold federal highway funding if Georgia failed to develop a strategy to reduce air pollution in Atlanta resulting primarily from the *100 million miles* driven per day in the area (Leinberger 1998); Pollard 2003;

Fed. Highway Admin. (2003). This megalopolis is the fastest growing human settlement in history and by most measures the most sprawling major metro area in America because Atlanta has chosen to grow out rather than up. Rather than curtailing this metastasis, Georgia demonstrated its unwillingness to make the hard choice and instead restricted open burning in the surrounding 43 counties from May through September. This policy is severely hampering efforts to restore the rare montane longleaf pine ecosystems of northwest Georgia, which require growing-season burns.

The problem with such lack of moral fortitude is that it is contagious. For example, in 2004, the EPA added Augusta, Columbus, and Macon to its list of nonattainment areas in Georgia. Many people were blindsided by the Georgia Environmental Protection Division (GAEPD), which unveiled its plan to appease the EPA by using the same strategy used in Atlanta, namely, the elimination of all open burning during the months of May through September in an 11-county area, which includes a national wildlife refuge, a national forest, and two large military bases, all of which depend on growing-season prescription fires for hazard reduction and T&E species management. Agricultural burns were to be exempted and when GAEPD was asked to explain the reason for the agriculture exemption, officials stated: "*They have a more powerful lobby.*" Further, GAEPD considers the contribution of carbon dioxide from agricultural burning as irrelevant because "*the carbon dioxide released is reabsorbed by crop regrowth in the next growing season*" (Georgia Dep. Nat. Resour. 1999). Using published GAEPD estimates that showed combustion of biomass contributes less than 1/10 of 1 percent of greenhouse emissions in Georgia, compared to 48 percent from petroleum and 44 percent from coal (there are two coal-fired power plants in the Macon vicinity) and that forests sequester more carbon dioxide each year than is produced by all sources of biomass combustion, (Georgia Dep. Nat. Resour. 1999), these proposed restrictions were scrapped and a compromise reached. Because such horror stories are all too common, we must be continually on guard and prepared to intelligently refute such tripe within the short time frame constitutionally allowed before it takes effect. Part of this preparation is to have a prescribed fire statute that makes prescription fire a landowner right and includes language to shield landowners from corrupt and/or misguided officials. Fire councils can demonstrate their usefulness in such situations by providing a prompt, recognizable, and unified voice for prescribed fire.

WHY WE NEED LAWS PROTECTING AND FACILITATING PRESCRIPTION FIRE

- The general public, to a large degree, suffers from the misconception that "all fire is bad" and thus views the intentional use of fire as counterproductive at best.
- Prescribed fire is a resource management tool that benefits the safety of landowners and the public, the environment, and the economy of community, state, and nation.
- There are risks associated with the use of fire:
  - Escapes that can damage another's property.
  - Smoke intrusions that can cause health and safety concerns.
  - These risks can be due to negligence on the part of the burner or due to unpredictable events.
  - Burners should be held responsible for damages when they do not follow appropriate standards, particularly where spelled out in the law.
- Burners should have personal protection from damages to others when appropriate precautions were taken and due caution exercised.
- Burners should be protected from nuisance complaints when they follow appropriate burn execution standards.
- Burners should be rewarded for enhancing their skills and ability to excel in their use of this risky undertaking.
- Without such a law, some resource managers "talked the talk" but continually found excuses not to burn. When nature does it, they are not held accountable.

Good rules to facilitate fire use will make the planning and execution of a burn more efficient and cost effective by eliminating needless and often repetitious paperwork requirements and multiple approvals, by recognizing and rewarding practitioners who strive to increase their knowledge and skills, and by creating a favorable legal environment that provides meaningful burner protection.

Neil Sampson (1995), executive vice president of The American Forestry Association, stated: "*Fire is the most powerful, unpredictable, and potentially deadly tool land managers can use*." A point of clarification is that fire behavior is predictable but the predictions are only as accurate as the inputs, for example, the weather forecast they were based on. See Weiner (1985), Siegel (1986), and Stanton (1995) for a discussion of the legal aspects of prescribed burning. We believe resource managers should be held more accountable for not using fire in fire-adapted ecosystems than for using this "Ecological Imperative" and experiencing a bad outcome that is beyond their control.

Because of the ecological necessity of periodic fire, the cultural heritage of managed fire, and the threat to fire use resulting from an increasing population, southern states have sought legislation to protect the future of prescribed fire as a management tool, but they are not alone. Sun (in press) reviewed the liability burdens placed on landowners who use prescription fire in all 50 States and recent statute changes. He found that all but six state statutes have evolved from "heavy liability burdens" on landowners using prescription fire toward a negligence tort rule (Appendix C), probably due to a resurgence in demand for prescription fire as a management tool. Easy-to-understand descriptions of the differences between strict tort liability, uncertain liability, simple negligence, and gross negligence are also given. Currently, 22 states have legislation protecting/encouraging the use of prescribed fire. Seventeen are east of the Great Plains, including 11 in the South: Alabama, Arkansas, Florida, Georgia, Louisiana, Mississippi, North Carolina, Oklahoma, South Carolina, Texas, and Virginia.

THE HUMAN FACTOR

Given the obvious need to drastically increase the amountof acreage treated with fire, what is the holdup? The simple answer is that wildland fire and people do not mix well and the South has the fastest growing population in the United States. But as Stowe (2004a) pointed out, "*While it is tempting to blame the increasing hassles associated with land management practices on these immigrants, we would do well to remember that many of these folks come from a region with a recent fire history much different than that of the South.*" Immigrants from the North are joined by affluent southern urbanites who want to escape the pollution and closeness of city life for a home in the woods surrounded by nature. Matthews (1992) defined the term "rurbanization" as "*the invasion of affluent urban and suburban-oriented people into rural areas, looking for a self-defined 'country' lifestyle, while importing urban attitudes and values and expecting urban amenities.*" These rurbanites generally are unaware of the hazardous buildup of fuels in their own backyard (Gardner and others 1987). Most view just blackened landscapes as damaged rather than as rejuvenated. Southern states have multifaceted outreach programs to educate new arrivals and their children on the biological necessity of wildland fire in the fuel types where they have chosen to live. But when asked about his agency's outreach regarding prescribed fire, Florida State Forester Mike Long lamented: "*They are immigrating faster then we can educate them.*"

Many rurbanites will tell you that they desire intact, fully functional ecosystems, including the judicious use of prescribed fire, but in reality their acceptance often is on a conceptual rather than on an operational level. Most do not like the smell of smoke, being inconvenienced by ashes in their swimming pools, or subjected to detours and delays when traveling because of smoke on the road. This situation will continue to deteriorate so long as people choose to build out rather than up. Many people have a strong desire to live in or at the edge of wildlands. But this places people and their homes among fire-prone vegetation. People living at this wildland urban interface (WUI) do not realize they can, or believe they should have to, reduce fuels around their homes to make them more fire safe; they expect the taxes they pay to provide complete protection even when unnatural, fuel-infested plant communities catch fire. The truth is that when fire enters the WUI, suppression forces have to make hard choices regarding the allocation of resources. Firefighters must allocate resources to evacuate residents and protect unprepared houses, both at the expense of resources that could have been used to combat the spread of the fire. When resources for suppressing fire are curtailed, it grows larger and threatens even more homes. People who choose to live in the WUI must realize that fire protection is not a certainty and that they must shoulder their share of the responsibility for protecting their homes.

Insurance rate structures are used to encourage better driving habits and to encourage homeowners to protect their homes from theft. Similar rate structures could be used to encourage homeowners to shoulder more of the responsibility to reduce the threat of wildfire on their property. If the insurance industry was forced to determine rates based on risk in a particular area rather than spreading the cost of insuring homes in high fire-risk areas across its entire domain, and if homeowners using wood shakes for roofing or siding or plastic soffit vent covers were unable to collect in the event the structure was damaged by wildland fire, we believe that problem would solve itself quickly. Another problem is that disaster loans and low premiums subsidize inappropriate and high-risk construction (Davis 1990).

State fire management agencies are committed to work with the public to make the interface a safer place to live, but they cannot do it alone. As fires in the WUI became more problematic, this issue has been addressed through land use planning, zoning, structural codes, and development design (Irwin 1987; Davis 1990; Rice and Davis 1991; Pumphrey 1993; Haines and Cleaves 1995). But as Monroe (2002) lamented: "*Unfortunately, few of these recommendations are easy to implement after homes are built, and few seem to be heeded during the planning phase.*" On a positive note, this affords an opportunity for local and state governments to help safeguard their constituencies by providing guidance to and requirements for developers, builders, and homeowners. State and local officials should be encouraged to seek federal funding support for fuels mitigation and for the use of federal prescribed fire use teams on nonfederal public and private lands. See Plevel (1997), Monroe (2002) and Long and others (2005) for suggestions.

IDENTIFYING THE RIGHT PEOPLE TO FACILITATE PRO FIRE LEGISLATION

Once the need for a prescribed fire law is recognized and the decision to craft a bill is made, the next step is to get the right people involved. To enhance the chances of success, you would do well to marshal a diverse group of fire practitioners that includes state wildfire and wildlife agencies; all state and federal land management agencies; the structural fire community; community planners; the insurance industry; municipal and county land managers; forest industry; the state forestry association; range management groups; nongovernment organizations involved in land management and restoration such as The Nature Conservancy; private landowners who use fire; and sportsman's clubs and other special interest groups such as the National Wild Turkey Federation. By creating a united front, the team can demonstrate this is a larger issue than any one entity. The team can also send the right messenger, or group of messengers, to meet with people who oppose the proposed legislation. A team also can deflect attacks on an individual member. Southern cattlemen and farmers are major contributors to the acreage burned with intentional fire, but both groups have initially tended to choose not to be covered under prescribed burn legislation because it would be much more restrictive to them. However, the situation may differ in your state, so we suggest you approach such groups because they generally have strong lobbies.

Every restriction to prescribed burning means fewer acres treated. The underlying purpose of a prescribed fire law is to help ensure that the assemblages of indigenous plants and animals that make up our wildlands continue to thrive as fully functional ecosystems. Legislation cannot make it so but it can and should promote, facilitate, and protect the intentional use of fire for the benefit of both private landowners and society as a whole. Current laws and attendant rules and regulations in many states severely curtail the use of prescription fire; in some states, open burning is banned during the day while in others the permitting process is a nightmare (see McCarthy and Foster 2002). When drafting pro-fire legislation, include language that addresses contradictory language in existing law.

Prescribed fire legislation should be drafted so that it both facilitates fire use and allows for adaptive management. We list a number of items/issues that should be considered when drafting pro-fire legislation (Appendix F). We do not want to once again enact fire statutes that dictate a specific course of action only to find later that they were flawed and/or shortsighted. An excellent strategy to assure this does not happen is to make such legislation general and have the statute delegate authority to a state agency to formulate the details through its rule-making process (See Appendix B). Another reason to keep the law general is that support for a broad concept can be developed by reaching out to a diverse group of organizations, but persuading these groups to reach a consensus on the details of such a law is difficult at best. Such a strategy enables the agency to practice adaptive management and make changes as warranted without exposing the statute itself to the fickle nature of legislatures and the political necessities of election year politics. It took nearly a century of fire exclusion to create the current situation and it will take decades to resolve the problem. Conditions will continue to change and new tools will become available.

It is essential that we take action because the no-action approach will not resolve this issue despite what some people want to believe. Vitousek and others (1997) described the necessity of active management well when they wrote: "*There is no clearer indication of the extent of human domination of Earth than the fact that maintaining the diversity of 'wild' species and the functioning of 'wild' ecosystems will require increasing human involvement.*" Howard (1974) believed it is our obligation: "*Man has a moral responsibility to manage nature once he disrupts it.*" That belief was echoed by Dr. Patrick Moore, cofounder and former president of *Greenpeace,* who stated: "*…we have a responsibility/obligation to use our knowledge and experience to keep US forests healthy.*"

There will be people who resist any action taken to manage natural lands. Leopold (1942) identified the root of the problem that causes well-meaning people to fight the very activities needed to perpetuate fully functional ecosystems. He wrote: "*I am convinced that most Americans have no idea what a decent forest looks like. The only way to tell them is to show them.*" There are numerous examples of communities where succession has been stabilized at a particular sere by chronic low-intensity fires, as well as examples of those that have been rejuvenated by having succession set back to an earlier successional stage with less frequent but higher intensity fires. The benefits of the proper application of fire can be seen on these sites which serve as standards we can emulate. But without the continued presence of fire they too will disappear. If we do not begin to use fire on a much grander scale, there will be far fewer of these sites to behold. It would be a shame if we end up with nothing worthwhile to show (Orr 1993). You can contact any of the authors for information on the location of healthy, fire maintained communities.

Consider organizing a fire council prior to trying to enact prescribed fire legislation. Fire councils are excellent forums for disseminating knowledge, and an excellent way to keep prescribed burners current on new research results, emerging technology, training opportunities and other fire-related issues in your state and beyond. A number of southern states have a fire council (Florida has three). If there is no fire council in your area, we urge you to form one (see Miller 1998).

PUBLIC OPINION CAN LEAD TO PRO-FIRE LEGISLATION

The public is confused because it continues to receive mixed messages. On the one hand, resource professionals are trying to educate the public about the benefits of the judicious use of fire and its biological necessity. On the other hand, newspapers and television often focus on destructive fires resulting from years of attempted fire exclusion. The dilemma that faced the Smoky Bear advertising campaign perhaps exemplifies the situation. It's been said this was the most successful ad campaign in history; since 1944 Smokey has convinced three generations of Americans that all forest fires are patently bad. On Smokey's website (www.smokeybear.com/), you now can find a link to a discussion of the intentional use of fire, though it is not well identified. In fact, the word "fire" is removed from the title. Smokey has never said he was wrong or apologized for misleading the public, and his website includes nothing that suggests he misinformed us, so it's no wonder much of the public is still confused. He needs to take a cue from his colleague Mark Trail and fully embrace prescription fire. One reason we are so adamant about Smokey is because it is still commonplace to see his *suggestio falsi* on posters and billboards. It is these falsehoods that create perceptions and opinions. According to the *Wildland [fire] Communicator's Guide* (National Interagency Fire Center, http://www.nifc.gov/preved/comm_guide/wildfire/index.html), which is linked to Smokey's website: *"…these perceptions, often misperceptions, can become inculcated into society to form 'public opinion'. That public opinion influences laws and policies related to wildland fire management."* It is up to all of us to try to educate the public and replace Smokey's flawed message with the truth. But this is not an easy task; for example, two of the coauthors were threatened with jail for putting a driptorch in Smokey's hand a number of years ago.

Our primary message is to follow President Lincoln's advice and educate the public so they also can reach the same conclusion as fire professionals, i.e., that wildland fire is truly the "Ecological Imperative." You will bolster your chances of success by using published scientific evidence and existing pro-fire legislation from other areas to support your case. If you are successful in these outreach efforts, your ultimate goal of getting pro-fire legislation passed will be much easier.

We also need to look inward at our everyday activities to see what we can do to promote fire use. Redford and Tabor (2000) observed: "*Conservation practitioners rarely write about the work they do,*" especially their failures. "*Renewal of funding is contingent upon success. Few have ever been rewarded for anything other than success. Inside this straightjacket we will not achieve effective conservation because we will never learn. Learning requires experimentation and experimentation sometimes means failure. When failure is not tolerated learning will never take place. This situation in which experimentation, failure, and learning are not tolerated is a death spiral for conservation. We and all we are trying to save will not survive if we do not break out of this inward-turning spiral and move into the uneven and unpredictable terrain of a highly self-critical adaptive management approach. Time is short as we try to slow the juggernaut of biotic impoverishment. We cannot waste time trying things that others have tried and found wanting. But we cannot do otherwise unless we all document our failures as well as our successes. What is needed is a 'safe-fail' environment where folks are encouraged to innovate, experiment, and learn; but most importantly to document what has been tried, and what has failed. We suggest that the long term success of conservation depends on willingness not only to admit our failures but to share them as well.*" As practitioners, we need to take the time to record what we have found that works and what does not, and disseminate that information.

Most bureaucrats and elected officials at all levels of government will publicly state that they are against the extinction of ecosystems. Given that fire is the Ecological Imperative necessary for the survival of fire-adapted ecosystems, it is in the best interest of public officials to demonstrate their support for the judicious use of prescribed fire by protecting it as a landowner right and by facilitating its use. South Carolina passed the Heritage Trust Act in 1976 (www.scstatehouse.net/code/t51c017.doc), the first law of its kind that specifically provided for prescription fire. Florida passed a law in 1977 that required absentee landowners to reduce fuel accumulations judged to be hazardous, or reimburse the state for doing so (Wade and Long 1979). The provisions of this statute were embodied in the 1999 changes (Brenner and Wade 2003) to the 1990 Florida Prescribed Fire Act (Brenner and Wade 1992) (Appendix A) wherein Florida followed the example set by Nevada in 1993 and raised its liability standard to gross negligence.

Walt Thomson with The Nature Conservancy in Florida pointed out a shortcoming in the Florida Prescribed Fire Act in that it offers safe harbor to prescribed burners through the standard of care level for gross negligence This essentially dictates that if you are trained and become certified, you are allowed to exercise a lower standard of care. He suggests that "safe harbor" should be gained through other strategies, e.g., levels of exemption or liability caps. He raises a valid point but one that we believe can be handled by the Florida Fire Control Bureau through its oversight of certified burners. For example, it could require certified burners to report all fire escapes that leave the landowner's property or that result in a smoke incident to the Bureau, which would then use established procedures to review the situation and strip the burner of his or her certification if warranted, thereby removing that individual's gross negligence protection.

At the local level, all 67 counties in Florida have passed resolutions supporting prescription fire (Appendix D) and 48 counties have ordinances stating that the use of prescribed fire is a property owner's right. Some counties, e.g., Flagler have ordinances that require property owners to clear "nuisance brush and pine" trees or else reimburse the Division of Forestry for the service; others, such as Sarasota County, have created smoke corridors for prescription burns and require realtors to provide a copy of such to all perspective property buyers within these corridors (Appendix E).

More in-depth discussions of Florida laws pertaining to prescription fire and accompanying administrative rules are found in Brenner and Wade (1992), Wade and Brenner (1995), and Brenner and Wade (2003). Both the Florida and Georgia prescribed burn laws have withstood challenges.

It should be noted that there are encouraging signs regarding wildland fire in other regions of the Nation, especially in educating and enticing landowners to take more responsibility in protecting their property from fire. McCarthy (2002b) described an innovative effort to educate homeowners undertaken by the Lands Council in Spokane, Washington. The nationwide FireWise program (www.firewise.org) and Canadian Partners in Protection (www.partnersinprotection.ab.ca/) are others.

CONCLUSION

Ultimately it will be the public that decides whether intentional fire will be used to manage fire-adapted ecosystems. We all must be proactive in outreach efforts to demonstrate that the public and our elected representatives that prescribed fire is the only practical (and rational) approach, and that given the latitude and legal protection to do so, fire managers have the will and skill to use fire in a safe and effective manner. It is now up to all of us to rise to the challenges ahead and to take action rather than hide from the challenge and risks and be reduced to simply reacting. The cost of no action is simply too great.

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List of Appendices:

A. Florida Prescribed Fire Act

B. Florida Division of Forestry Rules and Regulations pertaining to fire

C. Prescribed fire liability rules retained on U.S. forest land by state

D. Sarasota County, Florida resolution supporting prescribed fire

E. Sarasota County, Florida Notice of Proximity establishing smoke corridors

F. Items to consider when drafting a prescribed fire statute

APPENDIX A: THE FLORIDA PRESCRIBED BURNING ACT. A MODEL STATE STATUTE TO PROTECT AND PROMOTE THE USE OF PRESCRIPTION FIRE.

590.125 Open burng authorized by the division.—

(1) DEFINITIONS.—As used in this section, the term:

(a) "Prescribed burning" means the controlled application of fire in accordance with a written prescription for vegetative fuels under specified environmental conditions while following appropriate precautionary measures that ensure that the fire is confined to a predetermined area to accomplish the planned fire or land-management objectives.

(b) "Certified prescribed burn manager" means an individual who successfully completes the certification program of the division and possesses a valid certification number.

(c) "Prescription" means a written plan establishing the criteria necessary for starting, controlling, and extinguishing a prescribed burn.

(d) "Extinguished" means that no spreading flame for wild land burning or certified prescribed burning, and no visible flame, smoke, or emissions for vegetative land-clearing debris burning, exist.

(2) NONCERTIFIED BURNING.

(a) Persons may be authorized to burn wild land or vegetative land-clearing debris in accordance with this subsection if:

1. There is specific consent of the landowner or his or her designee;
2. Authorization has been obtained from the division or its designated agent before starting the burn;
3. There are adequate firebreaks at the burn site and sufficient personnel and firefighting equipment for the control of the fire;
4. The fire remains within the boundary of the authorized area;
5. Someone is present at the burn site until the fire is extinguished;
6. The division does not cancel the authorization; and
7. The division determines that air quality and fire danger are favorable for safe burning.

(b) A person who burns wild land or vegetative land-clearing debris in a manner that violates any requirement of this subsection commits a misdemeanor of the second degree, punishable as provided in s. 775.082 or s. 775.083.

(3) CERTIFIED PRESCRIBED BURNING; LEGISLATIVE FINDINGS AND PURPOSE.

(a) The application of prescribed burning is a land management tool that benefits the safety of the public, the environment, and the economy of the state. The Legislature finds that:

1. Prescribed burning reduces vegetative fuels within wild land areas. Reduction of the fuel load reduces the risk and severity of wildfire, thereby reducing the threat of loss of life and property, particularly in urban areas.
2. Most of Florida's natural communities require periodic fire for maintenance of their ecological integrity. Prescribed burning is essential to the perpetuation, restoration, and management of many plant and animal communities. Significant loss of the state's biological diversity will occur if fire is excluded from fire-dependent systems.

3. Forestland and rangeland constitute significant economic, biological, and aesthetic resources of statewide importance. Prescribed burning on forestland prepares sites for reforestation, removes undesirable competing vegetation, expedites nutrient cycling, and controls or eliminates certain forest pathogens. On rangeland, prescribed burning improves the quality and quantity of herbaceous vegetation necessary for livestock production.
4. The state purchased hundreds of thousands of acres of land for parks, preserves, wildlife management areas, forests, and other public purposes. The use of prescribed burning for management of public lands is essential to maintain the specific resource values for which these lands were acquired.
5. A public education program is necessary to make citizens and visitors aware of the public safety, resource, and economic benefits of prescribed burning.
6. Proper training in the use of prescribed burning is necessary to ensure maximum benefits and protection for the public.
7. As Florida's population continues to grow, pressures from liability issues and nuisance complaints inhibit the use of prescribed burning. Therefore, the division is urged to maximize the opportunities for prescribed burning conducted during its daytime and nighttime authorization process.

(b) Certified prescribed burning pertains only to broadcast burning. It must be conducted in accordance with this subsection and:

1. May be accomplished only when a certified prescribed burn manager is present on site with a copy of the prescription from ignition of the burn to its completion.
2. Requires that a written prescription be prepared before receiving authorization to burn from the division.
3. Requires that the specific consent of the landowner or his or her designee be obtained before requesting an authorization.
4 Requires that an authorization to burn be obtained from the division before igniting the burn.
5. Requires that there be adequate firebreaks at the burn site and sufficient personnel and firefighting equipment for the control of the fire.
6. Is considered to be in the public interest and does not constitute a public or private nuisance when conducted under applicable state air pollution statutes and rules.
7. Is considered to be a property right of the property owner if vegetative fuels are burned as required in this subsection.

(c) Neither a property owner nor his or her agent is liable pursuant to s. 590.13 for damage or injury caused by the fire or resulting smoke or considered to be in violation of subsection (2) for burns conducted in accordance with this subsection unless gross negligence is proven.

(d) Any certified burner who violates this section commits a misdemeanor of the second degree, punishable as provided in s. 775.082 or s. 775.083.

(e) The division shall adopt rules for the use of prescribed burning and for certifying and decertifying certified prescribed burn managers based on their past experience, training, and record of compliance with this section.

(4) WILDFIRE HAZARD REDUCTION TREATMENT BY THE DIVISION.—The division may conduct fuel reduction initiatives, including, but not limited to, burning and mechanical and chemical treatment, on any area of wild land within the state which is reasonably determined to be in danger of wildfire in accordance with the following procedures:

(a) Describe the areas that will receive fuels treatment to the affected local governmental entity.

(b) Publish a treatment notice, including a description of the area to be treated, in a conspicuous manner in at least one newspaper of general circulation in the area of the treatment not less than 10 days before the treatment.

(c) Prepare, and the county tax collector shall include with the annual tax statement, a notice to be sent to all landowners in each township designated by the division as a wildfire hazard area. The notice must describe particularly the area to be treated and the tentative date or dates of the treatment and must list the reasons for and the expected benefits from the wildfire hazard reduction.

(d) Consider any landowner objections to the fuels treatment of his or her property. The landowner may apply to the director of the division for a review of alternative methods of fuel reduction on the property. If the director or his or her designee does not resolve the landowner objection, the director shall convene a panel made up of the local forestry unit manager, the fire chief of the jurisdiction, and the affected county or city manager, or any of their designees. If the panel's recommendation is not acceptable to the landowner, the landowner may request further consideration by the Commissioner of Agriculture or his or her designee and shall thereafter be entitled to an administrative hearing pursuant to the provisions of chapter 120.

(5) DUTIES OF AGENCIES.—The Department of Education shall incorporate, where feasible and appropriate, the issues of fuels treatment, including prescribed burning, into its educational materials.

APPENDIX B: FL DIVISION OF FORESTRY RULES AND REGULATIONS PERTAINING TO APPENDIX A

5I-2.006 Open Burning Allowed.

(1) Open Burning in General. Authorization must be obtained from the Florida Division of forestry for burns relating to agriculture, silviculture and pile burning on the same day the burn is to take place or after 4:00 p.m. of the previous day. The Division of Forestry will set special requirements for authorizations in order to protect public health and safety, including; on site inspections, restricting wind direction, limiting the burning period, halt or limit burning when fire danger is too high in all, or specific parts of the state, and requiring specific personnel and control equipment on site. Any authorized burn that goes out of compliance, but has not escaped the authorized area will be allowed a maximum of two hours to be brought into compliance by the person responsible. In the event that the Division determines that there is a threat to life, public safety or property, immediate suppression action will be taken by the Division of Forestry.

(a) Daytime Non-Certified Authorizations will be issued for the burning to be conducted from 8:00 a.m. CT or 9:00 a.m. ET and the fire must discontinue spreading one hour before sunset.

(b) Nighttime Non-Certified Authorizations will be issued with a Dispersion Index of 8 or above for the burning to be conducted between one hour before sunset and 8:00 a.m. CT or 9:00 a.m. ET the following morning. Ignition of these fires is authorized up to midnight CT or ET (specific to the time zone where the fire is located), however the fire can continue to spread until 8:00 am CT or 9:00 a.m. ET the following day. If additional time is required a new daytime authorization must be obtained from the Division.

(2) Open Burning for Certified Prescribed Burn Managers (CPBM). (All burning conducted under this section is related to broadcast burning for the purposes of; Silviculture, Wildlife Management, Ecological Maintenance and Restoration, Range and Pasture Management. Open burning authorizations under this section require the Certified Prescribed Burn Manager's certification number be presented at the time of the request, and that a Certified Prescribed Burn Manager be on site for the entire burn.

(a) Prescription. A prescription for the burn must be completed prior to any ignition and it must be on site and available for inspection by a Department representative. The prescription will contain, as a minimum, (unless agreed to in writing locally between the burner and the District or Center Manager of the Division of Forestry) the following:

1. Stand or Site Description;

2. Map of the area to be burned;

3. Number of personnel and equipment types to be used on the prescribed burn;

4. Desired weather factors, including but not limited to surface wind speed and direction, transport wind speed and direction, minimum mixing height, minimum relative humidity, maximum temperature, and the minimum fine fuel moisture;

5. Desired fire behavior factors, such as type of burn technique, flame length, and rate of spread;

6.The time and date the prescription was prepared;

7. The authorization date and the time period of the authorization;

8. An evaluation and approval of the anticipated impact of the proposed burn on related smoke sensitive areas;

9. The signature and number of the Certified Prescribed Burn Manager.

(b) Open Burning Hours

1. Daytime CPBM Authorizations will be issued for the burning to be conducted from 8:00 a.m. CT or 9:00 a.m. ET and the fire must discontinue spreading one hour after sunset.

2. Nighttime CPBM Authorizations will be issued with a Dispersion Index of 6 or above for the burning to be conducted between one hour before sunset and 8:00 a.m. CT and 9:00 a.m. ET the following day. Ignition of these fires is authorized up to midnight, however the fire can continue to spread until 8:00 a.m. CT and 9:00 a.m. ET the following day. If additional time is required a new authorization (daytime) must be obtained from the Division. The Division will issue authorizations at other times, in designated areas, when the Division has determined that atmospheric conditions in the vicinity of the burn will allow good dispersement of emissions, and the resulting smoke from the burn will not adversely impact smoke sensitive areas, e.g., highways, hospitals and airports.

(c) Burn Manager Certification Process. Certification to become a Certified Prescribed Burn Manager is accomplished by:

1. Satisfactory completion of the Division of Forestry's Prescribed Fire Correspondence Course and direct experience in three prescribed burns prior to taking the course or;

2. Satisfactory completion of the Division of Forestry's Prescribed Fire Classroom version of the Correspondence Course and a minimum of managing three prescribed burns prior to taking the course or;

3. Satisfactory completion of the Florida Inter-Agency Basic Prescribed Fire Course and direct experience in three prescribed burns following successful completion of the classroom training. The burns conducted during the training do not count as part of this three burn requirement.

4. Applicants must submit a completed prescription for a proposed certifying burn to their local Florida Division of Forestry office prior to the burn for review and approval, and have the burn described in that prescription reviewed by the Division of Forestry during the burn operation. The local Division of Forestry District Manager (or their designee) will recommend DOF Prescribed Burn Manager certification upon satisfactory completion of both the prescription and required number of burns.

5. In order to continue to hold the Division of Forestry Prescribed Burn Manager Certification the burner must comply with 5I-2.006(2)(d), F.A.C., or Division Certification will terminate five years from the date of issue.

(d) Certification Renewal. A Certified Prescribed Burn Manager must satisfy the following requirements in order to retain certification.

1. Participation in a minimum of eight hours of Division of Forestry approved training every five years relating to the subject of prescribed fire, or participation in a Division of Forestry recognized Fire Council Meeting, and

2. The Certified Prescribed Burn Manager has submitted their certification number for two completed prescribed burns in the preceding five (5) years, or

3. Participation in five (5) burns and have this documented and verified in writing to the Forest Protection Bureau's Prescribed Fire Manager of the Division of Forestry by a current Certified Prescribed Burn Manager, or
4. Retaking either the Prescribed Fire Correspondence Course or the Inter-Agency Basic Prescribed Fire Course.

(e) Decertification. The Commissioner of Agriculture will revoke any Certified Prescribed Burn Manager's certification if they demonstrate that their practices and procedures repeatedly violated Florida law or agency rules or is a threat to public health, safety, or property. Recommendations for decertification by the Division of Forestry to the Commissioner of Agriculture will be based on the Certified Burner Violations—Point Assessment Table, effective July 1, 2003, which is incorporated by reference located at: http://www.fl-dof.com/wildfire/wf_pdfs/CBMpoints.pdf.

APPENDIX C. PRESCRIBED FIRE LIABILITY RULES RETAINED ON U.S. FOREST LAND BY STATE (FROM SUN [IN PRESS])

Strict Liability	Uncertain liability	Simple negligence	Gross negligence
Delaware	Arizona	Alabama	Florida
Hawaii	Colorado	Alaska	Georgia
Minnesota	Connecticut	Arkansas	Michigan
Pennsylvania	Idaho	California	Nevada
Rhode Island	Illinois	Kentucky	
Wisconsin	Indiana	Louisiana	
	Iowa	Maryland	
	Kansas	Mississippi	
	Maine	New Hampshire	
	Massachusetts	New Jersey	
	Missouri	New York	
	Montana	North Carolina	
	Nebraska	Oklahoma	
	New Mexico	Oregon	
	North Dakota	South Carolina	
	Ohio	Texas	
	South Dakota	Virginia	
	Tennessee	Washington	
	Utah		
	Vermont		
	West Virginia		
	Wyoming		
N =6	N =22	N =18	N =4

D. SARASOTA COUNTY, FLORIDA RESOLUTION SUPPORTING PRESCRIBED FIRE (ON FILE WITH SARASOTA COUNTY BOARD OF COUNTY COMMISSIONERS)

RESOLUTION OF THE BOARD OF CUNTY COMMISSINERS
OF SARASOTA COUNTY, FLORIDA

RESOLUTION NO 97-265

RE: RESOLUTION IN SUPPORT OF PRESCRIBED BURNING

WHEREAS, The Environment Chapter of Apoxsee: The *Revised and Updated Comprehensive Plan Policy* 5.5.10. states that Sarasota County is to "maintain and promote rural and natural resource land management practices such as prescribed burning;" and

WHEREAS, The Recreation and Open Space Chapter of Apoxsee: The *Revised and Updated Comprensive Plan Policy* 1.2.5. states that "natural area parks," once acquired, "will be kept in their natural state, receiving maintenance according to normal practice associated with native habitats;" and

WHEREAS, The Florida Legislature enacted Chapter 590.o26, Florida Statutes, that recognized the benefits of conducting prescribed burns; and

WHEREAS, prescribed burning is a critical resource management tool that when properly administered minimizes impacts on air quality, protects public safety and enhances scenic vistas; and

WHEREAS, a wide variety of ecosystems present within Sarasota County, many of which provide habitat for plants and animals classified as endangered, threatened or as species of special concern, require periodic burning to flourish; and

WHEREAS, prescribed burning reduces accumulated fuels and consequently lessens the likelihood and severity of uncontrolled and damaging wildfires; and

WHEREAS, rules governing open burning in Sarasota County require that proper authorization be obtained form the Florida Division of Forestry, Department of Agriculture and Consumer Services and the Sarasota County Pollution Control Division prior to conducting prescribed burns in Sarasota County;

NOW THEREFORE BE IT RESOLVED by the **BOARD OF COUNTY COMMISSIONERS OF SARASOTA COUNTY, FLORIDA**, in public meeting assembled that;

The Board of County Commissions officially supports the use of prescribed burning as a land management tool in Sarasota County.

APPENDIX E. SARASOTA COUNTY FLORIDA COMPREHENSIVE LAND MANAGEMENT PLAN DISCUSSION OF PRESCRIBED FIRE, AND A RESULTANT "NOTICE OF PROXIMITY CONSERVATION EASEMENT" CALLED EXHIBIT A IN THE U.S. 41/BLACKBURN POINT ROAD VILLADE ACTIVITY CENTER SECTOR PLAN NO. 89-02-SP.WHICH PROVIDES SMOKE CORRIDORS THROUGH ADJACENT PROPERTIES WHEN USING PRESCRIPTION FIRE ON DESIGNATED STATE AND COUNTY LANDS.

Environment Chapter
The Sarasota County Comprehensive Plan
Page 2-1

The Importance of Fire as a Management Tool

Very serious degradation of native plant and animal communities occurs when fire is suppressed in fire dependent habitats. Because frequent fire is essential to most native plant communities in Sarasota County, prescribed burning is now used, and has been used for generations by both preserve managers and ranchers to maintain pine flatwoods, scrub prairies and marshes. These plant communities are flammable by nature. When fire is intentionally excluded, fuels in the form of live and dead plant materials buildup, setting the stage for unintentional, often uncontrollable fires. Naturally occurring wildfires potentially threaten homes built in these areas. This is especially true as subdivisions are occurring in the rural and semi-rural areas.
Although residential destruction by wildfire in Florida is not as serious a problem as it is in western states, it is a reality. Since 1980, hundreds of Florida homes have been destroyed by wildfire. As in the recent history of North Port wildfires, these fires frequently start under conditions which are difficult to control; for example, during severe drought years. Prescribed ecological burning maintains native plant communities and reduces hazardous fuel accumulations.

Although an essential practice, prescribed burning must be regulated to minimize the impacts on surrounding land uses and residents. Burning impacts air quality as it releases particulates and carbon monoxide into the air. These impacts can be minimized through proper coordination with the Sarasota County Air Quality Monitoring Program which regulates open burning through Ordinance 72-37, as amended.

There is a growing conflict between rural land practices and suburban land uses. Recent development pressures in the rural areas of Sarasota County seriously threaten the continuance of prescribed burning and, thus, the maintenance of natural areas.

People who live and maintain farming operations in the rural areas may understand the role of fire in Florida's natural areas and may be tolerant of controlled burning. However people from urban and suburban areas may be less tolerant of nearby fire and smoke. Environmental education efforts should help to inform the public that prescribed burning is part of the character of Southwest Florida and Sarasota County, and essential to maintaining native plant communities.

To encourage the practice of prescribed burning and minimize its impacts on surrounding land uses, the County could recognize the concept of smoke management buffer zones; especially in the eastern portions of the County near Myakka River State Park, the T. Mabry, Carlton, Jr. Memorial Reserve, and surrounding ranchlands. A suggested smoke management zone would extend approximately two miles from the boundaries of areas that depend on prescribed burning for land and wildlife management practices. After smoke management zones are outlined, all development proposals (rezoning, DRI, and special exceptions) within these zones would include acceptance of occasional smoke, a pre-existing condition or character of the area, as a condition for approval.

Care taken to plan compatible development in smoke management zones and implementation of the above policy will help insure the future of the County's natural areas which are threatened by encroaching development restricting the necessary management practice of prescribed burning. As part of the management of conservation and preservation areas in urban and semi-rural settings, alternatives to prescribed fire should be researched and encouraged to maintain fire dependent communities.

EXHIBIT A

NOTICE OF PROXIMITY TO OSCAR SCHERER STATE
RECREATION AREA/CONSERVATION EASEMENT

This Notice date this ______ day of ________________________________, 199-, and entered into the public record by _______________________________________and _____________________________, as owners of the property described as:

SEE ATTACHED EXHIBIT I

(Insert description of subject property owned within U.S. 41/Blackburn Point Road Sector Plan No. 89-02-SP)

WHEREAS, it is the intent of this Notice to make known to the public-at-large that the property described in Exhibit "I" attached hereto is located in close proximity to the property known as the Oscar Scherer State Recreation Area/Conservation Easement

WHEREAS, it is further the intent of this Notice to advise potential tenants and purchasers of subdivided property located within the boundaries of the property described in Exhibit "I" attached hereto, that said property is in close proximity to the Oscar Scherer State Recreation Area/Conservation Easement.

NOW, THEREFORE, the general public and those parties specifically purchasing or leasing property within the area described in Exhibit "I" attached hereto are hereby notified that:

1. The subject property described in Exhibit "I" attached hereto is located in close proximity to the Oscar Scherer State Recreation Area/Conservation Easement.

2. This Notice is to further advise potential purchasers or tenants of property described in Exhibit "I" attached hereto that the proximity to the Oscar Scherer State Recreation Area/Conservation Easement may result in said purchasers or tenants being affected by: continuing current resource management practices to include but not be limited to ecological burning, pesticide usage, exotic plant and animal removal, usage of heavy equipment and machinery and other practices as may be deemed necessary for the proper management of the Oscar Scherer State Recreation Area/Conservation Easement.

3. The nature and extent of the effects of the operations of the Oscar Scherer State Recreation Area which shall include: All management practices as contained within the document entitled "Ecological Burn Plan Oscar Scherer State Recreation Area" adopted on April 3, 1990, and which may be amended from time to time.

4. All property owners which take title to property within the boundaries as described in Exibit "I" attached hereto, or tenants who may occupy the premises within the boundaries described in Exhibit "I" attached hereto, shall be deemed to have constructive knowledge of this Notice due to its recordation in the Public Records of Sarasota County, Florida, and further shall be deemed to have consented to said resource practices, including ecological burning, pesticide usage, exotic plant and animal removal, usage of heavy equipment and machinery and other practices as may be deemed necessary for the proper management of the Oscar Scherer State Recreation Area/Conservation Easement by the recording of a Warranty Deed or other instrument of conveyance, conveying the property within the boundaries in Exhibit "I" attached hereto, or by executing an occupancy agreement and delivering same to the owner of property contained within the boundaries of the property described in Exhibit "I", their successors or assigns.

IN WITNESS WHEREOF, the owners have hereunto set their hands and seals this ____ day of ____, 199-.

______________________________ ______________________________

______________________________ ______________________________

STATE OF FLORIDA
COUNTY OF SARASOTA

I HEREBY CERTIFY that on this day before me, an office duly qualified to take acknowledgments, personally appeared ______________________________ and ____________ ____________________, to me known to be the persons described in and who executed the foregoing instrument and acknowledged before me that they executed same.

WITNESS my hand and official seal in the County and State last aforesaid this ____ day of ______________________________________, 199-.

____
NOTARY PUBLIC

My Commission Expires:
(NOTARY SEAL)

093-007

APPENDIX F. ITEMS TO CONSIDER WHEN DRAFTING A STATUTE TO PROTECT AND FACILITATE THE INTENTIONAL USE OF FIRE.

1. Require an authorization or permit to conduct any free-spreading outdoor fire. Require and keep information on every requested burn including the purpose and size of the intended burn
2. Provide help in planning and executing burns
3. Provide standby during a burn as available
4. Clearly state that the intentional use of fire is a landowner right and, because it benefits the environment, society in general, and the economy of the state, it can't be declared a public nuisance
5. Authorize the state agency responsible for fire management (rather than the state agency responsible for air quality) to take the lead in formulating and enforcing smoke management guidelines
6. Not place any arbitrary time of day, day of the week, or holiday restrictions on when burning can occur. Such decisions should be based on measurable environmental conditions.
7. Develop a burner certification program with incentives that give such burners a higher level of liability protection and more leeway in the execution of burns under their direct supervision. Also develop re-certification procedures
8. Consider tax incentives for ecosystems maintained with the judicious use of fire
9. Include language that provides personnel from other organizations and states assisting in fire management operations at least the same level of protection their home organization/state provides in case of an accident
10. Provide wildland fire suppression equipment and more training for Volunteer Fire Departments
11. Supplement equipment that is no longer available from pulp and paper companies
12. Enhance fire prevention and *FireWise* outreach efforts
13. Make it a punishable offence if a landowner is asked to, but refuses to manage his fuels and a fire starts on his property, regardless of the cause, leaves his property, and damages the property of others
14. State unequivocally that suppression forces do not have to attempt to suppress fires on property where the owner has refused to manage fuel loads
15. Invest in continued training for fire staff. Facilitate their attendance at training and refresher courses on agency time, even if out-of-state and reimburse them for expenses
16. Provide funds to keep fire equipment up-to-date and well-maintained.
17. Provide personal protective gear for all fire staff and include language to make all other state agencies with fire staff to do the same. Make its use mandatory on all fires
18. Provide funds to enter into an agreement to have a state meteorologist provide daily fire danger forecasts and timely spot weather forecasts for complex burns upon request. Require and fund training and expose them to operational fires so they can hone their skills. Several southern state forestry agencies have their own forest meteorologist, or pay a neighboring state that has one, to handle their fire meteorology duties

19. Include language to try to protect fire staff and scheduled equipment replacement during tight budgets. Clearly identify fire staff as 1st responders in natural disasters and other emergencies. Train all fire staff on the Incident Command System (ICS) and require its use statewide in emergencies

20. Facilitate formation of a fire council, fund development and maintenance of a web site for it, encourage fire staff to attend and assume leadership roles on agency time, and reimburse them when they do.

21. Encourage the development of incentives to retain experience and to reward people who successfully accept these risks associated with burning.

GROWING A BURNING PROGRAM: CHALLENGES AND OPPORTUNITIES

Rex Mann[1]

Let me emphasize that I only claim to speak with authority about the task of restoring fire to the National Forests. Private landowners have their own special problems in using prescribed fire, size of forest tracts, lack of expertise, etc.

Before we can appreciate the challenges we face in returning fire to the oak forests of the Eastern United States, we must understand the old paradigm that says that "all fire in the forest is bad." We need to understand how the old paradigm came to be, how it was reinforced over the years, and how it still persists in the minds of many if not most of our citizens.

The fact that we had this conference illustrates the change in thinking of land managers and the scientific community on the proper role of fire in our oak-dominated forests. If we could travel back in time to the end of the 19th century and the early years of the 20th century, we might hear debates between fire advocates and fire opponents that sound much like those going on today.

There is ample evidence from early journals, letters, and other historical documents that the European settlers saw abundant use of fire by Native Americans. For example, John Smith commented that in the forests around Jamestown, Virginia, "a man may gallop a horse amongst these woods any waiy, but where the creeks and rivers shall hinder." Andrew White, on an expedition along the Potomac in 1633, observed that the forest was "not choked with an undergrowth of brambles and bushes, but as if laid out in a manner so open, that you might freely drive a four house chariot in the midst of the trees."

In 1630, Francis Higginson wrote about the country around Salem, Massachusetts, that:

> *…there is much ground cleared by the Indians, and especially about (their agricultural fields); and I am told that about three miles from us a man may stand on a hilly place and see thousands of acres of ground as good as need be, and not a Tree on the same.*

In 1637, Thomas Morton wrote that the Indians:

> *…are accustomed to set fire of the Country in all places where they come, and to burne it twize, in the year…as the Spring and fall of the leafe…so that hee that will looke to find large trees and good tymber…(will not) finde them on upland ground; but must seeke for them…in the lower grounds, where the grounds are wett.*

But frequent forest burning did more than reduce the undergrowth and improve the habitat for preferred species. In many cases it created grasslands in areas where forest otherwise would have existed. Prairies extended into Ohio, western Pennsylvania, and western New York. In Virginia, the Shenandoah Valley—a broad valley located between the Blue Ridge Mountains and the Allegheny's—was on vast grass prairie that covered more than 1,000 square miles. Native Americans burned the area annually. R.C. Anderson wrote that the eastern prairies and grasslands "would mostly have disappeared if it had not been for the nearly annual burning of these grasslands by the North American Indians." In the West, as well, Indian burning greatly extended the area of grasslands and reduced the area of forest.

Exactly why the Indians burned is less clear. Most likely they used fire, as did early peoples on all continents, to shape and affect landscapes in ways that made it easier for them to live. Fire was the tool that cleared land for agriculture and created conditions favorable for deer and other wildlife. This included creating patches of blueberries as well as conditions that favored oaks and chestnuts. In other words, the use of fire was closely related to obtaining food. Since the Indians had little use for hardwood sawlogs, you have to believe that firing the

[1]Emergency Management Specialist, USDA Forest Service, 1700 Bypass Road, Winchester, KY 40391, 859-745-3123, email: rmann@fs.fed.us.

woods was considered an act of stewardship rather than an act of destruction.

There also is ample evidence that early Europeans continued the custom of burning the forest, long after the Indian populations had collapsed from exposure to our diseases. This burning was mostly to promote grazing for livestock, including hogs.

In the closing years of the 19th century, the Nation was experiencing an unprecedented wave of destructive logging and land clearing. As Stephen Pyne expressed in "Year of the Fire": "Above all, fire–abusive fire—followed the ax, flames fed on the extravagant wreckage left by logging and land clearing." Steam engines, used in logging, provided sparks for easy ignition of logging slash. Destructive fires occurred in places that seldom have fire problems today, such as New England and New York. Entire towns and villages burned, with significant loss of life. Concurrent concerns about the liquidation of the nation's forests and the widespread fires set the stage for the creation of the USDA Forest Service in 1905 and the establishment of state forestry agencies in the immediate decades that followed.

Our early foresters were educated in Europe, the one place where agriculture and settlement had changed landscapes to the point where fire no longer played a natural role. They brought to America the view that fire had no place in the forest. As early as the 1809's, Bernard Fernow, the Prussian trained forester who headed the Division of Forestry, had denounced America's ceaseless burning as a product of "bad habits and loose morals." Gifford Pinchot, who more than anyone else shaped American forestry, had this to say: "The question of forest fires, like the question of slavery, may be shelved for a time, at enormous cost in the end, but sooner or later it must be faced."

There were voices arguing for continuing the "light burning" or Indian burning that had been practiced for centuries. Major John Wesley Powell, director of the U.S. Geological Survey, had come to believe that the practice of regular burning was a surer method of protecting the land from wildfire than attempting to abolish fire altogether, William B. Greeley Associate Chief of the Forest Service, contemptuously referred to this philosophy as "Paiute forestry."

Then came the fires of 1910, called the Big Blowup, when millions of acres burned and scores of firefighters died. Although largely started from steam engines and logging slash, these fires solidified the Forest Service's position that fire had no place in American forestry. The state forestry agencies and the Department of the Interior also adopted the "all fire is bad" paradigm.

This position on fire was maintained for decades, strengthened by Smokey Bear during World War II. It was only in the piney wood of the South that burning became an accepted practice.

By the end of the 20th century, many of the public lands suffered fire famine. Forests were diseased and dying and prone to catastrophic fire. In the East, species such as the red cockaded woodpecker were becoming more rare and species such as the American chaffseed were disappearing. The uplands had been invaded by fire intolerant species that likely had never grown there before.

American society, no longer rural, had shed its familiarity with fire and now feared it. Urban values had replaced rural values: fire was regarded as an enemy and not a useful tool or a vital process.

So here we are—a century after the Forest Service was formed, a century after the American public was told that fire in the forest was bad, a century after those who argued for "light burning" were ridiculed and dismissed—faced with the need to explain to the public that the founding foresters were wrong, that we now know that fire plays a vital role in sustaining our oak forests.

Of course, hindsight always is perfect. This epiphany that fire is now useful did not occur overnight. A good deal of research has been and is being conducted on the proper role of fire in our forests. Some of the most significant research has been paleo-botanical, where the stories told by ancient grains and charcoal fragments challenged the basic tenets of land managers who for decades had excluded fire.

On the national forests we have finally learned that whatever management program we undertake must have public support. We can do the burning but how do we gain that support?

What are our challenges?

1. Our public is now largely urban and in urban areas fire generally is considered an enemy or a bad thing.
2. The public had little knowledge of fire as a tool, farmers burning fields or pastures, burning brush piles, etc.
3. Our TV sets give us round-the-clock images of horrendous wildfires each summer that burn homes and kill people.
4. We have had some spectacular prescribed fire escapes, beginning with Mack Lake in 1980 on the Huron-Manistee National Forest in Michigan. This fire burned 20,000 acres and 44 homes and buildings; one firefighter was killed. The Las Alamos fires in 2000 destroyed 235 homes and triggered one of the largest and most expensive post-fire rehabilitation efforts in history. Recently, an escaped fire in Florida burned 38,000 acres.
5. The smoke from prescribed fires is ours no matter where it goes. Unlike the Indian burning of centuries ago, we must be concerned about where our smoke goes.
6. There are some who believe lightening fire is good, but human-ignited fire is bad. This belief is rooted in the myths of the pristine forest and the ecologically invisible Indian, in the mistaken belief that North America's First People could not or would not use fire to shape landscapes.
7. There are some who believe that scarring the base of a sawlog tree is morally wrong. Given the variable fire effects we can expect when we install a fire regime in the oak uplands, we know we can strive to minimize this type of damage, yet it is unreasonable to believe we can totally eliminate it.

Here are additional challenges:

1. Will our workforce and budgets allow us to grow this program to the required level?
2. Do we have internal resistance to burning on this scale? Is there significant opposition from Forest Service employees and/or our cooperating agencies?
3. How do we get around the limited burning days we have? Of all of the factors that limit our ability to grow a program, the dozen or so suitable burning days annually seem to be the most severe.

Last year on the Daniel Boone National Forest, we successfully prescribe burned more acreage than ever before—a little more than 19,000 acres. Our new forest plan calls for us to burn 50,000 acres annually by the end of the next decade. We intend to grow this program carefully but steadily by several thousand acres each year. On a forest of more than 700,000 acres, 50,000 acres burned per year on a 3- or 4-year cycle probably is inadequate for restoring and sustaining the forest conditions we are seeking on our uplands.

As stated earlier, of equal importance to growing the program is gaining public support or acceptance for the program. We intend to do this by reaching out to our publics to participate in the actual planning of landscape projects to promote a healthier forest condition. This is a critical point. Our new forest plan is driven by a concern for forest health, not how many board feet of lumber we can produce or how many acres we can burn. On the uplands of the Daniel Boone we will be creating a variety of forest conditions, such as pine and hardwood grassland communities, woodlands, and other habitat types that reduce the overcrowded conditions resulting from 70+ years of fire exclusion. These communities will be treated carefully. Commercial logging will be used to reduce the stocking and remove fire-sensitive species that have invaded the sites. To the degree possible, lands not immediately put into these special conditions will be thinned to reduce overcrowding.

Fire will be reintroduced into these uplands not as an end in itself but as a vital process that maintains the healthier conditions we have placed on the landscape. We are not viewing the reintroduction of fire as a single event, or as several successive burns, but as a permanent recurring process that is vital to sustaining what we believe are healthier, more resilient systems. Along with restoring fire, we intend to restore the American chestnut in partnership with the American Chestnut Foundation, and, hopefully, other lost species. I should say also that our goal is not to eliminate red maple, white pine, or hemlock but to confine these species to the wetter stream bottoms where they historically grew.

We are fully aware that this will not be an easy sell. We also are aware that much of the forest will not be burned, because of what I call social constraints e.g., interstate highways, airports, hospitals, nursing homes, and recreation areas, and our fear of affecting these areas with smoke.

We must educate the public as we involve it. It is critical that we speak of fire as a vital process in a larger effort of creating and sustaining healthier forest conditions. We must discard the old "all fire is bad" paradigm and help the public understand that, like wind and rain, fire is a vital, natural process.

At the implementation level, there are a number of things we must do to grow this program. We are limited by the number of burning days that occur. This means that on a suitable burning day, smoke must rise. Prescribed fire is not necessarily more important than the other jobs we do, but it is time sensitive. We must burn large areas; at least 500 acres in size It takes a day to burn a hundred acres or a thousand acres. We must use aerial ignition because it is so much faster and cheaper. Using ping-pong ball dispensers, we can ignite 1,000 acres in less than an hour. We must consider fall and winter burning and summer burning when our new habitat conditions are in place with a changed fuelbed. We must learn to ignite fire and fight fire at the same time—in Kentucky for example, arson fires are perhaps the greatest obstacles to growing a program. We must investigate the possibility of nighttime burning to extend our burning window.

We must do all this without succumbing to the urge to burn on days when conditions are too severe. The recent history of escaped burns teaches us that the public acceptance of controlled burning is fragile and disastrous escapes can kill a program.

Regardless of the challenges and obstacles, we must move wisely but aggressively to reintroduce fire to the uplands of the Daniel Boone. We recently lost most of our southern yellow pine trees and the northern-most group of red cockaded woodpeckers to a native insect, the Southern Pine Beetle. A major factor in this loss was the overcrowded, stressed conditions of this ecosystem, which had developed after decades of fire exclusion. The same disaster could lie in store for us in our hardwood uplands if we fail to act.

Poster Abstracts

OVERSTORY AND SEEDLING RESPONSE TO REPEATED FIRES IN EASTERN KENTUCKY

Heather D. Alexander[1†], Mary Arthur[2], David Loftis[3], and Stephanie Green[4]

Oak (*Quercus* spp.) regeneration failure throughout eastern deciduous forests has prompted forest managers to use prescribed fire as a potential tool to maintain oak communities. Within the Daniel Boone National Forest of eastern Kentucky, we tested the ability of repeated prescribed fires to increase oak seedling growth and survival by increasing light at the forest floor, decreasing competitor seedling survival, and enhancing oak seedling leaf characteristics. We measured stand structure, canopy cover, seedling survival, and seedling leaf characteristics in areas unburned, burned once, and burned twice. Pre-burn density measurements showed oaks dominated overstory classes but maples (*Acer* spp.) and other species dominated the midstory and understory. In areas burned twice, canopy cover temporarily decreased after the first fire but quickly returned to pre-burn levels after the second fire due to increased sprout density. Burning did not affect red oak (Erythrobalanus) survival, but white oak (Leucobalanus) survival decreased. Among competitor seedlings, burning had no effect on sassafras (*Sassafras albidum* (Nutt.) Nees)) survival, but maples (red and sugar maple combined) had the greatest mortality of all species. Within burned areas, competitor seedlings produced more leaves with greater leaf area than red or white oaks. Burning led to higher specific leaf mass (leaf thickness per unit area) for both oak groups compared to competitor seedlings; this increase was correlated with increased light levels. Repeated fires temporarily opened up the canopy but post-fire sprouting rapidly cancelled this positive effect. Burning also reduced the survival of red maple seedlings, but competitor seedlings that survived the fire grew more leaves and had higher leaf area than oaks. Burned oaks had thicker leaves but whether this leads to increased growth is unknown. These findings suggest multiple, repeated fires with short burn intervals may be needed to maintain canopy cover and sufficient understory competition to successfully regenerate oaks.

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[1]Ph.D. Candidate, University of Kentucky, Forestry Department;
[2]Associate Professor, University of Kentucky, Forestry Department;
[3]Project Leader, USDA Forest Service, Bent Creek Experimental Forest;
[4]Senior Land Management Specialist, Southwest Florida Water Management District; [†]Corresponding author, T.P. Cooper Bldg., Lexington, KY 40546, 859-257-2806, email: heather.alexander@uky.edu

THERMOCOUPLE PROBES IN SURFACE FIRES

Anthony S. Bova[1†] and Matthew B. Dickinson[2]

The maximum temperatures of thermocouple probes (TCPs) exposed to wildland flames often are called "fire temperatures" in forest fire literature, but such temperatures depend on TCP diameter, configuration, and physical properties, as well as flame temperature, dimensions, and velocity over the TCP surface. Thus, *TCP temperatures alone give no useful information about a surface fire*. However, in small experiments, TCP response was calibrated to provide general fire characteristics. For instance, we found that the maximum time-rate of change of TCP temperature and the area under the temperature-time curve correspond to the fireline intensity and fuel consumption, respectively. Calibration equations derived from these data can be used to estimate fireline intensity and fuel consumption from multiple TCPs installed at remote locations in prescribed burns.

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[1]Physical Scientist and [2]Ecologist, Northern Research Station;
[†]Corresponding author, 359 Main Road, Delaware, OH 43015,
740-368-0093, email: abova@fs.fed.us.

WILDFIRE HAZARD MITIGATION TREATMENTS APPLIED ABOVE GROUND IMPACT SOIL ECOLOGICAL PROPERTIES IN OAK FOREST ECOSYSTEMS

Jennifer A. Brinkman[1†] and R.E.J. Boerner[2]

Manipulations designed to reduce wildfire hazard in forests in which wildfire has been suppressed for many decades typically entails manipulating fuels with prescribed fire and mechanical treatment. Although the ecosystem properties being manipulated are above ground, these activities also may affect below-ground components of ecosystems. We evaluated soil organic matter and microbial activity in mixed-oak (*Quercus* spp.) forests of southern Ohio that were subjected to alternative wildfire hazard modalities (prescribed fire, thinning from below, and the combination of thinning and burning) and untreated controls over a 4-year period. During the first growing season after treatment, soil organic C quantity was significantly lower in soil from the thin+burn plots than from the other three treatments, but there was no significant difference in organic matter quality (C:N ratio) among treatments. By the fourth year after treatment, the soils from the thinned plots had significantly greater organic matter quantity and lower organic matter quality than soils from the other treatments. During the first year, net N mineralization increased after thinning and decreased after fire relative to the untreated control. Plots that were both thinned and burned did not differ from the control. By the fourth year, there were no differences among treatments in N mineralization. Proportional nitrification was significantly lower in thinned plots (with or without fire) than in control or burned plots in the first year, but this pattern was reversed by the fourth year. Acid phosphatase activity (an index of microbial breakdown of labile carbon compounds) was lower in the thin+burn plots that in the other three treatments the first year, but by the fourth year, acid phosphatase activity was greater in the thinned plots than in the other treatments. Phenol oxidase activity (an index of fungal breakdown of recalcitrant organic matter) was not significantly affected by any treatment in the first or fourth year after treatment. The responses to the hazard reduction treatments differed among response parameters and years; thus, ordination approaches were used to present a more holistic view of the impact of above-ground manipulations forest-soil ecology.

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[1]Research Assistant, The Ohio State University, Department of Evolution, Ecology, and Organismal Biology; [2]Professor, The Ohio State University, Department of Evolution, Ecology, and Organismal Biology; [†]Corresponding author, 318 W. 12th, 300 Aronoff Laboratory, Columbus, OH 43210 USA, 614-292-1688, email: brinkman.3@osu.edu.

FIRE RESEARCH IN THE PINE BARRENS OF NEW JERSEY

Kenneth Clark[1†], John Hom[1], Nick Skowronski[1], Yude Pan[1], Steve Van Tuyl[1], and Warren Heilman[2]

We are developing an interdisciplinary fire research program based at the Silas Little Experimental Forest in the Pine Barrens of New Jersey to better estimate wildfire risk, quantify the tradeoffs between hazardous fuel reduction treatments and carbon sequestration by forests, and measure the impact of hazardous fuel reduction on emissions of EPA-criterion pollutants near urban nonattainment areas. We are using an integrated network of field plots, fire weather and eddy flux towers, remotely sensed data layers, and validated fire weather and forest ecosystem models. Extensive field plots and LIDAR measurements have been used to estimate fuel loading and the presence of ladder fuels across the Pinelands. In addition, pre- and post-prescribed fire measurements of fuel loading are used to evaluate fuel reduction treatments. These activities provide better estimates of hazardous fuel loading and the cost effectiveness of fuel reduction treatments in the Pine Barrens. We installed and operate six new fire weather towers providing real-time weather data to fire managers over the internet (http://climate.rutgers.edu/stateclim/). A fuel moisture index based on forest energy balance measurements made from the flux towers is operational in New Jersey, and is being tested at a number of Ameriflux sites. We are using the network of fire weather and flux towers in conjunction with a SODAR to measure windspeed and direction up to a height of 700 m to validate predictions of MM5, a regional fire weather model, and smoke emission models. Biometric measurements, flux towers, and ecosystem- to landscape-scale models (PnET CN, BiomBGC) are used to estimate rates of forest productivity and fuel accumulation in the Pine Barrens. Collectively, these data are leading to the "next generation" of fuel models in which the dynamics of 1, 10, and 100 fuels and fuel-reduction treatments are integrated to estimate transitions among fuel models in a GIS database. These research products and other decision-support tools for managers and policymakers enable the latest science-based knowledge to be incorporated into decision processes at local to regional scales.

[1†] USDA Forest Service, Northern Research Station, Northern Global Change Program, Newtown Square, PA 19073, [2]Northern Research Station, USDA Forest Service, East Lansing, MI 48823 [†]Corresponding author; 609-894-0325, email: kennethclark@fs.fed.us.

EFFECTS OF FUEL REDUCTION TREATMENTS ON SOIL NITROGEN TRANSFORMATION IN THE SOUTHEASTERN PIEDMONT

T. Adam Coates[1†], Victor B. Shelburne[2], Thomas A. Waldrop[3], Bill R. Smith[4], and Hoke S. Hill, Jr.[5]

The Clemson Experimental Forest (CEF) in northwestern South Carolina is one of 13 sites included in the National Fire and Fire Surrogate Study. Fuel-reduction treatments (prescribed fire, thinning, and a combination of the two) were installed on the CEF beginning in the winter of 2000-2001. The inorganic nitrogen (N) contents of the soil (KCl extractable NH_4^+-N and NO_3^--N) were determined prior to treatment and in 2004 and were used to calculate three measures of soil N transformation: the rate of mineralization, the rate of nitrification, and proportional nitrification. Using an analysis of covariance, it was determined that the 2004 results for N transformation were not significantly influenced by the levels of N transformation before treatment. Analysis of variance for the 2004 results showed no significant differences among treatment means, and linear contrasts for these data suggest that none of the fuel reduction treatment means differed significantly from the control means. Differences before and after treatment were not significant for all variables when paired t-tests were used. The nitrification rate increased following these treatments in the thin only and burn only plots. These analyses suggest that the treatments have had no significant detrimental effect on soil N transformation over the 2- to 3.5-year posttreatment period, and may increase the rate of nitrification.

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[1]Graduate Student, Department of Forestry and Natural Resources, Clemson University; [2]Professor, Department of Forestry and Natural Resources, Clemson University; [3]Research Forester, USDA Forest Service, Southern Research Station; [4]Professor, Department of Entomology, Soils, and Plant Sciences, Clemson University; [5]Department Chair, Department of Applied Economics and Statistics, Clemson University; [†]Corresponding author, 265 Lehotsky Hall, Clemson, SC 29634, 864-656-3054, email: coatest@clemson.edu.

USE OF WILDLAND FIRE IN GREAT SMOKY MOUNTAINS NATIONAL PARK

Dana Cohen[1†], Beth Buchanan[2], Bob Dellinger[3], Leon Konz[4], Mark Taylor[5]

Great Smoky Mountains National Park, located in Tennessee and North Carolina, had a policy of suppressing all fires from its inception in 1934 until the adoption of wildland fire use policy in its 1996 Fire Management Plan. This enables park fire staff to manage lightning-caused fires to accomplish resource objectives so long as the fires meet certain predefined conditions. The Smokies was the first federal land management unit in the Southern Appalachians to use naturally ignited fires. Unlike prescribed fires, lightning-caused fires tend to burn over long periods with a variety of behavior and effects in a mosaic over the landscape. They have been an essential tool for better understanding the natural role of fire in the park.

[1]Fuels Specialist, Dixie National Forest; [2]Fire Ecologist, USDA Forest Service, Southern Region; [3]Biological Science Technician (Fire Effects), North Cascades National Park; [4]Fire Management Officer (ret.), Great Smoky Mountains National Park; [5]Fire Management Officer, Great Smoky Mountains National Park; [†]Corresponding author, P.O. Box 645, Panguitch, UT 84759, (435) 676-9372, email: dcohen@fs.fed.us.

HARDWOOD STEM INJURY AND MORTALITY IN SURFACE FIRES

Matthew B. Dickinson[1†] and Anthony S. Bova[2]

In surface fires, hardwood stems are heated primarily by flames.When the heat flux to those stems is sufficiently great or prolonged, necrosis of phloem and live sapwood tissues results. We are developing models of hardwood stem injury and mortality along several fronts. In collaboration with the Missoula Fire Sciences Laboratory, we are developing a stem injury and mortality model. A version of this model designed for fire managers is called FireStem (www.firelab.org). Estimating heat flux to stems is critical as an input to any stem heating model, but accurate estimates are difficult and expensive to obtain. To address this problem, we are developing an inverse heating methodology to estimate net stem heat flux that requires minimal instrumentation and has the additional advantage of using the tree stem itself as a sort of heat flux sensor. Once stem heating can be predicted, it is necessary to have a tissue necrosis model. Two approaches are being pursued for developing predictive models of tissue necrosis. In one, equations whose form is determined by heat transfer principles are parameterized by data on fire behavior, heat flux, and tissue necrosis. In a more biophysically realistic approach (that used in FireStem), tissue necrosis models are linked with the stem heating model. For small stems, heating may be roughly equal around stems, but uneven heating is the general result where fires burn on slopes or in wind. Field data and modeling are being used to improve our understanding of uneven heating so that it can be incorporated in stem injury and mortality models.

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[1]Ecologist and [2]Physical Scientist, USDA Forest Service, Northern Research Station; [†]Corresponding author, 359 Main Road, Delaware, OH 43015, 740-368-0093, email: mbdickinson@fs.fed.us.

VEGETATION RESPONSE TO PRESCRIBED FIRE ACROSS A MOISTURE/PRODUCTIVITY GRADIENT IN THE SOUTHERN APPALACHIANS

Katherine J. Elliott[†] and James M. Vose

We examined the effects of single, dormant-season fires on overstory, midstory, and ground flora species diversity and composition, overstory mortality, and tree seedling regeneration the first and second years following prescribed burns in eastern Tennessee, northern Georgia, and western North Carolina. We evaluated the effects of burning along a moisture/productivity gradient that ranged from dry, pine-hardwood to mesic, mixed-hardwood ecosystems. All treated sites were burned in spring before leaf-out (March-April), and adjacent sites were designated as controls. Within each site, vegetation plots were measured in layers: the overstory layer (trees ≥ 5.0 cm d.b.h.); the midstory layer (woody stems < 5.0 cm d.b.h. and ≥ 0.5 m height); and the ground flora layer (woody stems < 0.5 m height and all herbaceous species). Permanent plots were sampled before the prescribed burns and the first and second growing seasons after the burns. As a result of the fires, overstory mortality occurred primarily in the small size-class hardwoods and ranged from 23 percent in the dry, pine-hardwoods forest to < 2 percent in the mesic, mixed-hardwoods forest. In the midstory**,** fire reduced the basal area of woody species but prolific sprouting from hardwoods resulted in higher density of fire-sensitive hardwoods such as red maple (*Acer rubrum* L.), sourwood (*Oxydendrum arboretum* (L.) DC.)), and blackgum (*Nyssa sylvatica* Marshall). On the dry sites, blueberries (*Vaccinium* spp.), huckleberry (*Gaylussacia ursina* (M.A. Curtis) T & G)), and mountain laurel (*Kalmia latifolia* L.) increased in density. White pine (*Pinus strobus* L.) decreased in density by 20 to 50 percent and its basal area was reduced by 50 percent after burning. On the most mesic site, midstory density and basal area were not significantly changed after burning. We found no significant change in ground-flora percent cover or diversity after prescribed on mesic sites. We hypothesized that the ground flora was not significantly affected by these low- to moderate-intensity fires because the burns occurred in the spring before the emergence of herbaceous species, and the low soil depth of heat penetration (< 2.0 cm at 59°C) did not result in damage to below-ground rhizomes or buried seeds.

USDA Forest Service, Southern Research Station, Coweeta Hydrologic Laboratory; [†]Corresponding author, 3160 Coweeta Lab Rd, Otto, NC 28763, 828-524-2128, email: kelliott@fs.fed.us.

OAKS IN PRESETTLEMENT FIRE LANDSCAPES OF THE SOUTHEAST

Cecil Frost[1]

Oaks are found in all physiographic provinces of the Southeastern United States and occurred in presettlement times at all fire frequencies, including 2- to 3-year fire intervals in the most fire-exposed landscapes. Both vegetation structure and species diversity have been altered greatly by some 70 years of fire suppression. In the Sandhills of North and South Carolina, Michaux (1805) reported that "Seven-tenths of the country are covered with pines of one species, or *Pinus palustris.*" However, there was a variety of oak vegetation types within the pine-dominated southeastern landscape. The original percentage of upland oaks in one flat, south Atlantic Coastal Plain landscape was estimated at less than 1, 2.3 in a gently rolling site in the North Carolina Sandhills, and 15 at a more dissected site in South Carolina near the Sandhills-Piedmont transition. On the Piedmont itself was a band of post oak savanna in South Carolina and Georgia that ran parallel to and just west of the margin of the range of longleaf pine. With only a slightly lower fire frequency, about 2- to 5 years, this type consisted of an open canopy of post oak over a species-rich grassy herb layer. Habitats on the Southeastern Piedmont were more nearly balanced between pines—principally shortleaf pine—and hardwoods, especially white oak, post oak, mockernut hickory, red oak, and chestnut oak. In the Southern Appalachians, oak habitats were more varied and fire frequencies ranged from nearly annual in areas where the fire regime was dominated by Native American ignitions to about 5 years in some chestnut oak forests, to essentially fire free areas in cove hardwood stands where species such as white oak persisted through gap-phase succession rather than through regeneration by fire. In those stands where reproduction depended on fire, loss or depauperization of the herb layer in fire-suppressed oak forests is one of the most significant effects of fire suppression.

[1]Landscape Fire Ecologist, North Carolina Plant Conservation Program (ret.), 119 Potluck Farm Road, Rougemont, NC 27572, 336-364-1924, email: cecil.frost@earthlink.net.

EFFECTS OF PRESCRIBED BURNING ON INVASIBILITY BY NONNATIVE PLANT SPECIES IN THE CENTRAL HARDWOODS REGION

Lance Glasgow and Glenn Matlack[1]

Fire often promotes invasion by nonnative plant species, yet few studies have examined this process in forests of the Central Hardwood Region. Public agencies are considering prescribed fire as a management tool to favor the regeneration of oak (*Quercus spp*.) in this region, raising the possibility of widespread invasion of forests by nonnative species. To examine the potential role of burning in facilitating invasions, we set up an experiment simulating various aspects of forest fires at a forested site in southeastern Ohio. Treatments included high- and low-burn intensity, increased soil pH, litter removal, and a control. Seeds of two problematic nonnative species (*Microstegium vimineum* and *Rosa multiflora*) were sown experimentally following application of the treatments. Treatments were arranged in a randomized block design in two landscape positions (dry upland and moist lowland) and two canopy conditions (gap, no gap). We measured germination, stem height, leaf number, and seed production at four time intervals throughout the growing season. Germination rate was promoted by litter removal, and high- and low-intensity fire treatments for *M. vimineum*, and by high-intensity fire for *R. multiflora*. In both species, stem height and leaf number were greatest following high-intensity fire and under canopy gaps. We suggest that prescribed burning creates a disturbance suitable to invasion. Removal of leaf litter seems to be the most important factor controlling germination of our experimental species, while overall site quality (light and moisture) is most important for seedling survival. Growth and reproduction of both species were highest in the hot burn treatments, indicating that the most severe disturbances have the greatest effect on the invasibility of deciduous forests. We infer that the absence of invasion in previous studies is due to a lack of propagules rather than the unsuitability of the burned site for germination or growth.

[1]Environmental and Plant Biology, Porter Hall, Ohio University, Athens, Ohio, 45701, USA 740-593-1131; email Matlack@ohio.edu.

FUEL AND FIRE DYNAMICS IN EASTERN MIXED-OAK FORESTS

John B. Graham[1] and Brian C. McCarthy[2†]

In eastern mixed-oak forests, prescribed fire and thinning are being used as tools for a variety of management goals, including altering fuel loads. However, the specific effects of these treatments on fuel dynamics in hardwood forests are relatively unknown, as are the specific patterns of fire behavior based on fuel loading and moisture content. We conducted two experiments to address these deficiencies. First, we conducted a study to examine fuel dynamics 3 years after thinning, prescribed fire, and a combined treatment as a part of the Ohio Hills site of the Fire and Fire Surrogate program. Measurements techniques followed the Brown planar-intersect method to determine litter, duff, 1-hr (0 to 6 mm diameter), 10-hr (6 to 25 mm), 100-hr (25 to 75 mm), and sound (1,000S) or rotten (1,000R) 1,000-hr (75+ mm) fuel mass. Coarse woody debris (CWD) was evaluated for species, length, large- and small-end diameters, and decay class. Three years after treatments, thinning increased 100-hr, 1,000S, CWD, and litter, but reduced 1-hr fuels; burning increased 1,000S but reduced 1,000R; thinning followed by burning increased 100-hr, 1,000S, and CWD but reduced 1-hr, 10-hr, and duff. Changes in the larger, sound fuels (100-hr, 1000S, and CWD) appear to persist over time following treatments, while changes in finer fuels (litter, duff, 1-hr, and 10-hr) or less-sound fuels (1,000R) appear to be more ephemeral, recovering or shifting direction within 3 years after treatment. In the second experiment, we manipulated moisture and fuel-load levels (litter, 1-hr, and 10-hr) in a common garden, trial burn experiment using fuel load estimates covering the range estimated in the first portion of the study. As predicted, fires burned hotter and consumed more fuel with elevated fuel loads or lower fuel moistures. The results of this experiment can be used by managers to predict fire behavior based on fuel conditions, and can be linked to other research to predict stem mortality based on fire temperature. Collectively, our experiments can be used to evaluate the effects of silvicultural treatments on future prescribed fires. As such, they provide a realistic baseline for fuel loading and fire behavior in models of eastern mixed-oak forests.

[1]Graduate student, Ohio University, Dept. of Environmental and Plant Biology; [2]Professor of Forest Ecology, Ohio University, Dept. of Environmental and Plant Biology; [†]Corresponding author, 315 Porter Hall, Ohio University, Athens, OH 45701, 740-593-1615, email: mccarthy@ohio.edu.

USING PRESCRIBED FIRE AND MECHANICAL THINNING TO IMPROVE THE REGENERATIVE POTENTIAL OF OAK SPECIES

Scott Holevoet[1]† and Charles Ruffner[2]

Historical evidence and ecophysiological characteristics indicate a strong link between disturbance regime and the occurrence of oak (*Quercus*) species. Research indicates that many current oak forests developed following disturbances such as logging, fire, or a combination of logging and fire from the 18th century to the early 20th century. However, during the early and mid-20th century, many oak forests were entered for the last time and fire and cutting ceased. This resulted in the development of several cohorts of mixed mesophytic species, which are poised to replace the overstory with the next disturbance. To counter this, managers are being encouraged to use prescribed fire to improve the regenerative potential of oaks. These fires encourage prolific sprouting of oak trees while limiting the growth of many oak competitors such as sugar maple (*Acer saccharum*), red maple (*Acer rubrum*), American beech (*Fagus grandifolia*), yellow-poplar (*Liriodendron tulipifera*), and black cherry (*Prunus serotina*). In addition, oak seedlings develop slowly in the low light of a closed overstory and thick understory. Increased light improves growth of oak species but it also encourages the growth of competing vegetation. Prescribed fire has been viewed as an effective tool to limit competing vegetation and improve the growth and form of oak regeneration in thinned versus unburned areas. In a study in the Shawnee Hills region of southern Illinois in the spring of 2003, the benefits of prescribed burning, mechanical thinning, and a combination of the two were examined. This poster includes the results of this study and implications for current and future silvicultural recommendations.

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[1]Research Assistant, Southern Illinois University, Department of Forestry;
[2]Associate Professor of Forestry, Southern Illinois University, Department of Forestry †Corresponding Author 1448 Makanda Rd, Makanda, IL 62958: 618-319-0217, email: scottholevoet@yahoo.com.

THINNING AND BURNING TREATMENTS INFLUENCE WITHIN-CANOPY LEAF TRAITS IN SEVEN HARDWOOD SPECIES IN OHIO

Jyh-Min Chiang[1†] and Kim J. Brown[2]

Leaf nitrogen content (LN, %) and leaf mass per area (LMA, g m^{-2}) are important features that are closely linked to the photosynthetic performance of plants. Hence, models of ecosystem carbon balance often incorporate both leaf traits in simulations of potential net primary production. Forest management practices such as burning and thinning may change stand structure and soil dynamics, resulting in the changes in LN and LMA. For simulations at a larger spatial and temporal scale, additional information on species specific leaf traits and their plasticity to changing environments is needed. Our goal is to ultimately model the net effect of carbon sequestration under the future species rearrangement. The objective of this study was to understand how LN and LMA of different canopy tree species (*Quercus alba* L., *Q. coccinea* Muenchh., *Q. prinus* L., *Q. velutina* Lam., *Carya* spp., *Acer rubrum* L., *Liriodendron tulipifera* L.) responded to thinning and/or burning treatments; and different landscape and soil properties in southern Ohio. We selected five trees for each of the seven species at each of the 12 experimental units (three replicates of control, thin, thin + burn, and burn treatments) of the Fire and Fire Surrogate (FFS) study. Leaves from the top and bottom of the crown profile were collected (N = 850) in the summer of 2003. We found significant effects (positive) of FFS treatments on LMA for most species at the bottom of the tree crown. FFS treatments also significantly increased LN of *Q. coccinea* and *L. tulipifera* at the bottom of the tree crown. Sun leaves showed consistent LN and LMA regardless of FFS treatments. Regression tree analysis showed no effects of landscape features and soil properties on LN and LMA. Interspecies differences accounted for most variations of LN and SLM. Our study suggested FFS treatments increased the net primary production of leaves at the lower crown position. Different species tended to retain their respective LN and SLM under different landscape and soil features. This finding will facilitate our efforts to model the impacts of future species redistribution of ecosystem carbon balances.

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[1]Ph.D. Candidate, Department of Environmental and Plant Biology, Ohio University; [2]Assistant Professor, Department of Environmental and Plant Biology, Ohio University; [†]Corresponding author, 317 Porter Hall, Ohio University, Athens, OH 45701, 740-593-0405, email: jc293302@ohio.edu.

ESTIMATING FIRE BEHAVIOR FROM OVERHEAD IMAGERY IN EASTERN OAK FORESTS

Robert Kremens[1†], Anthony Bova[2], Matthew Dickinson[3], Jason Faulring[4], Shari McNamara[5]

We have used multiple-frame overhead infrared imagery to measure heat flux from a prescribed fire in Southeastern Ohio. These data were obtained using a specially designed wildfire airborne mapping camera that produces accurate geographic maps of short, mid, and longwave infrared emission along with very high-resolution visible images. These images may be overlaid to produce a total heat map. In-fire measurements were performed simultaneously with overhead image capture to provide a cross-calibration of ground leaving energy flux with sensor-reaching energy flux. This technique removes the effects of atmospheric absorption and uncertainties in instrument calibration. We correlated the heat release as measured by the airborne camera system to conventional measures of fire severity such as fuel burn-up. Forty plots were measured using conventional methods. Excellent agreement was obtained between conventional plot measurements of fuel consumption and the airborne heat-flux measurements. The airborne technique allows simultaneous observation of entire burn units and may provide a substantially better (higher accuracy, more synoptic view and lower cost) measure of fire characteristics than ground-based methods that are based on conventional sampling.

[1]Senior Research Scientist, Rochester Institute of Technology; [2]Physical Scientist, USDA Forest Service, Northern Research Station; [3]Ecologist, USDA Forest Service, Northern Research Station; [4]Integration Engineer, Rochester Institute of Technology; [5]Graduate Student, Rochester Institute of Technology; [†]Corresponding author, 54 Lomb Memorial Drive, Rochester, NY 14623, 585-475-7286, email: kremens@cis.rit.edu.

EFFECTS OF SILVICULTURAL TREATMENTS ON ACORN WEEVIL OCCURRENCE IN MIXED-OAK FORESTS

Jeffrey A. Lombardo[1†] and Brian C. McCarthy[2]

Oak regeneration failure in the hardwood forests of the Central and Eastern United States has been well documented. Strict policies of fire suppression likely have contributed to this decline by allowing shade-tolerant species to dominate the understory, suppressing the growth of oak seedlings and their subsequent release to larger size categories. What is not clear is the role that fire suppression may have played on the major oak seed predator, the acorn weevil (Coleoptera: Curculionidae). Weevils have been known to depredate a large percentage of acorns produced in a given growing season. This can result in lower seedling recruitment. The resulting low seedling density reduces the efficacy of the shelterwood-burn technique in regenerating oak stands. The purpose of this research was to determine the effects of three commonly used silvicultural treatments on weevil occurrence in mixed-oak forests of southeastern Ohio. For this study we used the silvicultural treatments of the National Fire and Fire Surrogate study at the Zaleski State Forest and Vinton Furnace Experimental Forest sites, both of which are located in Vinton County, Ohio. The treatments include: prescribed fire, thinning, thinning followed by prescribed fire, and an untreated control. We tested the hypothesis that burning in the spring reduces the number of weevils in an area by inducing mortality on the adults as they emerged from the soil, and that thinning increases weevil activity by allowing more sunlight to penetrate the canopy, thus increasing ambient temperature. Adult weevils were captured using insect traps; acorns traps were used to determine the presence of larvae in acorns. Data from the insect traps support our hypothesis at the Vinton Furnace site; however, the Zaleski site had a more even distribution of weevil occurrence, signifying a possible site-by-treatment interactive effect. Results of this study show that fire suppression may negatively affect oak regeneration by allowing a greater occurrence of acorn weevils in a stand, thus reducing the number of new oak seedlings.

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[1]Graduate Student, Department of Environmental and Plant Biology, Ohio University; [2]Professor of Forest Ecology, Department of Environmental and Plant Biology, Ohio University; †Corresponding author, 317 Porter Hall, Ohio University, Athens, OH 45701, 740-597-3116, email: jl292804@ohio.edu.

OAK AND MAPLE STUMP SPROUT DYNAMICS IN RESPONSE TO THINNING AND BURNING TREATMENTS IN SOUTHERN OHIO

Robert P. Long[1†], David K. Apsley[2], and Joanne Rebbeck[1]

Three southern Ohio sites (REMA, Tar Hollow, and Zaleski) are the focus for testing the efficacy of thinning (T) and the combination of thinning and burning (TB) on oak regeneration. In T and TB stands, sprouts on stumps (≥ 10 cm diameter) are an important component of regeneration dynamics. Thinning treatments, which reduced basal area by about 30 percent, were completed in winter-spring of 2000-2001. Stands were burned in March-April 2001. During 2004, we evaluated the density and growth of stump sprouts in sixteen 400-m^2 plots in each of six 20-ha treatment units. Treatment effects on red oak group (ROG = *Quercus rubra*, *Q. coccinea*, *Q. velutina*; n = 40 stumps) and white oak group (WOG = *Q. alba*, *Q. prinus*; n = 144 stumps) stump sprout density and growth were compared with their major competitor, red maple (*Acer rubrum*; n = 121 stumps). Analysis of variance showed that burning reduced the mean number of sprouts per hectare for all oaks and red maple combined. For instance, sprout densities were significantly ($P < 0.006$) lower in TB stands (491 sprouts/ha) than in T stands (875 sprouts/ha). There were consistently more red maple sprouts, which averaged 458/ha, across all six sites, compared with 62/ha for ROG species, and 162/ha for WOG species. However, these results may be related to the greater sprouting potential for smaller diameter red maple stumps, which averaged 25.5 cm (range: 10 to 50 cm) compared with mean stump diameters of 43.1 cm (range: 11 tot 73 cm) and 37.6 cm (range: 15 to 90 cm) for ROG and WOG stumps, respectively. Mean height and diameter growth for red maple, ROG, and WOG sprouts, evaluated by species group, was not significantly ($P > 0.50$) affected by treatments. The mean height of red maple sprouts, 3.3 m, was significantly ($P < 0.001$) greater than that of ROG and WOG sprouts, 2.5 and 2.0 m, respectively. Similarly, the mean basal diameter of red maple sprouts, 4.1 cm, was significantly greater than that of ROG and WOG sprouts, 2.2 and 2.0 cm, respectively. These results suggest that oak sprouts were not as competitive as red maple sprouts in thinned treatments, and that prescribed burning immediately after cutting did not improve the competitive status of oak stump sprouts.

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[1]USDA Forest Service, Northern Research Station; [2]Ohio State University Extension; [†]Corresponding author, 359 Main Road, Delaware, OH 43015-8640, 740-368-0050, email: rlong@fs.fed.us.

IMPACTS OF REPEATED PRESCRIBED FIRES ON FUELS IN A CENTRAL APPALACHIAN HARDWOOD FOREST

Jessi E. Lyons[1], Elizabeth Loucks[2], Mary A. Arthur[3†], David Loftis[4]

Throughout the central hardwood forest region, managers are increasingly using fire to attain various management objectives, including the promotion of oak regeneration, forest thinning, and fuel reduction. In the Daniel Boone National Forest in Kentucky, managers have used fire as a management tool in upland oak forests for more than 10 years. Funding for prescribed burning often comes with the justification that burning will reduce fuels and help prevent future unplanned fires. We examined fuel changes following one and two prescribed fires in spring. Fuel mass was sampled before the introduction of fire, after a single prescribed fire, after leaffall following one fire, and after two prescribed fires. Concerns about the potential for increased soil erosion following repeated fire prompted sampling of mineral soil exposure after the second prescribed fire. This study demonstrated that one and two prescribed fires decreased leaf litter, which was replaced in mass but not fuel continuity during leaffall. A single prescribed fire significantly decreased 10-h woody fuels, which were replenished the next leaffall in concurrence with leaf litter. Repeated prescribed fire significantly increased 1-h woody fuels. The duff layer decreased on burned treatments following a single fire, and decreased further after a second fire. After one burn, duff depth decreased more on subxeric landscape positions than on submesic positions. Mineral soil exposure also increased with number of fires. It was highest on subxeric and intermediate plots after one burn and higher on submesic plots after two burns. Subxeric plots showed a strong correlation between mineral soil exposure and litter consumed. Fuel consumption, duff depth, and mineral soil exposure varied with landscape position. On sites with steep slopes, repeated fire could lead to increased erosion due to exposed mineral soil. Common encounters with turkeys and their scratching has stimulated questions about their possible affinity for recently burned areas as a contributing cause for increased soil exposure.

[1]Lab Technician, Department of Forestry, University of Kentucky; [2]Former Graduate Student, Department of Forestry, University of Kentucky; [3]Associate Professor, Department of Forestry, University of Kentucky; [4]Research Forester, USDA Forest Service, Bent Creek Experimental Station; [†]Corresponding author, T.P. Cooper Building, Lexington, KY 40546-0073, 859-257-2752, email: marthur@uky.edu.

FACTORS INFLUENCING WILDFIRE OCCURRENCE AND DISTRIBUTION IN EASTERN KENTUCKY

John K. Maingi[1†] and Mary C. Henry[2]

Most wildfires fires in Kentucky occur in the heavily forested Appalachian counties in the eastern portion of the state. These fires have been attributed to causes ranging from weather conditions to socioeconomic factors. We reconstructed a brief fire history of eastern Kentucky using Landsat ™ and ETM+ images acquired between 1985 and 2002. We then examined relationships between fire frequency and area burned, and abiotic and human factors. Abiotic factors included Palmer Drought Severity Index (PDSI; negative values indicate drier conditions), slope, aspect, and elevation, and human factors such as county unemployment rates and distance to roads and populated areas. About 83 percent of the total area burned experienced fire only once, 14 percent twice, and 3 percent three times. More fires burned in the winter (January-March) fire season than in the fall (October-December) season, but the latter fires were larger on average and accounted for about 71 percent of the total area burned. Fire size was negatively correlated with the mean PDSI in both fire seasons. There was a higher frequency of burning both at higher elevations and on steeper slopes. There was no relationship between aspect and frequency of burning. Despite the perceived link between jobless claims in the region and frequency of fires, we found no correlations between monthly unemployment rates and arson-caused fires. The number of fires recorded was negatively correlated with both distance of fire to road ($p < 0.001$) and distance to populated area ($p < 0.001$).

[1,2]Assistant Professors, Department of Geography, Miami University;
[†]Corresponding author, 210 Shideler Hall, Oxford, OH 45056 USA,
(513) 529-5024, email: maingijk@muohio.edu.

HARDWOOD REGENERATION FOLLOWING PRESCRIBED FIRE AND THINNING IN MIXED-OAK FORESTS OF SOUTHEASTERN OHIO

Brian C. McCarthy[1†] and Matthew A. Albrecht[2]

Stand thinning and prescribed fire are widely used forest management tools throughout the hardwood forests of the Central Appalachians. Recent changes in historical disturbance regimes are hypothesized to have altered the structure and composition of many oak-dominated forests. In particular, there are few oaks in the regeneration layer (especially 1 to 4 inches d.b.h.) throughout the understory of most oak-dominated forests. Restoration efforts likely will require significant changes in disturbance regimes to alter the current successional trajectory. We examined the patterns of hardwood regeneration in a replicated field experiment over 5 years. Three forests were selected for study in southern Ohio. Within each forest, four adjacent 20 to 30-ha stands were selected, with one each allocated randomly to the following treatments: control, prescribed burn, thin, and thin followed by prescribed burn. Seedlings and saplings were sampled in replicated study plots in the year prior to treatment (2000), the year following treatment (2001), and 3 years later (2004). Structural data (density) were analyzed by analysis of covariance; compositional data (relative abundance) were analyzed by ordination procedures. Treatments had various effects dictated by species life history, reproductive episodes, and stochastic events. A single prescribed fire reduced seedling and sapling densities of red maple, the understory dominant in these forests. However, red maple rapidly recovered to pretreatment levels in all units four growing seasons after the disturbances. Mechanical thinning treatments accelerated understory recruitment of early successional, shade-intolerant tree species that regenerated from seed, e.g., tuliptree, and resprouted from a seedling bank, e.g., sassafras. Tuliptree also was strongly influenced by burning, resulting in greatly enhanced reproduction and advancement into the sapling size class via fast growth rates. White oak showed no response, black oak increased slightly, and chestnut oak varied moderately with respect to treatment. Overall, the treatments were effective at advancing a greater proportion of tuliptree into the succeeding forest. However, the effects of treatment on oak regeneration are unclear, so additional longitudinal data will be needed. Subsequent periodic fires may help to slow mesic competitors over the long run.

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[1]Professor of Forest Ecology, Department of Environmental and Plant Biology, Ohio University, [2]Graduate Student, Department of Environmental and Plant Biology, Ohio University; [†]Corresponding author, 416 Porter Hall, Ohio University, Athens, OH 45701, 740-593-1615, email: mccarthy@ohio.edu.

SOIL RESPIRATION RESPONSES TO TEMPORAL, TOPOGRAPHIC, AND SILVICULTURAL FACTORS IN A SOUTHEASTERN OHIO FOREST

Dawn R. McCarthy[1†] and Kim J. Brown[2]

Soil respiration (CO_2 efflux) is an important component of carbon loss from forest ecosystems. Attempts to understand the relationship between silvicultural practices and soil respiration are complicated by physical and biological heterogeneity at the landscape scale. We examined the effects of burning on soil temperature, soil moisture, and respiration while controlling for slope position and thinning intensity. Treatments included unburned control, low intensity fire, high intensity fire, lime fertilization, and leaf litter removal. These treatments (each 2×2 m) were replicated in 20 experimental blocks at different slope positions (upper vs. lower) and thinning treatments (closed canopy vs. 50-percent removal) in a mixed-oak forest in southeastern Ohio. Monthly field measurements were conducted from April to November in 2004. Treatment and slope position were significant factors principally during the growing season. Upslope and thinned plots experienced midsummer soil-water reductions of 6 to 10 percent, respectively. Leaf litter removal reduced respiration by 15 percent compared to the control, while lime fertilization increased respiration by 10 percent compared to the control. From the entire season's data we developed a nonlinear model of soil respiration as a function of soil temperature for each treatment. Model equation parameters differed only for slope position and fertilization treatment, revealing greater sensitivity in respiration rates at lower slopes compared to upper slopes. Overall, thinning did not significantly affect soil respiration, but there was a trend toward reduced soil respiration rates ($P = 0.09$) in the low-intensity burn plots. Resulting values for soil moisture and soil temperature are a function of both topography and land use, leading to unequal effects of thinning and burning across the landscape. While the current study is informative with respect to the microclimatological mechanisms of thinning and burning on soil respiration, the small treatment area prevents a large-scale extrapolation to a thin/burn scenario due to other factors, e.g., mechanical soil disturbance.

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[1]Graduate Student, Department of Environmental & Plant Biology, Ohio University; [2]Assistant Professor, Department of Environmental & Plant Biology, Ohio University; [†]Corresponding author, 317 Porter Hall, Ohio University, Athens, OH 45701, 740-707-1342, email: db642701@ohio.edu.

CHANGES IN THE STAND STRUCTURE AND SPECIES COMPOSITION OF OAK-DOMINATED COMMUNITIES FOLLOWING A PRESCRIBED BURN

Virginia L. McDaniel†, Michael A. Jenkins, and Robert N. Klein

Fire suppression has affected many forest communities in Eastern North America. It has played a significant role in deterring regeneration of oaks and encouraging regeneration of more shade-tolerant species in oak dominated communities. At Kings Mountain National Military Park in South Carolina, fire suppression has altered stand structure and species composition on oak-dominated ridges. It has given shade-tolerant, fire-intolerant species a competitive advantage over shade-intolerant, fire-tolerant species. In 2000, we sampled 19 plots prior to a prescribed burn and resampled the same plots a year after burning. Prior to burning, the overstory was dominated by oaks, but advanced regeneration was dominated by blackgum, red maple, and sourwood. The pre-burn diameter distribution of oak species was bell-shaped, but the distribution of other species (including black gum, sourwood, and red maple) was reverse J-shaped. This indicates that understory composition in fire suppressed stands has shifted from dominance by relatively xeric oak species to dominance by more mesic species. One year after the burn, the oaks continued to have a bell-shaped distribution, but the number of small stems of fire-intolerant species was greatly reduced, resulting in a more bell-shaped distribution.

Twin Creeks Natural Resources Center, Great Smoky Mountains National Park; †Corresponding author, 1314 Cherokee Orchard Rd., Gatlinburg, TN 37738, 865-430-4751, email: virginia_mcdaniel@nps.gov.

EXPERIMENTAL EVALUATION OF FIRE HISTORY RECONSTRUCTION USING DENDROCHRONOLOGY IN WHITE OAK (*QUERCUS ALBA*)

Ryan W. McEwan[1], Todd Hutchinson[2], Robert Ford[3], Brian C. McCarthy[4†]

Fire history often is reconstruction through dendroecological analysis of fire scars on tree boles. An understanding of historical fire regimes is critical for understanding current forest structure and forest dynamics, and for evaluating forest management practices. Even so, the validity of this approach has rarely been evaluated experimentally. Do we know whether fire scars accurately record fire history in dendroecological samples? We collected basal cross sections from stands where prescribed fires had occurred to examine the efficacy of fire history reconstruction in eastern hardwood forests. We collected samples from 82 trees in two sites, each having experienced three fire treatment intervals (frequent, infrequent, and control) over the last nine years. Prescribed fires were set in 15 area/time combinations. Of these, 10 were recorded as fire scars. In all areas, the first year of prescribed fire was recorded in at least one of the samples, and scarring occurred when there had been pauses in the burning regime of more than 3 years. Although the known fire history generally was reconstructed through tree-ring analysis, the proportion of samples bearing scars was exceedingly low (12 percent). Fires that occurred in subsequent years generally were not recorded in the tree-ring record (one of six cases), suggesting that multiyear fuel accumulation is necessary to create fire intensity needed to wound trees. Our study suggests that tree-ring analysis is an effective tool for reconstructing burning regimes when the fire-free interval is long enough to allow for fuel accumulation; however, a frequent burning regime (intervals of 1 to t3 years) may be undetectable.

[1]Doctoral Candidate, Ohio University, Department of Environmental and Plant Biology; [2]Research Ecologist, USDA Forest Service, Northeastern Research Station; [3]Dendrochronologist, USDA Forest Service, Northeastern Research Station; [4]Professor of Forest Ecology, Ohio University, Department of Environmental and Plant Biology; [†]Corresponding author, 315 Porter Hall, Ohio University, Athens, OH 45701, 740.593.1615, email: mccarthy@ohio.edu.

IMPACT OF FIRE ON SOIL RESOURCE PATTERNS IN MIXED-CONIFER FORESTS IN THE SOUTHERN CASCADE RANGE OF NORTHERN CALIFORNIA

Jessica Miesel[1†], Carl Skinner[2], and Ralph Boerner[3]

The montane ecosystems of the Cascade Range have been subjected to repeated manipulation and active fire suppression for more than a century. This has resulted in changes in community structure that contribute to increased wildfire hazard and severity. Ecosystem restoration to reduce the wildfire hazard has received substantial attention in recent years, though many ecological questions remain unanswered. This study addresses below-ground impacts of restorative treatments. We report preliminary results of the application of prescribed fire and the combination of fire and thinning on soil chemical and microbial parameters in treatment units of 10 ha each in the Klamath National Forest of northern California. Soil pH and total inorganic N increased in burned units (with and without thinning) one year after treatment, but differed significantly between treatments. Soil organic C decreased and C:N ratio increased as a result of fire in the burn-only treatment; no significant changes were observed for thin+burn treatment plots. Nitrogen mineralization rates did not change as a result of fire for both burn-only and thin+burn plots. Activity of acid phosphatase was reduced by fire, with and without thinning, whereas activity of chitinase was reduced by fire in thin+burn plots only. Changes in levels of phenol oxidase activity as a result of fire were not significant for treatments with or without fire. Our study contributes to an understanding of the ecological effects of fire on soil chemical and microbial properties. This project is part of the interdisciplinary national Fire and Fire Surrogate Network study and will provide information for informed land management decisions.

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[1]Graduate Research Assistant, The Ohio State University, Department of Evolution, Ecology, and Organismal Biology; [2]Geographer, USDA Forest Service Pacific Southwest Research Station; [3]Professor, The Ohio State University, Department of Evolution, Ecology, and Organismal Biology; [†]Corresponding author, 300 Aronoff Laboratory, 318 W. 12th Ave., Columbus, OH 43210, (614) 292-9373, email: miesel.1@osu.edu.

AVIAN COMMUNITY RESPONSES TO PRESCRIBED BURNS AND SHELTERWOOD HARVESTS

Donald B. Miles[1†], Henry Streby[2], and Dan Weber[2]

Previous forest management practices in U. S. Forests emphasized the suppression of fires. A consequence of this management practice has been the excessive accumulation of fuel and high tree densities. To avoid severe and catastrophic fires, resource managers now use a variety of treatments to diminish fuel loads. Alternative treatments include prescribed burns and shelterwood harvests, both of which are assumed to mimic natural ecosystem function. However, avian responses to silviculture treatments have been investigated only recently. We present the results of a study to determine whether avian community composition, productivity, and foraging behavior are affected by fire and fire surrogate treatments. Avian species composition and abundance in each of four treatment types were estimated using point-count census methods. The treatment plots were control, burn, thin, and burn + thin. The treatments were replicated at three sites. We also determined differences in nest productivity. Avian censuses were conducted 1 year before the treatments to obtain baseline data on species composition and relative abundance, and 3 years posttreatment. Our results show acute and relatively long-term responses to forest disturbance. We compared community composition, nest productivity, and foraging behavior using data from the baseline samples to determine whether species composition was homogeneous among the treatments. Species composition was significantly different among treatments, with the highest diversity in thin and burn plots. However, there was considerable heterogeneity in the data. Average nest productivity also varied across treatments. We document how species composition shifts in response to the thin and burn treatments.

[1]Professor of Vertebrate Zoology, Ohio University, Department of Biological Sciences, [†]Corresponding author, Athens, Ohio 45701,
[2]Graduate Research Biologist, Department of Biological Sciences, Ohio University, Athens, OH 45701, 740-593-2317, email: dmiles2@ohio.edu.

REHABILITATION OF AN UPLAND HARDWOOD FOREST USING PRESCRIBED FIRE AND HERBICIDE

Rebecca Montgomery[1†], Jeff Earl[1], Chris Stuhlinger[2], Matthew Pelkki[3], and Kyle Cunningham[4]

Many upland hardwood forests have been degraded due to improper management (high-grading) or insect/disease outbreaks. The absence of periodic fires has changed the forest composition to a higher proportion of undesirable shade-tolerant species. Some desirable species such as oaks have declined. Methods are needed to rehabilitate these forests by encouraging the regeneration and growth of desirable tree species and controlling undesirable species. The application of prescribed fire and herbicide are being evaluated in an Ozark upland hardwood forest in northwestern Arkansas. In a landscape where the wildland-urban interface is expanding rapidly, concern is increasing with respect to the use of herbicides and fire as management tools. The advantages and limitations of each method must be understood so that appropriate recommendations can be made for specific situations. In this study, treatment plots were left alone, burned in the spring, treated with Chopper® herbicide in the fall to kill undesirable trees, or treated with both fire and herbicide. Fire intensity on the burned plots was measured with Tempilaq® (Tempil, South Plainfield, NJ) temperature-indicating liquids applied to metal tags, and varied widely depending on available fuels. Fire intensity (temperature) can be related to the understory kill of small-diameter stems and subsequent regeneration of desirable species. Early observations indicate successful control of larger stems by herbicide, while burning may only temporarily control sprouting and growth of small undesirable stems. Prescribing repeated burns, or applying herbicide, might achieve the desired result of restoring a significant oak component to these forests.

[1]Program Technicians, University of Arkansas – Monticello, School of Forest Resources; [2]University System Forest Manager, Arkansas Forest Resources Center; [3]George H. Clippert Endowed Chair, University of Arkansas – Monticello, School of Forest Resources; [4]Forestry Instructor, University of Arkansas, Cooperative Extension Service; †Corresponding author, P.O. Box 3468, Monticello, AR 71656, 870-460-1052, email: montgomeryr@uamont.edu.

ALTERED DISTURBANCE REGIMES: DEMISE OF FIRE IN THE EASTERN UNITED STATES

Gregory J. Nowacki[1†] and Robert A. Carr[2]

A series of maps was generated to help alert and educate people to the pervasiveness of fire regime changes across the Eastern United States. Using geographic information systems, we assigned fire regimes to spatial vegetation databases to depict past and current conditions. Comparisons revealed substantial reductions in fire throughout the East. The most dramatic shifts took place in the former midwestern grasslands and across a broad swath of Southern and Central States, where pine and oak communities historically dominated. Land-use changes, e.g., agricultural and forest-type conversions, and 20th century fire suppression largely explain these shifts. Fire regime change was somewhat limited in northern climes (northern hardwood systems), in the mixed mesophytic region, and within the Mississippi Embayment. Unforeseen consequences of prolonged fire suppression are mounting while restoration opportunities are waning.

[1]Regional Ecologist and [2]Regional GIS Planning Analyst, USDA Forest Service, Eastern Region; [†]Corresponding author, 626 E. Wisconsin Ave., Milwaukee, WI 53202, 414-297-1977, email: gnowacki@fs.fed.us.

LIGHTNING-INITIATED GROWING SEASON FIRES CHARACTERIZED OAK-DOMINATED ECOSYSTEMS OF SOUTHERN OHIO

Sheryl M. Petersen[1] and Paul B. Drewa[2†]

The persistence of oak-dominated ecosystems throughout the Central Hardwoods Region has been attributed to chronic, noncatastrophic fire regimes prior to Euro-American settlement. The nature of these fires is not entirely understood. We used lightning strike, thunderstorm activity, and precipitation data to elucidate the seasonal timing of natural fire regimes for southern Ohio. Our results suggest that fires are most likely to be initiated during the late growing season (especially August) when there is a high frequency of lightning strikes and thunderstorm activity that occurs simultaneously with dry ground-cover fuel conditions. We hypothesize that the use of prescribed growing-season fires will effectively deter encroachment of shade-tolerant hardwood vegetation and foster oak regeneration. Such fires will provide an evolutionary basis for the conservation of oak-dominated ecosystems of southern Ohio.

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[1]Ph.D. Graduate Student; [2]Assistant Professor, Department of Biology, Case Western Reserve University; [†]Corresponding author, Department of Biology, Case Western Reserve University, 10900 Euclid Ave., Cleveland, OH 44106, 216-368-4288, email: paul.drewa@case.edu.

USE OF TEMPERATURE-SENSITIVE PAINTS AS AN INDEX OF HEAT OUTPUT FROM FIRE

Joanne Rebbeck[1†], Daniel Yaussy[2], Louis Iverson[3], Todd Hutchinson[3], Robert Long[4], Anthony Bova[5] and Matthew Dickinson[6]

Early spring, dormant-season prescribed surface fires were conducted in 2001 and repeated in 2005 within thinned and three unthinned mixed-oak forests of southeastern Ohio. In all, 120 ha were burned on three forests. Research has shown that maximum temperatures reached by measurement devices placed in fires can be useful in evaluating fire effects on vegetation. Most commonly used are tags painted with Tempilaq® temperature sensitive paints (Tempil Inc., South Plainfield, NJ). Paints melt and change color at specific temperatures. This low-cost ($0.05 per unit), "low tech" method of measuring the maximum temperature was compared with a high-cost ($110 per unit), high-tech Type K® thermocouple probes and Hobo® data logger (Onset Computer Corporation, Bourne, MA) system. Maximum temperatures have been shown to indicate fuel consumption and fireline intensity. Aluminum tags (2.5 x 7 cm) were painted with six (2001) or nine (2005) Tempilaq paints ranging from 79° to 427°C. Tags were hung at the same height (25 cm) and location as the probe tips. Data loggers were programmed to record probe temperature every 1 to 2 seconds. There was very good agreement between the maximum temperatures recorded by the two devices during both fires. Maximum temperatures were higher in the 2005 than in the 2001 fires (the first fires on these sites in decades). During 2001, maximum temperatures in the burn only units were higher than those in the thined and burned units. No treatment differences were observed in 2005. The impacts of more intense fires on vegetation are being monitored. Paint tags indicate maximum temperatures and are inexpensive, but thermocouple probes provide more information and better correlations with fire behavior.

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[1]Plant Physiologist, [2]Forester, [3]Ecologist, [4]Plant Pathologist, [5]Physical Scientist, and [6]Ecologist, USDA Forest Service, Northern Research Station; [†]Corresponding author, 359 Main Road, Delaware, OH, 43015, 740-368-0054, email: jrebbeck@fs.fed.us

IMPACT OF FIRE, SILVICULTURAL MANIPULATION, AND MICROSITE CONDITIONS ON WOODY DEBRIS DECAY DYNAMICS IN A MIXED-OAK FOREST

Darrin L. Rubino[1†], Lindsey E. Hesch[2], and Brian C. McCarthy[3]

Woody debris (downed limbs and boles) is an important component of forested ecosystems. It serves as habitat for a wide variety of organisms and plays an important role in nutrient cycling and dynamics. However, little is known about the wood-decay process, especially in the topographically complex mixed-oak forests of Eastern North America. The goal of this investigation was to determine how fire and silvicultural management affect the decay rate of woody debris by using wood blocks cut from commercially available native lumber. This study was conducted in Zaleski State Forest in southeastern Ohio (an Ohio Hills study site in the long-term Fire and Fire Surrogate study). Following silvicultural treatment and burning, wood blocks of “red” oak (*Quercus* L. subgenus *Erythrobalanus*) and “white” oak (*Quercus* L. subgenus *Lepidobalanus*) were placed on the forest floor in burned, thinned, thinned + burned, and control (no manipulation) units. In each unit, blocks were placed in xeric, e.g., ridges and southwest-facing slopes, and mesic, e.g., valleys and northeast-facing slopes, microsites to also investigate the impact of moisture availability on the decay process. Site moisture availability was determined using the integrated moisture index. After 27 months of decay, mean mass loss was significantly ($P < 0.05$) greater for red than white oak. Mean mass loss was 51.9 ± 1.7 (SE) percent for red oak and 36.3 ± 1.3 percent for white oak. Using a two-way ANOVA to compare the mean percent mass loss in each unit, microsite, and microsite × unit combination revealed no significant differences in decay rate for white oak. Alone, microsite and unit had no significant affect on decay rate in red oak. However, the microsite × unit interaction was observed for red oak. The fastest decay rates were observed in mesic microsites in the thin + burn unit and the slowest in the mesic microsites in the thinned unit. Decay dynamics appear to be species specific and fire and silvicultural manipulation may lead to complex changes in decay dynamics in topographically dissected mixed-oak forests of Eastern North America.

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[1]Assitant Professor of Biology, Biology Department, Hanover College;
[2]Graduate Assistant, Botany Department, Miami University of Ohio;
[3]Professor of Forest Ecology, Department of Environmental and Plant Biology, Ohio University; [†]Corresponding author, P.O. Box 890, Hanover, IN 47243-0890, 812-866-7247, email: rubino@hanover.edu.

EFFECTS OF FOREST MANAGEMENT ON SOIL SEEDBANK SPECIES COMPOSITION IN A MIXED-OAK FOREST IN SOUTHEASTERN OHIO

Lisa R. Schelling[1†] and Brian C. McCarthy[2]

Burning and thinning are common management practices in mixed-oak forests. The impacts of these practices on the soil seedbank have not been examined. This information is important to consider to evaluate the full impact of these treatments on plant community dynamics in the understory. Species composition of the soil seedbank was examined under four forest management treatments: thin, burn, thin followed by burning, and an untreated control. Sampling areas were located in Vinton County, Ohio, at Zaleski State Forest and the Raccoon Ecological Management Area. Both sites are part of the USDA's Fire and Fire Surrogate study. Thinning was conducted in the fall of 2000 and burns were conducted in the spring of 2001. Soil samples were collected in March 2004, 3 years following treatment. Samples were placed in the greenhouse and the emergence method was used to assess the species composition of seeds present in the soil. Seventy-one species were found in the soil seedbank. Ruderal and early successional species such as fireweed (*Erechtites hieraciifolia* (L.) Raf.), sedges (*Carex* spp.), and brambles (*Rubus* spp.) were the most commonly occurring species across all treatments. Multivariate analysis techniques revealed no grouping by stand management treatment. Log-linear analysis showed a treatment by functional group effect which, while not statistically significant (P = 0.0872), suggests possible biological significance. In different treatments there were different proportions of the various functional groups (grasses, perennial herbs, etc.). There seems to be no detectable difference in the species composition of the soil seedbank after short-term exposure to different forest management treatments, probably because the species that maintain a seedbank in the soil tend to be early successional. These species are not particularly sensitive to disturbance, but usually respond positively to disturbance. More disturbance-sensitive forest herb species tend not to maintain a soil seedbank, and thus did not register a response in this study. Studies examining above-ground species dynamics likely are sufficient to assess the impact of forest management on understory dynamics.

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[1]Graduate Student, Department of Environmental and Plant Biology, Ohio University; [2]Professor of Forest Ecology, Department of Environmental and Plant Biology, Ohio University; [†]Corresponding author, 315 Porter Hall, Ohio University, Athens, OH 45701, 740-597-3116, email: ls903103@ohio.edu.

OAK-GRASSLAND RESTORATION DEMONSTRATION AREA PROJECT

Leslie M. Smith[1†] and James R. McCoy[2]

Land Between the Lakes is home to two oak-grassland restoration demonstration areas. The demonstration areas create conditions over a large contiguous landscape to demonstrate the feasibility of ecological restoration of an oak-grassland forest and the benefits it can provide to native wildlife and public recreation. Historical accounts and ecological research indicates these areas had an open oak canopy and understory dominated by grasses created and maintained through the use fire by the American Indians. The demonstration area totals about 8,000 acres divided into the north unit in Kentucky (3,000 acres) and the south unit in Tennessee (about 5,000 acres). The units are located near educational outreach facilities that will encourage public interest and learning opportunities. Treatments will focus on the tree thinning and prescribed fire. Fire effects monitoring and research will continue throughout the restoration effort. "Core" areas adjacent to the demonstration areas serve as controls to gauge the effectiveness of the restoration. Nearly 350 acres in the southern restoration unit were treated with prescribed fire in the spring of 2005, and 70 acres of timber have been marked for thinning. Fire-effects monitoring plots have been established and observations are continuing.

[1]Wildlife Biologist-Trainee, USDA Forest Service, Land Between the Lakes; [2]Wildlife Biologist/Fire Management Officer, USDA Forest Service, Land Between the Lakes; [†]Corresponding author, 100 Van Morgan Drive, Golden Pond, KY 42211, 270-924-2084, email: lesliemsmith@fs.fed.us.

INTEGRATED UPLAND HARDWOOD RESEARCH IN ARKANSAS' BOSTON MOUNTAINS

Martin A. Spetich[1]

A series of interrelated studies is examining upland hardwood forest dynamics and development in the southwestern lobe of the Central Hardwood Region. The overall objective of this research is: 1) address practical, applied forest management needs based on a combination of current scientific thought and needs expressed by field practitioners, and 2) disseminate results to scientists, practitioners, and the public in ways that will benefit both society and the environment. The studies are addressing prescribed fire and fire history, growth, woody species reproduction, competitive capacity, stand dynamics, stand composition, forest species restoration, quantitative silviculture, development of forest management methods, forest ecology, disturbance ecology, and diversity of these upland hardwood forests.

[1]USDA Forest Service, Southern Research Station, Arkansas Forestry Sciences Laboratory, P.O. Box 1270, Hot Springs, AR 71902, 501-623-1180 Ext. 105, email: mspetich@fs.fed.us.

ASSIGNING FIRE REGIMES ON THE MONONGAHELA NATIONAL FOREST, WEST VIRGINIA

Melissa Thomas-Van Gundy[1]

The use of prescribed fire on the Monongahela National Forest is expected to increase as the role of fire is better understood and appreciated. Ecological knowledge of an area's fire regime is essential in meeting the intent of a burn program. A fire-adapted vegetation model emulating the coarse-scale work was developed in 2002 as a first attempt to assign fire regimes on the Monongahela. Spatial analyses and maps were generated using geographic information systems, specifically the Model Builder extension in ArcView 3.2 (ESRI, Redlands, CA). Available resource themes were reviewed for relevancy in estimating fire regimes. From these, four themes were selected: land type associations, fire-adapted vegetation, potential natural vegetation, and current forest types. All themes were converted to 20- by 20-m grids for applicability at the stand level (10's to 100's of ha). Selected features in the themes were given scaling values of 1 through 5 to represent vegetation fire adaptation, with 1 representing vegetation most adapted to fire and 5 the least. Weights were assigned to each theme according to its estimated influence or importance on fire regimes. Current forest type and potential natural vegetation were weighted equally and higher than other input themes. The resulting weighted scores were then categorized into fire regime groups based on ecological knowledge of fire-vegetation relationships. Fire regimes were assigned as follows: model ranking one = fire regime IV (35 to 100 years, stand-replacement severity), model ranking two = fire regime I (0 to 35 years, low severity), model ranking three = fire regime III (35 to 100 years, mixed severity), and rankings four and five = fire regime V (200+ years, stand-replacement severity). Fire regime II (0 to 35 years, stand-replacement severity) was not considered applicable to landscapes on the Monongahela.

[1]Forest Ecologist, Monongahela National Forest, 200 Sycamore Street,
Elkins, WV 26241, 304-636-1800, email: mthomasvangundy@fs.fed.us.

INFLUENCE OF PRESCRIBED FIRE ON STEM GIRDLING AND MORTALITY

Jeffrey S. Ward[1†] and Patrick H. Brose[2]

This study examined the response of individual stems to prescribed fire in the eastern oak (*Quercus* spp.) forest. Following seven prescribed fires in Connecticut, all stems at least 4.5 feet tall were tallied within thirty 45- by 45-foot plots. To increase the sample of stems with diameters of at least 2.5 inches, trees that were within 45 feet of each plot center but outside of the interior plots, also were measured. Diameters were measured to the nearest 0.04 inches for all stems with diameters of at least 0.4 inch. Heights were measured to the nearest 0.4 inch to a maximum of 80 inches. The fraction of each stem girdled by fire was recorded in increments of 10 percent. Trees with dead tops (topkilled) were recorded as 100-percent girdled. In all, 3,476 stems were examined. Temperatures were recorded during the fires with thermocouples at each plot corner and at plot center. Average maximum temperatures within plots was 360°F (range: 142 to 696°F). Average stem girdling was 95 percent or more for seedlings (< 0.4 inch diameter) regardless of species. Average stem girdling decreased by approximately 5 percent per 1 inch increase in diameter for larger stems. Less girdling was noted for oaks than for other species in the poletimber size class (4.5 to 10.5 inches diameter). Some sawtimber trees (> 10.5 inches diameter) also were injured by prescribed fire. Twenty percent of sawtimber oaks had at least 25-percent girdling compared with 28 percent for non-oak sawtimber. A similar pattern was noted for stems that were topkilled. There was no difference among species in the proportion of saplings (0.4 to 4.5 inches diameter) that were topkilled. Fewer poletimber and sawtimber oaks were topkilled than non-oak species. Topkilled oaks sprouted more frequently than non-oaks, 68 vs. 44 percent, respectively. However, nonoak sprouts were more numerous and taller. For all species, both girdling and topkill increased as the average maximum temperature of the prescribed fire increased. For example, 45 percent of trees in the 4-inch diameter class were topkilled when exposed to temperatures > 300°F compared with only 5 percent for stems exposed to temperatures < 300°F.

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[1]Chief Scientist, The Connecticut Agricultural Experiment Station, Department of Forestry and Horticulture; [2]Forester, USDA Forest Service, Northern Research Station; [†]Corresponding author, PO Box 1106, New Haven, CT 06504, 203-974-8495, email: jeffrey.ward@po.state.ct.us.

PUTTING FIRE ON THE GROUND WITH PRESCRIBED BURNING ASSOCIATIONS

John R. Weir[1†], Terrence G. Bidwell[2], and Chris M. O'Meilia[3]

A prescribed burn association is a group of landowners and other concerned citizens that form a partnership to conduct prescribed burns. Prescribed burning is the key land management tool used to restore and maintain native plant communities to their former diversity and productivity for livestock production and wildlife habitat. There are four excuses that are used by land managers when asked why they do not use prescribed fire: liability, no training or experience, not enough labor, and not enough equipment. Prescribed burning associations can be the answer to all of these excuses. To start an association, get a group of interested people together. Be sure to involve key landholders, agency personnel, and fire departments within the community. Associations already have been formed that have developed goals and guidelines that other groups can use. The association should incorporate so that all dues, donations, and gifts are tax deductible and the association is eligible for grants from public and private groups. Some local rural fire departments assist the associations during burning operations. This cooperative effort provides the association with added equipment and personnel and provides the fire department with training time, possible added income, and community service. The primary benefit of a prescribed burning association is its ability to influence politics. Also, the increased use of prescribed fire will influence the continued safe use of fire with future generations. At this time there are eight prescribed burning associations conducting fires in Oklahoma.

[1]Superientendent, OSU Research Range, OAES Field and Research Services Unit; [2]Professor and Extension Specialist, Rangeland Ecology and Management, OSU; [3]Biologist, US Fish & Wildlife Service;
[†]Corresponding author, 4922 South Coyle Road, Stillwater, OK 74074, 405-743-4714, email weir@okstate.edu.

FIRE RESEARCH ON THE RACCOON ECOLOGICAL MANAGEMENT AREA

Daniel Yaussy[1†], Todd Hutchinson[2], Matthew Dickinson[2], Joanne Rebbeck[3]

The Raccoon Ecological Management Area (REMA) consists of 16,000 acres of forest located on the unglaciated Allegheny Plateau in southeastern Ohio. The area was cutover to provide charcoal to the iron furnaces that operated during the mid 19th century. The overstory is dominated by oaks with abundant red maple in the midstory and understory. Research on the use of fire to restore a mixed-oak ecosystem was initiated on the REMA in 1995 with a grant from the USDA Forest Service's Ecosystem Management Program. Periodic and annual prescribed burns were compared with respect to their effect on flora and fauna. These low-intensity fires did not increase the light reaching the forest floor sufficiently to promote oak regeneration. The Joint Fire Science Program provided funding for a replication of the national Fire and Fire Surrogate study, which was installed on the REMA in 2001 as part of the Ohio Hills site. This ongoing study combines thinning from below and prescribed fire to provide varying light levels and oak competition control to promote mixed-oak ecosystem sustainability. Many additional investigations by Forest Service and university scientists have been added to the original experimental design, which increases insight into the effects of these management practices. The processes determining fire behavior and effects are being modeled with funds provided by the National Fire Plan. Fire monitoring technologies and fire effects models are being developed. A hydrology and ecosystem process model is being applied to the sites to provide fuel characteristics for fire modeling. A study is being established on the REMA in which herbicide will be used prior to harvesting under a shelterwood system to control the sprouting of the pole-sized oak competitors. Prescribed fire will be used to control new seedling competition 3 to 5 years following a major acorn crop. It is hypothesized that the combination of these management practices will promote sustainability of the mixed-oak ecosystem. The research of fire effects and behavior on the REMA have made it on of the premier sites for producing information on the management of mixed-oak ecosystems in the United States.

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[1]Forester, [2]Ecologists, and [3]Plant Physiologist, USDA Forest Service, Northern Research Station, [†]Corresponding author; 359 Main Road, Delaware, OH 43015, 740-368-0101, email: dyaussy@fs.fed.us.

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