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Proceedings of the 3rd Fire in Eastern Oak Forests Conference

Abstract

Contains 10 full-length papers and 12 abstracts of posters that were presented at the 3rd Fire in Eastern Oak Forests conference, held in Carbondale, IL, May 20-22, 2008. The conference was attended by over 200 people from a variety of groups, including federal and state agencies, nongovernmental organizations, universities, and private citizens.

Review Process

Each paper was peer-reviewed by two reviewers. The authors worked with the editor to revise their manuscripts, based on the reviewers' comments. Revised manuscripts were submitted to the Northern Research Station for technical editing and publishing. The authors are responsible for the accuracy and content of the papers. Reviews were provided by Mary Arthur, Patrick Brose, Timothy Carter, Joseph Charney, Steve Croy, Larry Hedrick, Bill Jackson, R. Andrew King, Dave Minney, Paul Nelson, David Peterson, Kevin Robertson, Tom Schuler, Stephen Shifley, Marty Spetich, and John Waldron.

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Cover Photos

Top: Mixed-oak forest after several prescribed fires; Photo by Robert Long, U.S Forest Service.

Middle: Prescribed fire in mixed-oak forest; Photo by Mike Broecker, used with permission.

Bottom: White oak (*Quercus alba*) seedling; Photo by Todd Hutchinson, U.S. Forest Service

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Proceedings of the 3rd Fire in Eastern Oak Forests Conference

May 20-22, 2008

Southern Illinois University, Carbondale, Illinois

Edited by

Todd F. Hutchinson
U.S. Forest Service, Northern Research Station

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POSTER ABSTRACTS

FIRE EFFECTS ON VEGETATION AND WILDLIFE

A REVIEW OF FIRE AND OAK REGENERATION AND OVERSTORY RECRUITMENT

Daniel C. Dey[1] **and Zhaofei Fan**[2]

[1]Research Forester, U.S. Forest Service, Northern Research Station, Columbia, MO
[2]Assistant Professor, Department of Forestry, College of Forest Resources, Mississippi State University
DCD is corresponding author; to contact, call (573) 875-5341 or email at ddey@fs.fed.us

Abstract.—Fire has played a prominent role in the history of oak in eastern North America, and it is useful today for promoting oak regeneration where competition with other woody vegetation is a problem and for managing savannas and woodlands. We spent the last century extinguishing wildfire from forests for good reason, but now we must spend some time relearning how to use fire as a tool for sustaining oak-dominated ecosystems. The use of fire to favor oak in forested settings where timber production is a goal is very different from its role in managing savannas and woodlands. Fire as a tool originated in wildlife management and then found application in savanna and woodland restoration. In the past 30 years, there has been increased emphasis on fire in silviculture research and forest management. This paper discusses the use of fire in the context of oak ecology and silviculture with emphasis on promoting oak regeneration and dominance. Highlights include: (1) fire interactions with acorns and oak seedlings, and advance reproduction; (2) fire-induced changes in stand structure and understory and implications for oak regeneration; (3) fire as part of the regeneration prescription; and (4) the role of fire in the process of oak recruitment into the overstory.

The future looks bright
for those unafraid to light,
but who know how to use fire
to get what they desire.
Dey 2008

INTRODUCTION

The history of fire in eastern North America has been brought to public attention (Pyne 1982, Williams 1989, Guyette et al. 2002), and fire's importance in promoting the distribution and dominance of oak species is now widely recognized (Abrams 1992). The lack of fire in modern times is listed as one of the major contributors to the successional replacement of oak by mesophytic hardwood species. These competitors are favored by increasing forest density and disturbances that create small forest openings the size of single to small groups of canopy trees (Nowacki and Abrams 2008).

In the East, fire is beginning to be used to (1) favor oak regeneration in an attempt to sustain oak forests; and (2) restore oak woodlands and savannas. Through research and operational use of fire in eastern forests, we are learning how to use fire to manage oak systems that occur over such a large and varied landscape. This paper reviews some of what we know about fire and oak ecology and silviculture. We discuss how fire may be used to favor oak regeneration and recruitment in forests, woodlands, and savannas. We review key fire effects on oak regeneration from acorn germination to seedling establishment and development. Fire in combination with thinning and regeneration harvesting is discussed in relation to managing stand structure with the goal of sustaining oak beginning with the regeneration process. Following regeneration, oak trees must be successful in competing through stand development to achieve the goal of sustaining oak in the mature overstory, a process we refer to as "recruitment." Together, regeneration and recruitment of oak are essential to sustain oak forests, woodlands, and savannas.

ACORN GERMINATION AND SEEDLING ESTABLISHMENT

Acorn germination in species of the white oak group (*Quercus* section *Quercus*) begins in the fall when the radicle emerges before the seed goes into winter dormancy. White oak germination is completed in the spring with development of the epicotyl. In contrast, red oak group species (*Quercus* section *Lobatae*) have embryo dormancy and require a period of cold stratification before germination in the spring. Regardless of species, acorns must maintain high seed moisture content to remain viable. The white oak group species require moisture content in excess of 40 percent to maintain seed viability (Korstian 1927). Red oak group species can tolerate greater seed drying and viability is good above 25-percent moisture content. In either case, seed desiccation during the winter is a major cause of seed loss in many ecosystems (Korstian 1927).

Acorns fall to the ground in autumn just before or during leaf drop. The covering of leaves helps maintain seed moisture content. In more northern climates, permanent winter snowpack creates an excellent environment for seed storage and stratification. At the ground-snow interface, temperatures and humidity are just right for acorns. Contact with, or burial in, mineral soil provides further protection against seed drying, and buried acorns may be less available to seed predators. Some small mammals and birds are key agents of acorn dispersal and burial, but along with insects and other wildlife species, they also consume much of the annual acorn crop in the process. Hence, seedling establishment is greatest in years of good to bumper acorn crops, when seed predators are overwhelmed with a bountiful seed supply.

Deep litter (> 2 inches) can be a physical barrier to epicotyl emergence from acorns located beneath the litter layer. Furthermore, radicles emerging from acorns lying on or mixed in thick litter over duff layers may be unable to penetrate into mineral soil before they dry out and die. Fire that reduces litter and duff layers to less than 2 inches thick before the autumn acorn and leaf drop favors the establishment of oak seedlings. The benefit of a fire removing litter and duff layers lasts for several years because it takes that long for these layers to reaccumulate. In undisturbed, mature oak-hickory forests of the Missouri Ozarks, Stambaugh and others (2006) estimated that it takes 4 years following a fire for the majority (75 percent) of the litter layer to reaccumulate to preburn levels.

Late autumn and early winter fires that burn after seed drop and leaf fall tend to remove the beneficial covering offered by the current year's leaves, thus promoting seed desiccation over the winter. In addition, the heat from a fire after acorn drop directly kills a high proportion of the seed crop. Auchmoody and Smith (1993) reported that a cool fall prescribed fire in northwestern Pennsylvania killed 40 to 49 percent of the acorns in the litter, and the combination of fire and acorn weevils lowered germination rates by 20 percent compared to unburned acorns. Dey (unpublished data) found that a spring surface fire in an Ontario northern red oak-maple stand reduced acorn germination capacity from 85 to 16 percent. Acorns buried in mineral soil are better able to survive surface fires because (1) they are insulated from the heat of surface fires by soil, which is generally a poor heat conductor when soil moisture is at field capacity or less; (2) low intensity fires, common in the East, are less likely to heat the soil to lethal temperatures; and (3) dormant-season fires occur when ambient temperatures and seed physiological activity are low.

Iverson and Hutchinson (2002) and Iverson and others (2004) measured soil temperatures during spring surface fires in Ohio oak forests and found that the greatest maximum temperature observed at one location was 82 °F at 0.4 inches deep in the soil even though maximum air temperatures 4 inches above the ground ranged from 325 to 600 °F. On average, they found that surface soil temperatures during fires reached only 45 to 51 °F, depending on fire frequency. These soil temperatures were insufficient to cause death to living organisms and plant tissue in the soil, including buried acorns and tree roots. If a good crop of acorns is on the ground, it is best to delay burning until seedlings are established and have grown large enough to respond by sprouting vigorously after burning. This process may take 3 years or more, depending on overstory density (i.e., degree of shading) and site quality (Brose 2008). Mechanical scarification may be an alternative to fire for breaking up deep litter layers and burying acorns. Both Rathfon and others (2008) and Lhotka and Zaczek (2003) found that mechanical scarification increased the density of oak seedlings that established after seed fall.

SEEDLING DEVELOPMENT

Fire damage to a tree's cambium, leading to death of the shoot or the entire individual, is related to temperature and duration of the tree's exposure to fire's heat. In general, plant tissue dies when it is exposed to ≥ 140 °F for 60 seconds (Hare 1965). Longer exposure to lower temperatures can also be lethal. Average maximum temperatures at or near the ground (i.e., within 10 inches) during spring (dormant-season) prescribed fires in hardwood forests have been recorded from 183 °F to 698 °F in the Missouri Ozarks (Dey and Hartman 2005), Ohio (Iverson et al. 2004, Rebbeck et al. 2004, Hutchinson et al. 2005, Phillips et al. 2007) and the southern Appalachian Mountains (Elliott and Vose 2005, Phillips et al. 2007). Clearly, dormant-season burns are capable of causing death or topkill in hardwood and pine seedlings.

In trees of any species, bark thickness is a major determinant of fire resistance (Hengst and Dawson 1994). A tree's fire resistance increases exponentially as bark thickness increases, and stem diameter has been shown to be significantly correlated to tree survival after fire (Loomis 1973, Regelbrugge and Smith 1994, Dey and Hartman 2005). Oak seedlings that are less than 3 years old suffer high mortality (>70 percent) after a single low intensity dormant-season fire (Johnson 1974, Dey and Parker 1996) in part because of thin bark and low root carbohydrate reserves. Further, because germination in oak is hypogeal, location of the acorn at time of germination, i.e., in the litter or buried in soil, determines to a large extent its ability to survive a surface fire (Brose and Van Lear 2004). The multitude of dormant buds located near the root collar, which give oak its great ability to sprout after death of the shoot, are at much higher risk of mortality when the acorn germinates in the litter than when it germinates an inch or two in the soil. Soil protects seedling roots and other tissues from lethal temperatures during dormant-season fires in Midwestern hardwood forests (Iverson and Hutchinson 2002, Boerner 2006).

Prescribed fires typical of those conducted in eastern North America may cause the death or, more commonly, shoot dieback of hardwood stems that are less than 5 inches in diameter (Reich et al. 1990, Waldrop et al. 1992, Barnes and Van Lear 1998, Brose and Van Lear 1998). A single fire is sufficient to kill the shoot in young, small-diameter hardwoods (Dey and Hartman 2005). Most hardwood species, however, respond to this damage by sprouting from surviving root systems, especially larger advance reproduction in the seedling and sapling size classes (Fig. 1). Only new and recent germinants are significantly more subject to complete mortality following low intensity fires regardless of species. Seedlings growing slowly in the dense shade of mature forest (typical on high quality, mesic sites) are vulnerable to fire mortality for a much longer duration than those growing rapidly in more open environments, where they reach fire-resistant sizes sooner. Factors other than tree size also play an important role in determining whether mortality occurs; for instance, fire intensity, seasonality, and frequency; and plant physiological activity (Whelan 1995, Debano et al. 1998, Dey 2002).

A tree's ability to sprout after the death or removal of the shoot increases exponentially with increasing stem diameter up to a threshold diameter, which varies by species (Fig. 1) (Dey 1991). As tree size increases beyond the threshold diameter, sprouting capacity declines (Fig. 2) (Dey et al. 1996a). Declines in sprouting capacity in larger and older trees are probably a function of (1) increased mechanical resistance to bud emergence as bark thickness increases; and (2) physiological senescence in mature and over-mature trees.

For similarly sized trees, mortality after a single fire varies by species, and oak and hickory species are more likely to survive than many of their common competitors, such as sugar maple (*Acer saccharum* Marsh.), red maple (*Acer rubrum* L.), black birch (*Betula lenta* L.), and yellow-poplar (*Liriodendron tulipifera* L.) (Kruger and Reich 1997, Brose and Van Lear 1998). Dey (2002) and Brose and others (2006) classified common eastern species

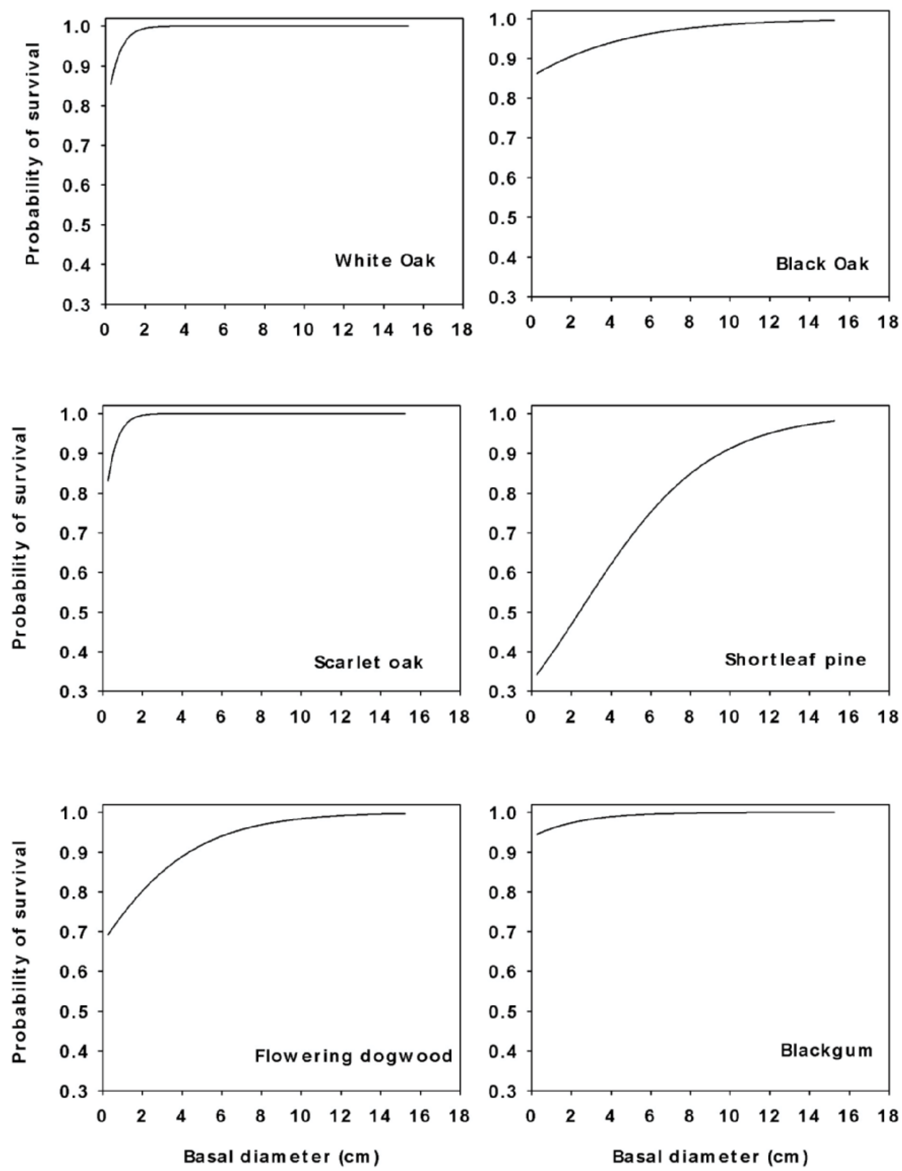

Figure 1.—The probability of advance reproduction's being alive one growing season after a spring (March-April) prescribed burn on The Nature Conservancy's Chilton Creek, MO, property (from Dey and Hartman 2005). Most stems were topkilled by the fire but produced multiple sprouts.

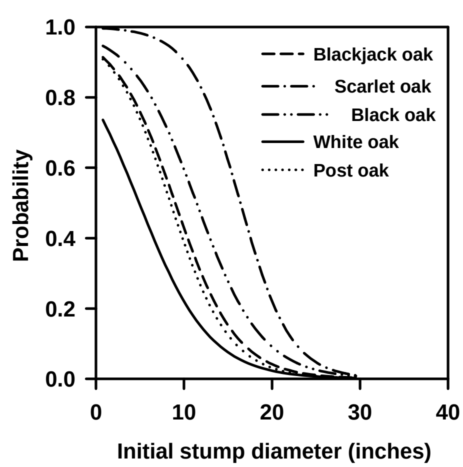

Figure 2.—Estimated probability of having a live stump sprout 5 years after clearcutting, based on initial stump diameter for common oak species in the Missouri Ozarks (from Dey et al. 1996).

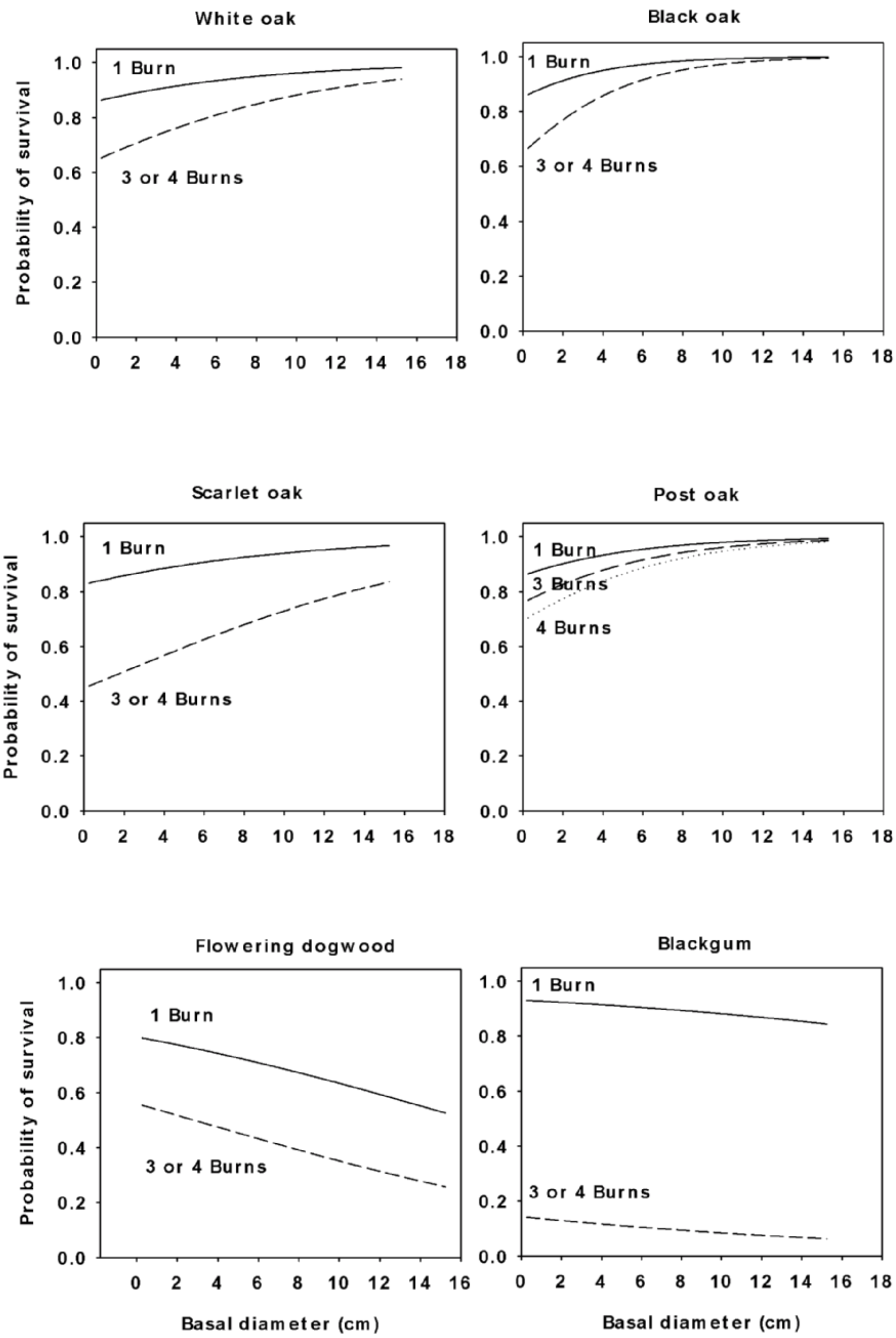

Figure 3.—Probability of advance reproduction's being alive either 4 years after one burn, 2 years after three burns, or 1 year after four annual burns in mature fully stocked Missouri Ozark forests (averaging 78 ft^2/ac basal area and 69 percent stocking) subjected to multiple spring prescribed fires (from Dey and Hartman 2005). Basal diameter of advance reproduction ranged from 0.1 to 6 inches for seedlings and saplings (up to 50 feet tall) before any burning.

Table 1.—Classification of relative ability to survive intact or persist through vegetative reproduction in a disturbance regime of frequent fires for common eastern tree species (from Dey [2002] and Brose et al. [2005]). (* species that are fire-sensitive when young but have greater ability to survive when mature; ** species that are fire-sensitive as older, larger individuals, but are prolific stump or root sprouters).

Very sensitive	Sensitive	Intermediate	Resistant	Very resistant
Balsam fir	American holly	American beech **	Aspen *, **	Bear oak
Eastern redcedar	American Basswood	Cherrybark oak	Black locust **	Blackjack oak *
Hemlock *	American beech **	Blackgum	Black oak	Bluejack oak
Northern white cedar	Aspen *, **	Blackjack oak *	Bur oak	Dwarf chinkapin oak
Red pine *	Black cherry	Nuttall oak	Chestnut oak	Longleaf pine
Virginia pine	Black walnut	Overcup oak	Chinkapin oak	Post oak
White pine *	Cottonwood *	Pin oak	Cottonwood *	Slash pine
White spruce	Elm	Pin cherry	Hickory	Turkey oak
Yellow-poplar *	Flowering dogwood	Red maple **	Northern pin oak	
	Ironwood	Scarlet oak	Northern red oak	
	Laurel oak	Southern red oak	Red pine *	
	Magnolia	Swamp chestnut oak	Sassafras **	
	Red maple **	Swamp white oak	Shortleaf pine	
	Sassafras **	Southern red oak	White oak	
	Silver maple	Swamp chestnut oak	White pine *	
	Striped maple	Swamp white oak	Yellow-poplar *	
	Sugar maple			
	Sweet birch			
	Sweetgum			
	Sycamore			
	Water oak			
	White ash			
	Willow oak			

by their ability to survive as adults or persist through basal stem and root sprouting in a disturbance regime of frequent fires (Table 1). Differences in mortality among species are more significant in the small seedling size classes and are related to ability to regenerate after frequent fire damage. Species differ in morphological and physiological adaptations that protect them from high temperatures during fires, or allow them to regenerate after suffering damage. The type and location of their reproductive structures (dormant buds) and seeds (chemical and thermal seed dormancy and longevity in the forest floor seedbank) determines whether they individually or as a species will survive and persist after fire. Capacity to produce biomass and allocation of carbon within the plant make some species' strategies more competitive than others in a regime of frequent fire. Oaks are survivalists in a world of fire.

One fire seldom causes long-term shifts in species composition (McGee 1979, Wendell and Smith 1986, Van Lear 1991, Van Lear and Waldrop 1991). Only after repeated fires (i.e., annual to every 5 years), whether applied before or after regeneration harvesting, do tree species begin dropping out of mature forest understories, or young stands. Oak reproduction is better adapted than many of its woody competitors because it has numerous dormant buds near the root collar, which is often located buried in the soil, protected from most surface fires. In addition, oak seedlings preferentially allocate carbon to the root system, even at the expense of shoot growth (Kolb and Steiner 1990, Walters et al. 1993). With sufficient light, oak advance reproduction is able to build a large root system with high carbohydrate reserves that supports rapid shoot growth following disturbances causing shoot dieback. This survival strategy is advantageous in environments that are subjected to drought, herbivores that feed on hardwoods, or fire that destroys the shoot. Where large oak advance reproduction accumulate through natural disturbances in "intrinsic oak accumulator systems," i.e., xeric oak forest (Johnson et al. 2002), they often occur in sufficient numbers to sustain oak dominance. In these forests, frequent fire often benefits oak preferentially by improving its competitive status in the regeneration layer (Brose et al. 2006) in part because oak seedlings are large enough to recover from intermittent fire and benefit from the temporary reduction in competition. In "recalcitrant oak accumulator systems," i.e., mesic and hydric forests, oak advance reproduction is unable to persist in the understory and has low capacity to replace the oak in the parent stand. In these forests, small oak seedlings are periodically numerous following a good acorn crop, but shear numbers of small oak reproduction are not enough to ensure success in oak regeneration. Young and small oak advance reproduction have low competitive capacity to regenerate and sustain oak, and they are vulnerable to mortality following intense or frequent fires.

Although frequent fires and high intensity fires may reduce the density of oak seedlings and saplings, oak's relative competiveness is often enhanced by burning because fire's effect on other species is more severe (Brose et al. 2006). The Santee Fire Study in South Carolina (Waldrop et al. 1992) nicely illustrates the tenacity of oaks as they outlasted their competitors after 43 years of frequent burning in summer or winter. In Coastal Plain loblolly pine (*Pinus taeda* L.) forests, it took up to 20 years of annual summer burning to practically eliminate hardwood advance reproduction in the 0- to 1-inch diameter at breast height (d.b.h.) class, and oaks were one of the last species groups to drop out. All woody vegetation less than 4.5 feet tall had been eliminated after 43 years of annual summer burning in these forests, but density of woody species was still high (e.g., 8,000 to 15,000 stems per acre for all species) under periodic or biennial summer or winter burning, and annual winter burning. Allowing 2 to 7 years between fires permitted the hardwood sprouts to continue building root biomass and increase in vigor, especially for the oak species growing under a mixed pine-hardwood overstory. Even with annual winter burning, oak sprouts had an entire growing season to recover from shoot dieback and replenish the supply of root carbohydrates before the next burn. Waldrop and others (1992) predicted that all fire treatments except the annual summer burning would sustain a hardwood midstory that would develop quickly upon cessation of burning, and oak would be a dominant species. Many other studies throughout eastern North America report on oak's persistence and improved competitiveness after long-term frequent burning (e.g., Dey and Hartman 2005).

In recalcitrant oak accumulator systems (Johnson et al. 2002), light levels in the understory are so low (e.g., 1 percent of full sunlight) that oak advance reproduction occurring after a good acorn crop remains small in size, and cohort survival over a 10-year period is typically low, e.g., less than 10 percent (Beck 1970, Loftis 1988, Crow 1992, Parker and Dey 2008). Burning forests with small oak advance reproduction often causes high mortality in oak seedlings and accelerates succession to shade-tolerant species. Shade-tolerant advance reproduction is able to grow to larger sizes in the understory than oak. This ability improves the likelihood that shade-tolerant reproduction will either survive intact or sprout in the shade after a single fire, or infrequent fires. Oak sprouts from surviving root stocks are unable to meet their respiration needs in such low-light environments. Oak species are intolerant to intermediate in shade tolerance, and require 30 to 50 percent of full sunlight for good growth (Phares 1971, Teskey and Shrestha 1985, McGraw et al. 1990, Ashton and Berlyn 1994, Gottschalk 1994, Brose 2008). Therefore, burning in mature, fully stocked stands on high quality sites is usually of little benefit to small oak advance reproduction.

FIRE AND STAND STRUCTURE

Removal of the midstory canopy in mature forests is one way to increase light to small oak advance reproduction. Prescribed burns typical of those conducted in eastern North America are capable of reducing the density of midstory trees (1- to 6-inch d.b.h.) in mature forests (e.g., Waldrop et al. 1992). Midstory density is reduced by fires that kill or topkill trees in that size class. However, density in the smaller size classes often increases dramatically after a fire because many of the hardwood saplings that are topkilled by the fire sprout. Height growth of these hardwood sprouts is slow in the understory of closed-canopy forests because of relatively

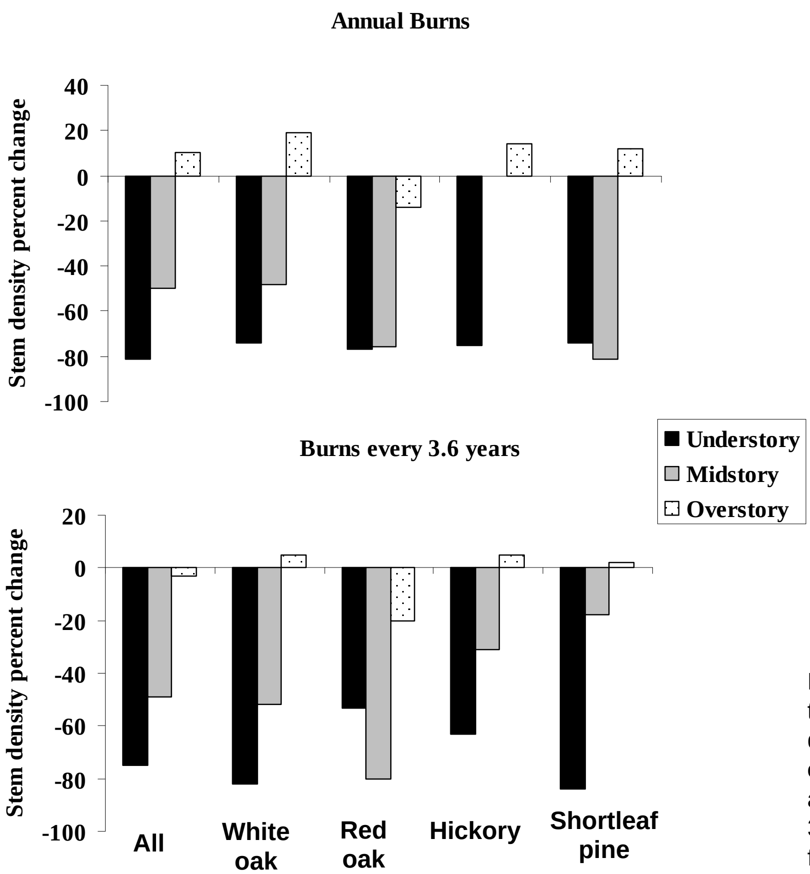

Figure 4.—Percent change in density of trees in the understory (3 feet tall and up to 1.5 inches d.b.h.), midstory (1.5 to < 4.5 inches d.b.h.), and overstory (≥ 4.5 inches d.b.h.) after 10 years of annual spring burns and frequent fires (every 3.6 years on average) in mature oak-hickory forests of the Missouri Ozarks.

low light levels. Four years after a single spring burn in a mature Ozark forest Dey and Hartman (2005) observed that height growth of flowering dogwood (*Cornus florida* L.) and white oak (*Quercus alba* L.) sprouts was slow, and 75 percent or more of the sprouts, some from saplings that were initially as tall as 20 feet or more, were still in the 3-foot height class. Frequent fires over 4 years kept practically all advance reproduction of any species in the smallest height class (1 foot). Thus, one spring fire was effective in reducing the midstory hardwood strata. No doubt understory light levels were increased to somewhere between 10 and 20 percent of full sunlight as has been reported in a number of studies (Lorimer et al. 1994, Miller et al. 2004, Motsinger 2006).

Under the intact closed canopy of the overstory (trees ≥ 4.5 inches d.b.h.), suppression of sprout height growth persists for a number of years, thereby negating the need for annual burning if the objective is to keep understory woody structure low. In mature Missouri Ozark forests, annual or frequent burning (mean fire return interval = 3.6 years) for 10 years did not improve light conditions beyond what was gained from one fire primarily because of its lack of effect on overstory density, i.e., dormant-season fires caused little mortality of overstory trees. However, these frequent spring fires caused 50-percent reductions in the density of midstory trees (i.e., ≥ 1.5 inches to < 4.5 inches d.b.h.) and 75 percent or more in understory trees (i.e., ≥ 3 feet tall and < 1.5 inches d.b.h.) (Figs. 4 and 5). Finally, the effect of surface fires on the midstory varied by oak species. Density reductions in midstory white oak species (40 percent) were half that observed in red oak species (80 percent) after 10 years of burning. In contrast, overstory density was unaffected, or even increased, over the 10 years of frequent burning when spring fires were of low intensity. Moderate to high intensity fires are needed to cause reductions in overstory density through tree mortality. These types of fires may be more appropriate in managing savannas and woodlands across larger landscapes (e.g., thousands of acres), but in forests a more direct and controlled way of managing overstory density and composition is through timber harvesting and stand thinning.

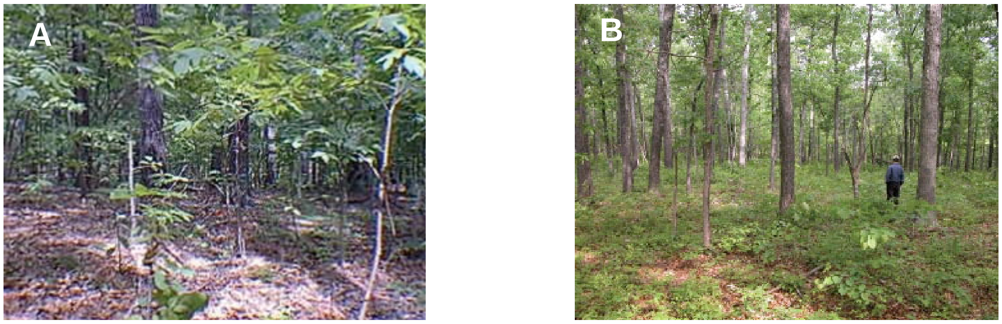

Figure 5.—Low intensity surface fires can alter forest structure in the understory of mature eastern hardwood forests by killing smaller advance reproduction and causing shoot dieback in larger seedlings and saplings. Shown in the photographs is a mature Missouri Ozark forest (A) before prescribed burning and (B) after 10 years of frequent burns averaging one fire every 3 years.

FIRE AND REGENERATION METHODS

Silvicultural prescriptions for regenerating oak should consider combining fire with regeneration methods such as clearcut, shelterwood, seedtree, or group and single-tree selection for managing forest structure to provide a desirable environment for oak reproduction development. Single fires are largely indiscriminant in the species they affect in the smaller size classes. Fire's ability to reduce stand density is stochastic, giving managers less control over species composition, stand structure, and stem distribution within the stand. Prescribed fires are often set in the winter or early spring because (1) burning windows are more common considering weather and fine fuel condition; (2) personnel are less likely to be needed to fight wildfires; and (3) fires are safer to conduct. Dormant-season burning limits the killing power of fire, which may be a plus or minus depending on the outcome managers desire. Fire alone is limited in how big a tree it is able to kill or set back, generally having less effect as tree diameter increases above 5 inches d.b.h.. Fuel loading must be increased or burning done in seasons with high burning indices to increase fire's ability to kill larger trees for reduction of stand density. For these reasons, other methods such as timber harvesting, and thinning by chemical or mechanical practices, are in many cases better suited than fire for managing the density of larger trees (i.e., > 5 inches d.b.h.) and composition of the mature forest overstory.

Light is a major limiting factor in oak regeneration (Abrams 1992, Lorimer 1993, Johnson et al. 2002). Managers control light available to reproduction by managing the various strata of forest vegetation. Reduction of the midstory, whether by mechanical cutting, herbicide application, or prescribed burning, may increase light levels to 10 to 20 percent of full sunlight, which is an improvement over the extremely low light levels (e.g., <5 percent of full sunlight) in mature forests, especially on productive, mesic sites (Lorimer et al. 1994, Barnes and Van Lear 1998, Miller et al. 2004, Motsinger 2006). Survival and growth of oak advance reproduction is improved by this increase in light, but oak reproduction requires 30 to 50 percent of full sunlight to achieve maximum photosynthesis and growth rates (Gottschalk 1987, Hanson et al. 1987, Gottschalk 1994).

Combining fire with one of the regeneration methods is a good way to provide enough light to maximize oak reproduction growth while controlling competing vegetation. Single-tree selection harvesting does not increase understory light much above that in uncut forests and is not often recommended for regenerating oak on mesic sites, or most other places outside of the xeric forests of the Missouri Ozarks (Johnson et al. 2002). A further complication is that fire may scar young trees, which remain for decades to grow to maturity before being harvested. Decay resulting from such scarring takes decades to progress in trees and may cause significant loss of sound volume and declines in tree grade and log quality. Group selection openings of sufficient size (e.g., opening diameter equal to one to two times the height of the dominant trees in adjacent stands) can be used to regenerate oak, but it is hard to apply fire to control competing vegetation within the group openings because the groups are isolated and scattered throughout the stand. Additionally, group selection is often done in combination with single-tree selection harvesting between the group openings.

Fire with clearcutting or shelterwood harvesting probably has the greatest promise for regenerating oaks. Clearcutting to regenerate oak in mixed hardwood forests is most successful when there is adequate regeneration potential from oak advance reproduction and stump sprouts to give some certainty that oak stocking desired at stand maturity will be met. Models for assessing oak regeneration potential based on success probabilities for advance reproduction and stump sprouts are available for some species and regions in eastern North America:

- Loftis (1990a) presented models for estimating performance of northern red oak advance reproduction in southern Appalachian clearcuts
- Loftis (1989), and Schweitzer and others (2004) are developing a regional regeneration model for the southern Appalachian Mountains and Cumberland Plateau
- Dey and others (1996b) developed ACORn for predicting stand size structure and composition for natural regeneration in upland oak-hickory forests in the Missouri Ozarks managed by even-aged methods
- Brose and others (2008) developed SILVAH for regenerating oak forests in the mid-Atlantic Region

Many other reports have been published on oak stump sprouting probabilities (Wendell 1975, Johnson 1977, McGee 1978, Lynch and Bassett 1987, Dey et al. 1996a, Weigel and Peng 2002, Weigel et al. 2006, Dey et al. 2008) and success probabilities for oak planted in shelterwoods and clearcuts (Johnson 1984, Spetich et al. 2002).

When oak advance reproduction is absent, a three-stage shelterwood is a good approach for regenerating and sustaining oak advance reproduction.

1) Fire can be used any number of times before a good acorn crop to begin reducing the density of competing species in the mid- and understory.
2) A light harvesting is done to release the crowns of potential seed-bearing oak trees and to remove the seed producers of competing species, especially species like yellow-poplar.
3) An establishment cutting may be done during a good acorn crop year to increase understory light levels and take the opportunity to manage competing vegetation, but fire should not be used after acorn drop.
4) Once seedlings establish, fire should be withheld until they are large enough to have good sprouting capacity after burning.

If adequate numbers of small oak advance reproduction are already present in the stand, then

1) The initial harvest of a two-stage shelterwood can be used to promote oak seedling growth by increasing the amount of light (Brose et al. 1999a, b).
2) Post-harvest burning may be considered 3 to 5 years later to give oak time to increase in size (e.g., 0.5 to 1.0 inches basal diameter).
3) Monitoring for free-to-grow oak will indicate whether a second burn is needed to control competing woody vegetation. The intent is to get the oak advance reproduction as big as possible before burning.

If fire is undesirable because mortality of small oak would occur, but competing vegetation needs to be reduced, then

1) Mechanical or herbicide control of the midstory and unmerchantable trees can be done in conjunction with shelterwood harvesting. Herbicide application must be done carefully to avoid killing the oak.

2) Higher shelterwood densities are recommended on high quality sites to help retard shade-intolerant competing vegetation (Loftis 1990b, Schlesinger et al. 1993). Yellow-poplar can overwhelm oak reproduction in clearcuts and low density shelterwoods on productive sites.

3) Brose and others (1999b) recommend that oak seedlings be 0.5 to 1.0 inch in basal diameter before burning is considered. Burning may occur at the time of the final shelterwood removal or 3 to 5 years afterwards. Oak does benefit from increased light after removing the shelterwood for a number of years, until crown canopy competition begins limiting light availability to oak.

There is much flexibility in designing a shelterwood prescription that promotes oak by meeting its resource needs while limiting development of competing vegetation. Success requires monitoring and adaptive management. A key ingredient of the regeneration prescription is the management of stand structure to provide adequate light to developing oak reproduction. Although not all oak species are the same in silvical requirements for good growth, most oak reproduction develops more quickly under a shelterwood when light is between 30 and 50 percent of full sunlight. Managers usually underestimate how much of stand stocking must be removed to increase light appreciably in forest understories. Based on a number of studies in widely divergent eastern forest types, 30 to 50 percent of full sunlight requires reducing stand stocking by more than 40 percent, basal area by more than 50 percent, or crown cover by more than 30 percent by harvesting from below and removal of any midstory layer if present (Godman and Tubbs 1973, Sander 1979, Leak and Tubbs 1983, Schlesinger et al. 1993, Schweitzer 2004, Gardiner and Yeiser 2006, Parker and Dey 2008).

Once an adequate density of large oak advance reproduction has developed, final shelterwood removal maximizes oak growth and probabilities of dominance throughout stand development. Retaining overstory trees at some point affects tree growth and eventually species composition of the regeneration. Miller and others (2006) found that clearcutting with reserves, leaving about 23 ft^2/ac in Appalachian mixed hardwood forests, caused 30- to 40-percent reductions in basal area of reproduction compared to similar stands that had been clearcut. They reported that reserve trees suppressed reproduction growth and shifted composition toward more shade-tolerant species. Larsen and others (1997) found that harvesting treatments that left overstory densities greater than 61 ft^2/ac, or 70 percent crown cover reduced oak seedling growth and survival in the Missouri Ozark forests. In the same region, Dey and others (2008) reported that single-tree selection harvesting that left 62 ft^2/ac, or 58 percent crown cover on average significantly reduced the growth of oak stump sprouts compared to those growing in clearcuts. Green (2008) observed decreasing densities of large oak reproduction as overstory basal area increased following regeneration harvesting including clearcutting and single-tree selection. Overstory basal areas above 50 ft^2/ac caused substantial reductions in numbers of large oak reproduction. Furthermore, significant reductions in oak reproduction height growth occurred under only 20 ft^2/ac compared to oaks growing in clearcuts.

Kabrick and others (2008) reported that single-tree selection harvesting in the Missouri Ozarks significantly suppressed the density of oak reproduction (sized ≥ 3.3 feet tall and up to 1.5 inches d.b.h.) compared to clearcutting. Moreover, black oak (*Quercus velutina* Lam.) and scarlet oak (*Quercus coccinea* Muenchh.) regeneration were prominent only in clearcuts. Arthur and others (1998) also noted that openings much larger than multiple tree canopy gaps in oak-pine forests in the Cumberland Plateau were needed for black and scarlet oak regeneration to be successful. They reported that two dormant-season surface fires separated by 2 years and a spring wildfire in mature, ridgetop oak-pine forests resulted in mortality of 16 and 20 percent in the overstory (i.e., trees > 4.0 inches d.b.h.), respectively, causing single-tree and multiple-tree gaps in the canopy. These small to mid-sized gaps were insufficient alone for regeneration of the shade-intolerant oak species.

RECRUITMENT INTO THE OVERSTORY

Regeneration by even-aged methods can be managed to favor oak using prescribed burning to modify competitive relations. Fire can easily topkill stems below 5 inches d.b.h., and repeated fire favors oak species over their competitors. Young stands can be kept in a perpetual state of stand initiation with frequent fire. Eventually, however, fire must be withheld for forest development to proceed if the goal is a mature forest. Once stands reach crown closure, it is probably preferable to manage oaks and competing species by mechanical or chemical stand thinning rather than prescribed fire. To promote recruitment of oak into the overstory, oaks should be initially free to grow, unencumbered by overly dense residual overstories. Fire-free periods are needed to allow for oak trees to grow beyond the sapling size class and develop enough diameter, bark thickness, and height to increase the probability that they can avoid shoot dieback if a fire should occur again.

In Missouri, the average diameter growth for codominant white oak saplings (1 to 3 inches d.b.h.) is 1.5 inches in 10 years (Shifley and Smith 1982). Therefore, a 33-year fire-free period is needed to allow such a white oak to grow to 5 inches d.b.h. and have a better chance of surviving a fire intact. Shifley and Smith (1982) found that average growth rates for red oak species were similar. An important point to note is that half of the trees in their analyses grew at rates greater than the average, assuming a normal distribution in oak diameter growth, and the upper 5 percent of oak trees had significantly greater diameter growth, which would lessen the time needed to develop thicker bark and greater fire resistance. It then becomes a matter of having sufficient numbers of large, fire-resistant oak to meet management goals for oak stocking at maturity.

The fire-free period may not need to be so long for oak stump sprouts, which exhibit an accelerated growth rate for some time after initial sprouting. Dey and others (2008) reported that oak stump sprouts in Missouri Ozark clearcuts grew to an average of 3.1 inches for scarlet oak, 2.3 inches for black oak, and 2.2 inches for white oak in 10 years. If the objective is to use fire to regenerate oak-dominated forests, then once the oaks are free to grow, burning does not have to be considered again until the period when the forest manager needs to create advance oak reproduction for the next rotation, which could be two to three decades before the final regeneration harvest. In woodland or savanna management, periodic recruitment of oak is necessary to sustain the desired overstory density. Because trees of white oak species may live to be 250 to 400 years old, and 80 to 200 years old for red oak species, oak recruitment need only occur occasionally in savannas and woodlands. However, when it is needed, there must be a sufficient fire-free period.

Newly regenerated hardwood stands in eastern North America have thousands of trees per acre, but stand density rapidly declines with time, especially after crown closure (Miller et al. 2007). The oaks that express dominance early have the best, or only, chance of surviving to have a place of prominence in the upper canopy crown classes at maturity (Ward and Stephens 1994). Competition is severe on high quality sites, and oaks have little chance of maintaining dominance without some type of management intervention (Hilt 1985), such as crop-tree thinning applied when the stand reaches crown closure (Miller et al. 2007).

FINAL THOUGHTS

Efforts to use fire to sustain oak-dominated forests, and to restore oak woodlands and savannas, have increased over the past 20 years, especially in the Great Plains-Eastern Forest ecotone. Historically, fire promoted the invasion of prairies into eastern forests as far as Ohio, creating the Prairie Peninsula Region (Transeau 1935). During this time, Native Americans had a great effect on the distribution of forest and prairie in the East through their use of fire. An eyewitness account by Black Hawk, a Sauk warrior in the early 1800s, testifies to this influence. In 1832, Black Hawk led his people in a fight to secure their homeland in northern Illinois and southern Wisconsin from settlement by frontiersmen and their families. Defeated in battle, Black Hawk and a small group of chiefs and leaders were transported east to meet with President Andrew Jackson. They were taken on a tour through Philadelphia, Baltimore, and New York

City to impress upon them the strength of the United States and the futility of further fight with the invading European settlers. In New York the Indian delegation viewed a fireworks display, of which Black Hawk noted in his autobiography originally published in 1833:

> *The chiefs* [U.S. military and political leaders] *were particular in showing us every thing that they thought would be pleasing or gratifying to us. We went with them to Castle-Garden to see the fireworks, which was quite an agreeable entertainment–but…less magnificent than the sight of one of our large prairies would be when on fire.*

Fire was not merely an entertainment spectacle for the Indians; they used it to manage their lands to provide for the needs of the people. The Sauk and neighboring tribes used fire to cultivate large (e.g., 800-acre) crop fields of corn and other vegetables and manage native prairies as pastures for their horses and wild game (Black Hawk 1833).

In modern times in the East, fire is still largely an anthropogenic phenomenon, but we suppress fires before they burn much land, and this action has greatly altered the fire regimes promulgated by the Native Americans. The ill effects of fire on forests from a timber perspective and the shadow of Smokey Bear still have a strong influence on the use of fire in forest management. Today, our understanding is improving in how fire:

1) acts on plants through biological processes,
2) modifies a species' regeneration and development through ecological processes,
3) changes the competitive relations among species, and
4) affects long-term site productivity and ecosystem integrity by modifying physical properties and biological components of the site.

We are quickly moving away from the simple approach of "return frequent fire to the system and all will be well." We are moving toward a more realistic and holistic management system approach by combining prescribed fire with other silvicultural practices to manage forest regeneration and succession within an ecological classification system to achieve well defined objectives desired by society. As in the past, the ecosystem goods and services we desire today guide the role of fire in forest plans and silvicultural prescriptions. Through research, monitoring, and adaptive management, we are exploring new ways to use fire judiciously in managing oak forests for a multitude of benefits including timber production.

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EFFECTS OF OVERSTORY STAND DENSITY AND FIRE ON GROUND LAYER VEGETATION IN OAK WOODLAND AND SAVANNA HABITATS

John B. Taft[1]

[1] Plant Ecologist,Illinois Natural History Survey, Champaign, IL
JBH is corresponding author; to contact, call (217) 244-5046 or email at taft@inhs.uiuc.edu

Abstract.—Vegetation changes underway in oak woodland and savanna communities in the eastern and midwestern United States, primarily a result of reduced fire frequency or fire absence, include increased tree density and shading and loss of species diversity in the ground layer. However, some habitats, particularly on dry-mesic to xeric sites, retain considerable restoration potential, and insights are needed for directing management efforts, such as with prescribed fire, where they can be most effective. This paper examines overstory and ground layer interactions in flatwoods, barrens, and dry-mesic woodland habitats, together with fire effects, and highlights structural characteristics relevant to ecological condition and restoration potential.

In nested vegetation sample plots throughout Illinois, patterns of ground layer diversity were strongly related to levels of tree density at several study sites. At these sites, with increasing tree density, or "thicketization," the diversity of ground layer species and plant functional groups declines. Where this pattern was lacking suggests two possibilities: tree density has yet to become a limiting factor for ground layer species and functional group diversity, or, more typically, tree density has progressed to the point where remaining ground layer species are sparsely distributed, shade-tolerant, and relatively indifferent to levels of tree density. The pattern of ground layer species attrition with increasing thicketization is interpreted as a signal for productive restoration opportunities compared to the latter, low diversity condition. Efforts to restore these former communities with reintroduction of fire have shown promise. Use of fire on sites even with extensive thicketization seems to improve overstory structure and ground layer diversity; however, in these cases, the soil seed bank may be limited following prolonged periods of stand closure.

Interactions between overstory and ground layer vegetation also were examined for ordered patterns of plant functional group losses (community disassembly) with increasing tree density. Documenting ordered patterns along the structural gradient of tree density provides additional insights regarding ecological condition and restoration potential of sites or habitats. Functional groups that appear most sensitive to thicketization and are the first to decline with extended fire absence are herbaceous vines (relatively scarce), perennial grasses, sedges, and perennial forbs. While opportunities remain to restore aspects of oak woodland and savanna habitats, the attrition pattern suggests a limited time frame for the full restoration of ground layer diversity, even in dry to xeric sites.

INTRODUCTION

Oak-dominated savanna, barrens, and woodland habitats occur throughout North America but have undergone extensive habitat loss and degradation, particularly as a result of declining fire frequency and fire absence, and are considered imperiled ecosystems (Anderson et al. 1999, Abrams 2005). Restoring these communities has become a priority for conservation agencies and habitat managers. The prairie-forest transition zone (Anderson 1983), a region centered on Illinois and overlapping with the leading edge of the "prairie peninsula" (Transeau 1935), provides opportunities for testing the restoration potential of oak woodland and savanna-like habitats. While about 55 percent of the landcover of Illinois was prairie in the early 1800s (Szafoni et al. 2002), the remaining lands primarily were forest, woodland, and savanna. These vegetation types formed a heterogeneous mosaic; wooded habitats were most prevalent in dissected terrain and prairie predominated where slopes were less than 4 percent (Anderson 1991). Fires that contributed to the maintenance of prairie also maintained the structural and compositional characteristics of savanna and woodland habitats (Nuzzo 1986, Taft 1997, Anderson and Bowles 1999). This fire history, while seemingly in the distant past, is within the life-span of some long-lived trees still standing (e.g., bur [*Quercus macrocarpa*], white [*Q. alba*], and post oaks [*Q. stellata*]). The open-grown form of many of these veteran trees serves as a reminder of the more open structure of woodlands and savannas in the pre-settlement period. In some cases, burning of woodlands was a practice that continued into the early 20th century, when it was strongly discouraged and disparaged as a "savage custom" (Miller 1920). However, once fire frequency declined, many of these stands quickly underwent closure, or "thicketization" (Archer et al. 2004, Breshears 2006), resulting in conversion of prairies and savannas to woodlands and then forests.

Some savanna and woodland habitats in the prairie-forest transition zone on seasonally dry sites such as flatwoods and barrens demonstrate a degree of resistance to change during long fire-free intervals. The great majority of floristic diversity in these habitats is in the ground layer (Taft 1997); consequently, they provide an opportunity to examine the patterns of influence of overstory structure on ground layer composition and diversity for evidence of interactions that may be absent in less ecologically stressful habitats, where extended periods of fire absence have resulted in prolonged ground layer shading (e.g., Hutchinson et al. 2005).

Interactions between overstory and ground layer vegetation in forests and woodlands can be reciprocal, with overstory influencing ground layer and ground layer influencing successful regeneration among tree species (Gilliam and Roberts 2003). Using flatwoods, barrens, and dry-mesic woodlands as examples, this paper will focus on how overstory structure in woodland and savanna-like habitats is related to and possibly affects patterns of ground-layer species diversity and composition. Maintaining a species pool at a given habitat can be influenced by both regional and local processes and involves a balance of processes at each scale (Huston 1999). Local factors that tend to decrease diversity include competition (Huston 1994), such as from woody overstory species in savanna-like habitats (Peterson and Reich 2008), an effect that could be ameliorated by regional interactions promoting species exchanges among habitats (Ricklefs and Schluter 1993). However, habitat fragmentation would tend to limit species exchanges while ecological islands such as dry to xeric habitats also would limit the types of species that could become established. Consequently, it is expected in the highly fragmented prairie-forest transition zone (e.g., Schwartz 1997) that with thicketization, especially in dry to xeric sites, the interaction between woodland/savanna overstory and ground layer vegetation would be manifested as a pattern of species loss, or attrition, in the ground layer. Ground layer species in forest habitats also can be very diverse (Gilliam 2007); however, forest species generally adapted to mesic conditions could not be expected to compensate for attrition of species in closed dry savanna and woodland habitats, particularly in fragmented landscapes.

Under the framework in the current study, situations where tree density and ground-layer species diversity are found to be inversely related would provide a signal, perhaps a finite opportunity, of restoration potential and an indicator where management with prescribed fire could be focused. A significant regression of tree density on ground layer diversity ($P < 0.05$) will be referred to as fitting the post-fire species attrition hypothesis. This hypothesis is based on the concept of plant functional groups, which are assemblages of species with a shared set of attributes that may include morphological and physiological characteristics, ecological roles, resource use, and response to disturbance (Symstad 2002). Diversity in functional groups has been shown to be closely linked with ecosystem functioning and stability, and can serve as an indicator of community integrity (Hooper and Vitousek 1997, Tilman et al. 1997, Mason et al. 2003). Monitoring the density of plant functional groups shows promise as a rapid assessment technique in prairies for evaluating habitat integrity, as it is highly correlated with species richness and estimates of floristic integrity (Sivicek 2007). It is expected that the diversity of plant functional groups would decline with stand closure. This pattern will be referred to as the post-fire functional group attrition hypothesis.

Where such a pattern is lacking would suggest one of two null hypotheses: 1) relatively stable condition where

Table 1.—Summary of habitat type, sites name, sample intensity (0.05-ha circular tree plots), and fire history for sites examined in this study. Most data are baseline, preburn data. INAI = Illinois Naural Areas Inventory sites.

Community Type[a]	Site name	Plot #	Baseline fire treatments	Citations
Flatwoods (INAI)	Reckers Woods	8	none	Taft et al. 1995
Flatwoods (INAI)	Chip-O-Will Woods	8	none	Taft et al. 1995
Flatwoods (INAI)	Posen Woods	8	none	Taft et al. 1995
Flatwoods (INAI)	Wms Creek Woods	8	none	Taft et al. 1995
Flatwoods (INAI)	Jackson Slough	8	none	Taft et al. 1995
Flatwoods (INAI)	Lake Sara Flatwoods	10	20 yrs, annual	Taft et al. 1995
Flatwoods	Mt. Vernon	23	none (preburn)	Taft 2005
Dry Barrens (INAI)	Gibbons Creek	15	none (preburn)	Taft and Solecki 2002
Dry Barrens (INAI)	Forest Service	8	none	Taft and Solecki 2002
Woodland[b]	Beaver Dam SP	12	1-3 burns	Taft 2006
Woodland[b]	Beaver Dam SP	15	none	Taft 2006

[a] Community classification follows White and Madany (1978)
[b] Community classification follows White and Madany (1978), with revisions.

tree density has yet to limit ground layer diversity; or, more typically 2) the progression of thicketization to the point where remaining ground layer species are shade-tolerant and indifferent to levels of tree density. The differences in these alternative null states can be readily distinguished by evidence of thicketization and the extent of herbaceous species in the ground layer vegetation. Sites fitting the second null model would be densely shaded stands with few herbaceous species, a predominance of woody species, and substantial bare ground. Although quite rare, null condition 1 would be optimal in open-woodland and savanna habitats; the negative attrition profile would indicate greater opportunities for restoration compared with null condition 2. The attrition pattern would suggest an existing heterogeneous stand structure with gaps containing more and different ground layer species (e.g., Leach and Givnish 1999, Peterson and Reich 2008) compared with areas with greater tree density. Consequently, with the attrition pattern, the species pool retains shade-intolerant taxa that are responding, albeit negatively, to stand thicketization. Attempting restoration at closed stands where these species are absent would rely on diversity found off-site (limited by habitat fragmentation and ecological condition) or among the selected species residing in the soil seed bank that could be stimulated with fire treatments.

METHODS AND DATA SOURCES

Vegetation Sampling

Data for the following habitat descriptions and analyses were collected employing similar vegetation sample methods involving stratified-random distribution of nested sample units with 0.05-ha circular plots for trees (≥ 6 cm diameter at breast height [d.b.h.]), 0.005-ha subplots for shrubs and saplings (> 50 cm tall), and 12 to 25 ground-layer sample quadrats (primarily 1/4-m^2; one study [Taft 2005] with ½-m^2 quadrats) on a transect within each tree plot. The data come from several sources (Table 1). Habitat characterization for flatwoods is based on a quantitative survey of vegetation and soils at 50 plots (0.05-ha) sampled among six sites on the Illinoian till plain (Taft et al. 1995). All six sites are recognized by the Illinois Natural Areas Inventory ([INAI], White 1978) as high quality habitats. Data from an additional site, Mt. Vernon Flatwoods ([MVF], Taft 2005), also are used. Data for barrens were recorded from a pair of sites in southern Illinois monitored since 1989 as part of a fire effects study (Taft 2003) and include more recent unpublished data. The INAI recognized the fire treatment site, Gibbons Creek Barrens (GCB), as a high quality barrens remnant. The fire-free reference site, Forest Service Barrens (FSB), was similar in many ways to the fire treatment site (Taft and Solecki 2002).

Data from dry-mesic woodland habitat were collected from six management units at Beaver Dam State Park (BDSP) in central Illinois, including three units with and three without recent burns as part of a fire management program (Taft 2006).

Habitat Descriptions

Dominant species and summary parameters for canopy, shrub/sapling, and ground layer strata are shown in Appendix 1, combining data from flatwoods (Taft et al. 1995), barrens (Taft and Solecki 2002), and woodland habitats (Taft 2006). Dominance is based on importance values calculated as the sum of two relative values (IV 200): basal area and density for trees, frequency and density for saplings and shrubs, and percent cover and frequency for ground layer species, including woody plants to 50 cm tall. The top-ranking 10, 15, and 50 species from canopy, shrub/sapling, and ground layer strata, respectively, are shown. Many species in the woody strata are present across habitats. By contrast, as an indication of dissimilarity among habitats in the ground layer stratum, only two species (Virginia creeper [*Parthenocissus quinquefolia*] and poison ivy [*Toxicodendron radicans*]) were found in all three habitats. Botanical nomenclature follows Mohlenbrock (1986) and community classification follows that developed for the Illinois Natural Areas Inventory (modified from White and Madany 1978). Eleven plant functional groups were recognized for the ground layer species in these habitats: ferns, annual/biennial forbs, herbaceous vines, legumes, perennial forbs, C3 perennial grasses (cool season), C4 perennial grasses (warm season), perennial sedges, shrubs, trees (seedlings to 50 cm), and woody vines.

<u>Flatwoods</u> - The flatwoods included in this study also are termed “southern flatwoods” (White 1978) or “post oak flatwoods” due to the predominance of post oak, a species accounting for about 57 percent of the IV. Oak and hickory species combined account for 93 percent of the IV in the canopy, but were relatively scarce in the shrub/sapling stratum. The shrub/sapling layer had just over 4,000 stems/ha (Appendix 1), an indication of thicketization.

Flatwoods are level woodlands (≤ ~2-percent slope) with claypan soils. Claypans (argillic horizons) are subsurface soil horizons characterized by a sharp increase in clay content compared to overlying soil horizons. Claypans can result in a perched water table in the spring; however, with evapo-transpiration during summer months, the surface soil horizons typically become very dry. These woodlands formerly were widespread on the Illinoian till plain (Telford 1926, Braun 1936, Aldridge and Homoya 1984) but have been reduced by extensive logging and land-use changes (Taft et al. 1995).

Flatwoods remnants can be instructive for examining overstory-ground layer interactions because variance in overstory structure and composition ordinarily attributable to degree of slope and aspect is reduced to variation in edaphic characteristics, such as soil nutrient content, depth to claypan, and soil available water holding capacity (AWC). For example, total density of woody stems is positively related to depth to claypan and particularly to soil AWC (Taft et al. 1995). Sites with more mesic soil conditions will be the fastest to undergo vegetation changes with fire absence and perhaps the most challenging to restore. Accordingly, stability in overstory structure and composition varies widely among flatwoods remnants with some demonstrating both structural and compositional instability, the typical condition for many oak woodlands (Abrams 1992, Moser et al. 2005). In contrast, some stands demonstrate compositional and structural stability with apparently sustainable levels of post oak regeneration. Differences can be traced to variance in soil parent materials. Unstable sites with poor oak recruitment and prolific regeneration of invading tree species tend to be found on relatively deeper soils with greater AWC. These tend to be soil types developed in loess deposits with a silty-loam texture. Structurally stable sites have been found on lacustrine deposits with considerable sand content in the surface soil horizons (about 40-50 percent) and, consequently, lower AWC (Taft et al. 1995).

At MVF (not included in Appendix 1), an old second-growth flatwoods, white oak was co-dominant with post oak. Tree density was 465/ha (≥ 6 cm d.b.h.), basal area was 24.7 m^2/ha, and sapling/shrub density was just over 7,000 stems/ha (Taft 2005), demonstrating considerable thicketization. Ground layer vegetation was very sparse and dominated by woody seedlings and vines; Virginia creeper, poison ivy, sassafras, riverbank grape (*Vitis riparia*), Mayapple (*Podophyllum peltatum*), and black oak accounted for more than half of the species occurrences. No grasses and only a single occurrence of one sedge was found in baseline samples.

Barrens - Barrens is a term applied to a wide range of physiographic and environmental conditions to describe distinctly different types of vegetation (e.g., Hutchison 1994, Bowles and McBride 1994). As the term is used in forested regions of the Midwest, barrens typically refer to a savanna-like community characterized by local assemblages of a prairie flora (White and Madany 1978). Today, barrens remnants occur primarily in dissected terrain on south- and southwest-facing aspects with shallow soils over bedrock (Heikens and Robertson 1995). During extended periods of fire absence, barrens tend to convert to closed woodland.

Baseline data from a pair of dry barrens in southern Illinois examined for a long-term fire effects study indicate that post oak was the dominant tree species, accounting for about 55 percent of the IV for trees (Taft and Solecki 2002). These sites, GCB (the fire treatment site) and FSB (a fire-free reference site), occur on shallow soils developed in loess and residuum over sandstone. Oak and hickory species combined accounted for 77 percent of the IV for trees. The size class distribution pattern for trees indicated compositional stability because post oak likely will continue to be dominant; however, prolific regeneration of post oak forming a strong reverse-J-shaped curve indicates a structurally unstable site and portends stand closure. Stem density for saplings and shrubs was just over 5,000 stems/ha, indicating thicketization was well underway (Appendix 1, pg. 38).

Dry-Mesic Woodland - Woodland habitat at BDSP, Macoupin County, IL, occurs primarily on rolling, dissected glacial moraines. Soils are developed in glacial till deposits and generally have a silty-loam texture (Tegeler 2004). Fire management units at Beaver Dam State Park included woodlands with one to three recent burns as well as unburned units. Oak and hickory species accounted for 49 percent of the total IV. Oak species were dominant in terms of basal area, but the high density of slippery elm (*Ulmus rubra*) and hackberry *(Celtis occidentalis*) made them prominent species as well. Overstory regeneration generally was a pattern of instability with little successful oak recruitment. Total shrub/sapling stem density was just over 14,000/ha, indicating advanced thicketization. Previously burned units had slightly lower tree density but slightly higher tree basal area compared to unburned units, much lower stem density in the shrub/sapling stratum, and higher ground-layer species density and diversity.

Perennial forbs are the most species-rich functional group in the ground layer of flatwoods, barrens, and woodland habitats in this study. However, the profiles of plant functional group importance in these habitats show a shift from woody vine dominance and near-equal secondary dominance of forbs, sedges, and grasses in flatwoods, to graminoid (grass and sedge) dominance in barrens and forb dominance in woodlands (Fig. 1).

DATA ANALYSIS

Data from canopy cover estimates made with a spherical densiometer and overstory structure were recorded in both flatwoods and barrens habitats. Correlations between canopy cover and total density of trees were positive, particularly in flatwoods ($r = 0.57$). However, tree density explained more of the variance in ground layer vegetation than did canopy cover ($r = -0.67$ vs. $r = -0.44$). Tree basal area was unrelated to ground layer diversity patterns in flatwoods, barrens, and woodland datasets. Consequently, tree density is used to examine the interactions between overstory and ground layer vegetation.

Interactions between overstory tree density and ground layer species and functional groups were examined with regression analysis; mean comparisons of tree density classes and ground layer parameters were done with ANOVA and Tukey post-hoc tests (SPSS ver. 10, Chicago, IL). Tree density classes of high, medium, and low levels were created for each habitat to examine trends among ground-layer functional groups. Tree density classes were formed by dividing the range of tree densities into equal thirds; in each case, mean differences

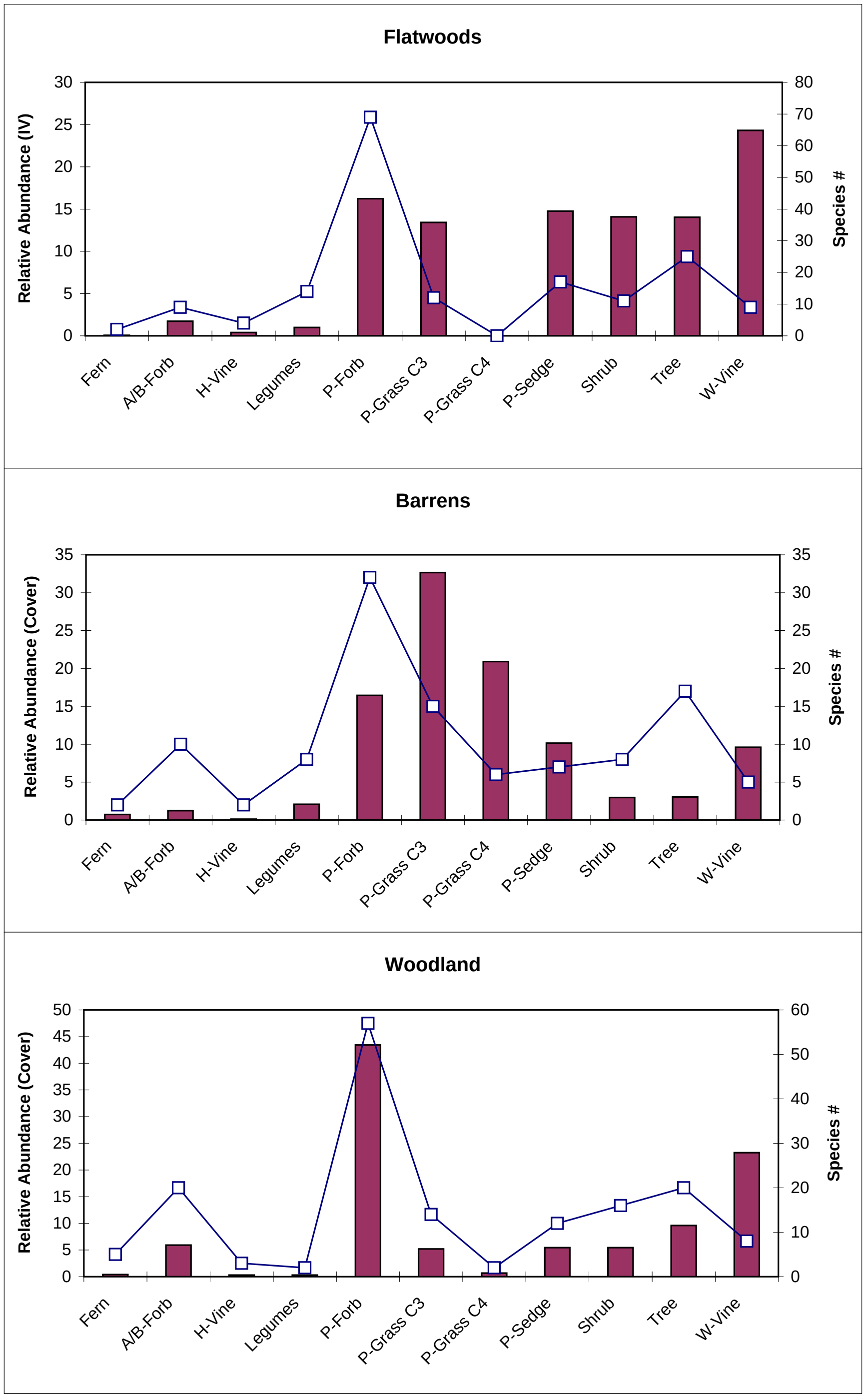

Figure 1.—Relative abundance (bars [left y axis]) and species richness (line and point data [right y axis]) of plant functional groups from vegetation sample data in flatwoods, barrens, and woodland habitats in Illiniois. A/B = annual/biennial, H = herbaceous, P = perennial, W = woody. Quadrat numbers for each community type were: flatwoods - 1,080, barrens - 276, and woodlands - 312.

in tree density classes were significant overall and with all pairwise Tukey post-hoc comparisons. Species density in the ground layer stratum is the average number of species per quadrat (generally, ¼ m^2). Species diversity is the Shannon-Wiener Index (H'). Functional group richness is calculated as the total number of plant functional groups in ground layer quadrat samples nested within tree plots.

RESULTS

Overstory-Ground Layer Interactions

Regression analysis of the interaction between tree density and ground layer diversity for all sample data combined from flatwoods, barrens, and woodland habitats showed no pattern. However, interactions were found between overstory and ground layer strata when individual community types were examined separately.

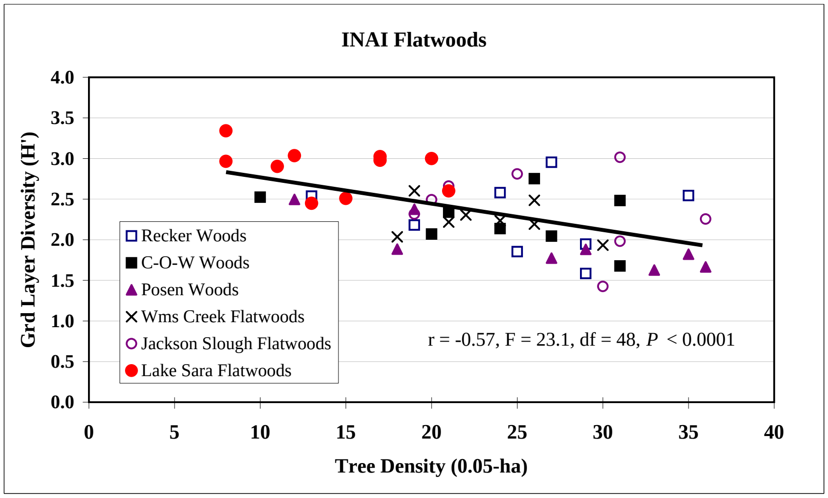

Figure 2.—Interaction between overstory tree density and ground layer diversity fitting the species attrition hypothesis for six flatwoods recognized by the Illinois Natural Areas Inventory (INAI). Ground layer diversity is the average Shannon-Wiener Index (H') from 12 (Lake Sara) to 25 quadrats for each tree plot. Tree density is based on stems ≥ 6 cm d.b.h.

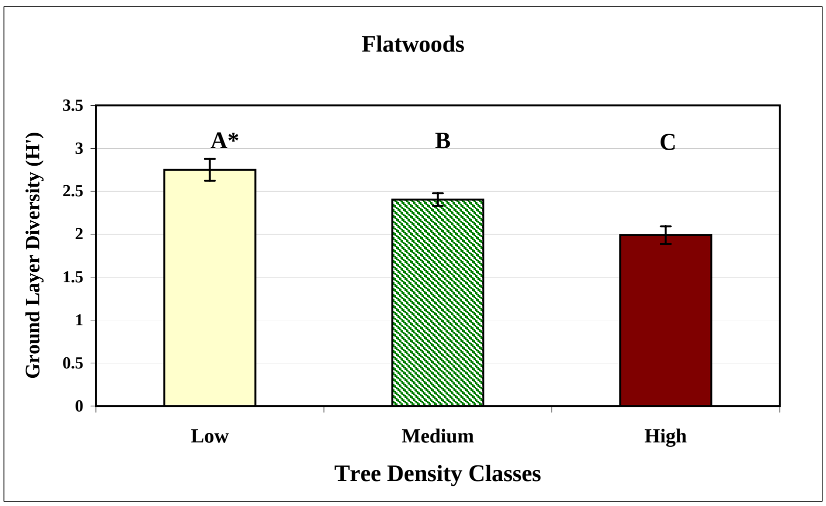

Figure 3.—Mean comparison of ground layer diversity (Shannon-Wiener Index [H') among tree density classes for six flatwoods recognized by the Illinois Natural Areas Inventory. The overall differences are significant (ANOVA: F = 11.6, df = 2, P < 0.0001); different letters indicate significant pairwise comparisons with Tukey post-hoc test (*difference between low and medium tree density is marginally significant [P = 0.055]). Error bars are SE.

Flatwoods

Survey of Six INAI Sites on the Illinoian Till Plain - Ground layer species diversity was inversely related to tree density (Fig. 2) on all plots from the six sites examined together and on four of the six sites examined separately. The two exceptional sites fit the null attrition model 2; these latter sites had the greatest mean soil AWC among flatwoods study sites. The overall pattern due to regression of tree density on ground-layer diversity (Fig. 2) fits the species attrition hypothesis (r = -0.57, F = 23.1, df = 48, $P < 0.0001$) and is made particularly strong by including Lake Sara Flatwoods (LSF), a site with an extensive fire history (20 years of nearly annual fire). Without the burned unit, the pattern is weaker but still significant (r = -0.37, F = 6.1, df = 38, P = 0.017). Mean comparisons of ground layer diversity (H') by tree density classes (low, medium, and high) were significantly different (Fig. 3), with the greatest diversity in the low tree density class. All pairwise comparisons also were different. (The difference between low and medium tree density was marginally significant [P = 0.055]).

Functional group richness also was inversely related to tree density (Fig. 4), but this pattern was true primarily at the community level. Only one individual site, Posen Woods, showed this pattern. Overall differences in functional group richness among tree-density classes were significant (Fig. 5). Functional groups proportionately more common in low- and medium-density portions of stands include nitrogen-fixing species (legumes), annual/biennial species, perennial forbs, perennial sedges, and perennial grasses. Woody functional groups (vines, shrubs, tree seedlings) were predominant in the high tree-density sites (Fig. 6).

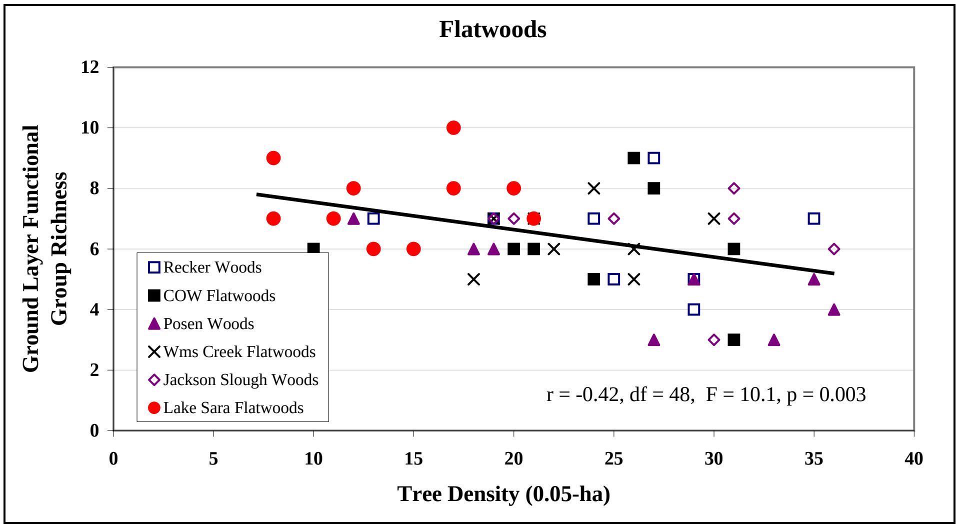

Figure 4.—Interaction between overstory tree density and ground layer functional group richness for six flatwoods recognized by the Illinois Natural Areas Inventory. The pattern fits the functional group attrition hypothesis. Functional group richness is the total count among sample quadrats for each tree plot (n = 12 for Lake Sara and n = 25 for all other sites). Tree density is based on stems ≥ 6 cm d.b.h.

Figure 5.—Comparison of average ground-layer functional group richness among tree density classes in flatwoods. Differences from mean comparisons are significant (ANOVA: $F = 5.8$, $df = 2$, $P < 0.006$). Different letters indicate significant differences in pairwise comparisons with Tukey post-hoc test. Error bars are SE.

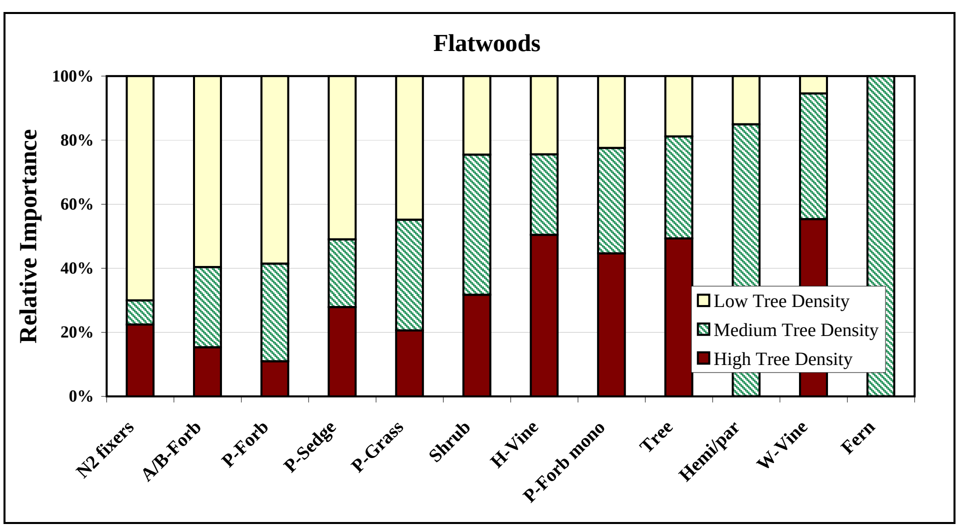

Figure 6.—Distribution of ground-layer functional groups proportionate to tree density classes in flatwoods. Tree density classes had significantly different means. N2 = nitrogen fixers, A/B = annual/biennial, H = herbaceous, P = perennial, Hemi/par. = Hemiparasites, W = woody.

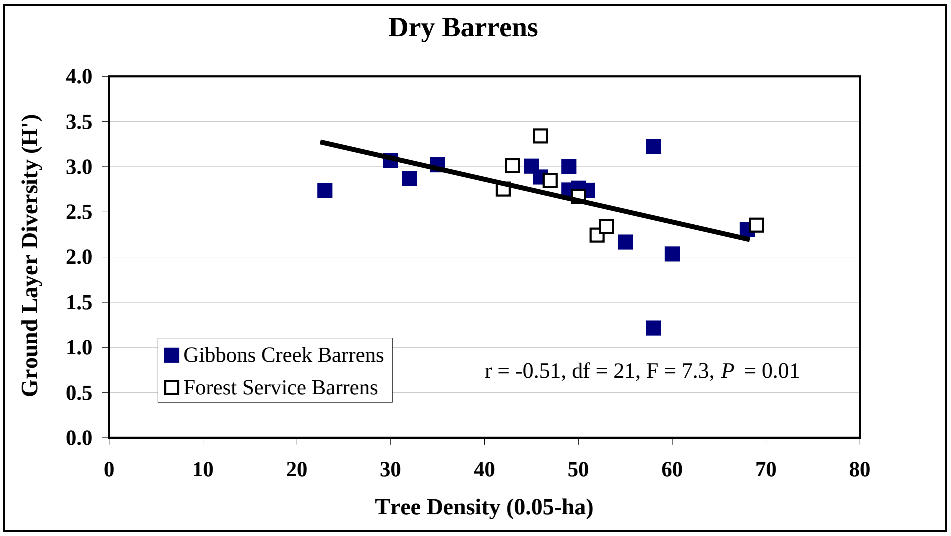

Figure 7.—Interaction between overstory tree density and ground layer diversity fitting the species attrition hypothesis from 23 plots sampled at Gibbons Creek and Forest Service barrens. Ground layer diversity is the average Shannon-Wiener Index (H') for each tree plot. Tree density includes stems ≥ 6 cm d.b.h.

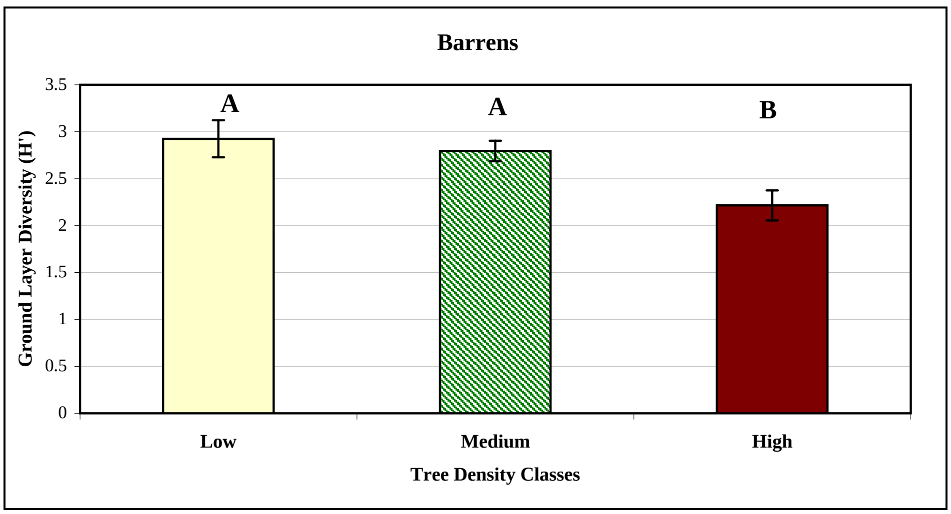

Figure 8.—Mean comparison of ground layer diversity (Shannon-Wiener Index [H']) among tree density classes in barrens sample data. The overall differences are significant (ANOVA: F = 5.4, df = 2, P = 0.01); different letters indicate significant pairwise comparisons with Tukey post-hoc test. Error bars are SE.

Mt. Vernon Flatwoods - Sample data from MVF are in contrast with most of the flatwoods dataset described above. At MVF there was no correlation between stand density and ground layer diversity. Fifteen of the 20 most frequent taxa in the ground layer were woody species. These conditions suggest null attrition model 2. A fire management study was conducted to determine restoration potential of this site (Taft 2005; see below).

Barrens

Gibbons Creek and Forest Service Barrens - Ground-layer species diversity trends lower with increasing tree density (Fig. 7). The pattern due to regression fits the species attrition hypothesis when baseline, preburn data from both sites are combined. Density of post oak alone, by far the dominant tree, explains even more of the variance in ground layer diversity (F = 25.8, $P < 0.001$). Differences in mean ground layer diversity among tree-density classes were significant with the greatest diversity in the low tree density class; however, there were no differences between low and medium tree-density classes (Fig. 8).

Functional group richness also trends lower with increased tree density (Fig. 9). Differences in functional group richness among tree-density classes were significant with the greatest richness in the low tree density class, though there were no differences between low and medium tree-density classes (Fig. 10). Functional groups proportionately more common in low density portions of stands are herbaceous vines, ferns, cool and warm season grasses, perennial forbs, and sedges (Fig. 11).

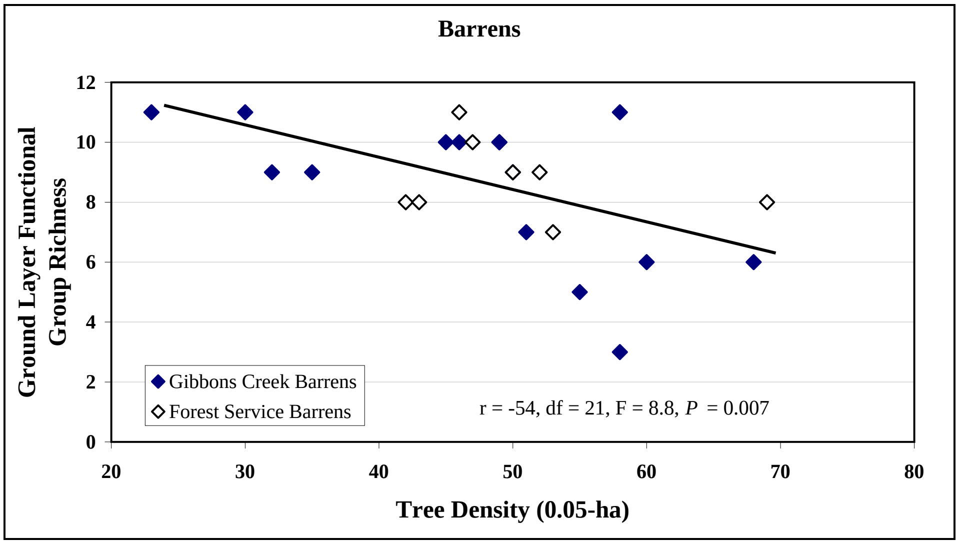

Figure 9.—Interaction between overstory tree density and ground-layer functional group richness in baseline (preburn) barrens sample data. The pattern fits the functional group attrition hypothesis. Functional group richness is the total count among sample quadrats for each tree plot (n = 12). Tree density is based on stems ≥ 6 cm d.b.h.

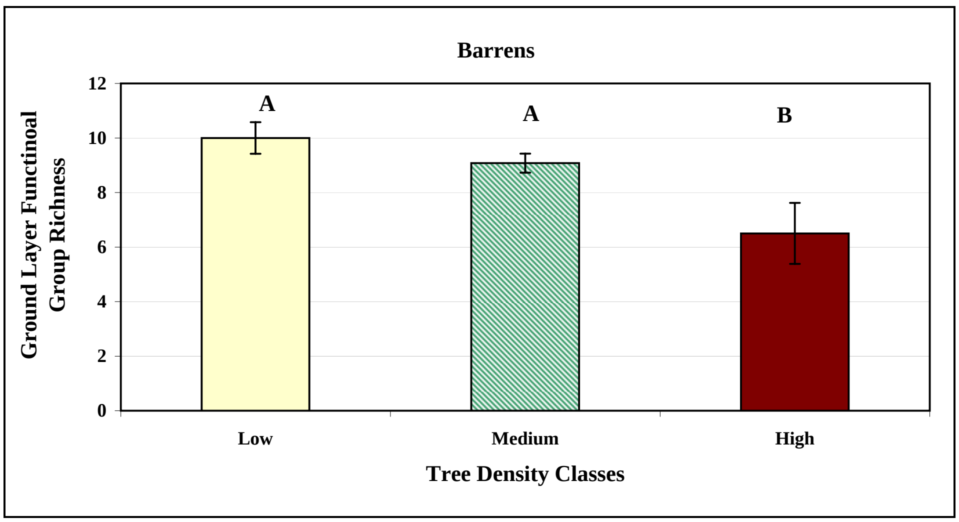

Figure 10.—Comparison of average ground-layer functional group richness among tree density classes in barrens. Differences from mean comparisons are significant (ANOVA: F = 6.2, df = 2, P = 0.008). Different letters indicate significant differences in pairwise comparisons with Tukey post-hoc test. Error bars are SE.

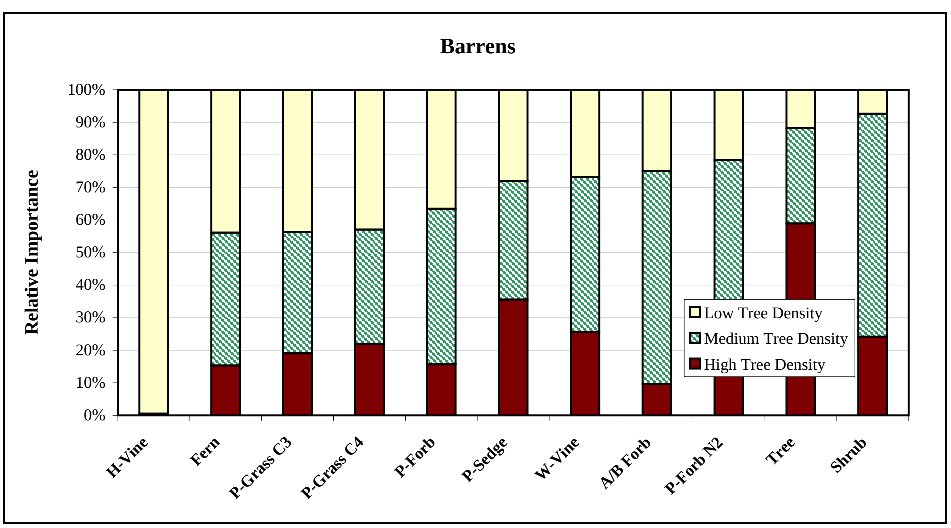

Figure 11.—Distribution of ground layer functional groups proportionate to tree density classes in barrens. Tree density classes had significantly different means. H = herbaceous, P = perennial, C3 (cool season), C4 (warm season), A/B = annual/biennial, W = woody, N2 = nitrogen fixers.

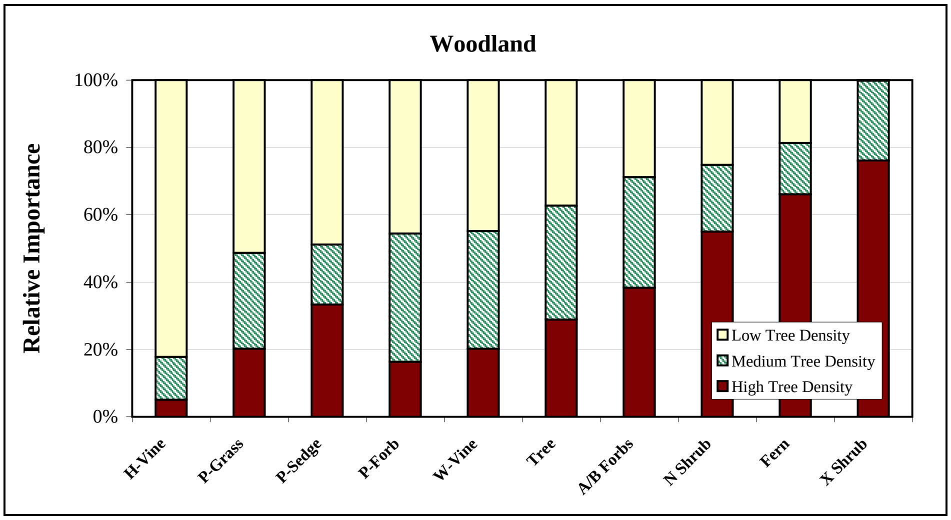

Figure 12.—Distribution of ground-layer functional groups proportionate to tree density classes in woodlands. Tree density classes had significantly different means. H = herbaceous, A/B = annual/biennial, P = perennial, W = woody, N = native, X = nonnative.

Figure 13.—Lake Sara Flatwoods (left) in Effingham County, IL, following a 20-year period of annual burns. Photo taken in July 1990. Note the absence of fire scars. Contrast with Mt. Vernon Flatwoods (right) in Jefferson County, IL, on the same soil series (Wynoose) but lacking recent fire. While tree densities (stems ≥ 6 cm d.b.h.) differ greatly (284 vs. 465 stems/ha), basal area is more similar (20.2 vs. 24.7 m^2/ha).

Dry-Mesic Woodland

Beaver Dam State Park - Among all sampled management units at BDSP, there is no correlation between tree density and ground-cover species density or diversity of ground-layer functional groups, suggesting null attrition model 2 with the extensive thicketization. Previously burned units show a weak pattern consistent with the species attrition hypothesis; however, the pattern due to regression is nonsignificant. Functional group richness also did not show a pattern of attrition with greater tree density. However, functional groups with affinity for low tree-density zones include herbaceous vines, perennial grasses, perennial sedges, perennial forbs, and woody vines (Fig. 12).

Fire Effects on Interactions Among Strata

Flatwoods - Lake Sara Flatwoods, a site with an extensive fire history, strengthened the species attrition pattern found among sample units with relatively high species diversity and low tree density. Ground-layer species density was over four times the mean species density for all other sites. LSF and MVF occur on the same soil series, Wynoose silt loam, but differ greatly in ground layer cover, composition, and diversity (Fig. 13). Following three burns at MVF, species density increased three-fold and tree density declined 26 percent; however, there was a 270-percent increase in density of stems in the shrub/sapling stratum (Taft 2005). Following three burns, there remained no discernable species attrition

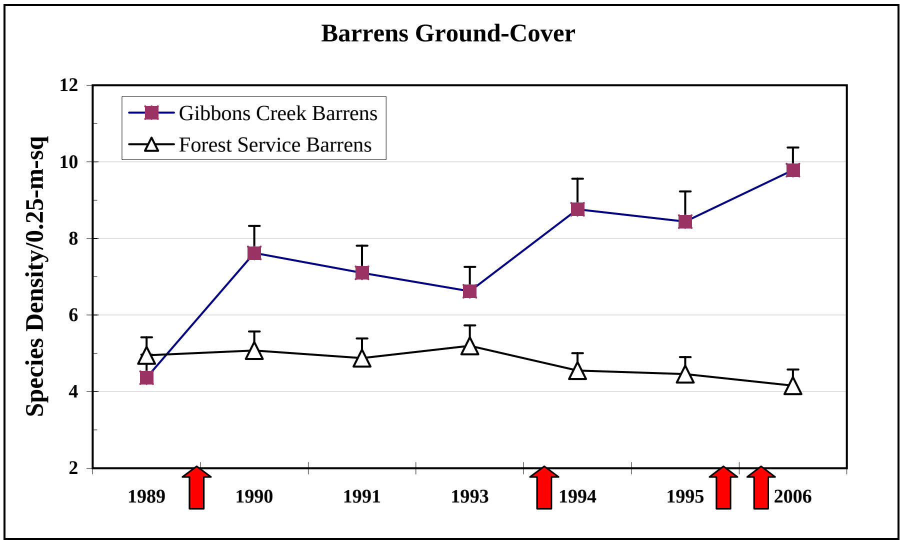

Figure 14.—Comparison of trends in ground-layer species density between a barrens remnant managed with prescribed fire (Gibbons Creek Barrens) and a fire-free reference unit (Forest Service Barrens) in Pope County, IL. Arrows indicate times of burns. Error bars are SE.

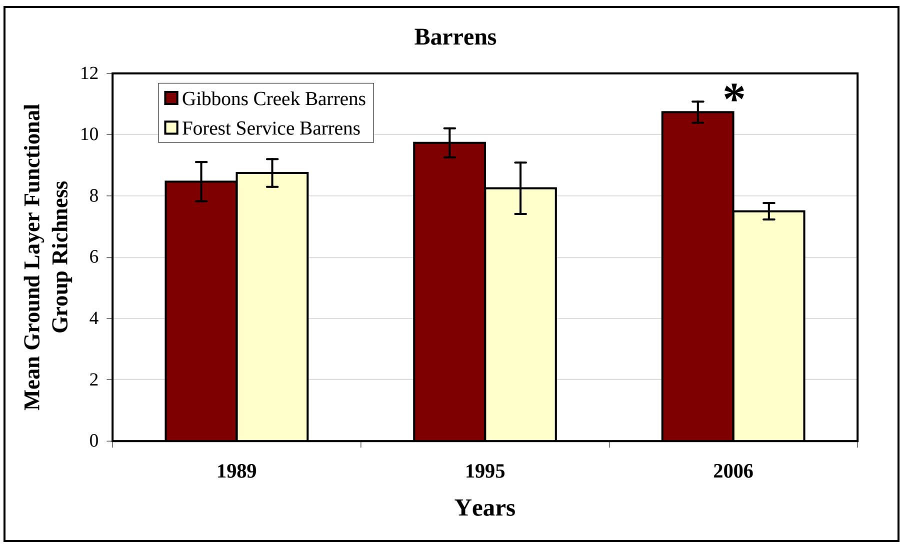

Figure 15.—Comparison of changes in mean functional group richness in fire treatment and fire-free reference units in dry barrens habitats. Two fire treatments occurred before 1995 and two after. * indicates significant difference (t-test: $t = 7.4$, $df = 21$, $P < 0.0001$). Error bars are SE.

pattern. The increase in ground layer diversity resulted primarily from species stored in the soil seed bank. Eight sedge species that were absent in baseline samples emerged following the burns, but perennial grasses dominant at other flatwoods remnants remained very sparse or absent.

Barrens - Following four burns over a 17-year period at GCB, ground layer species density more than doubled (Fig. 14), tree density declined 29 percent, shrub/sapling density declined 35 percent, and the pattern of ground-layer species attrition was strengthened compared to the baseline sample ($r = -0.70$ compared to $r = -0.51$), but only when the fire-free reference plots are included. Data from the fire treatment unit following four burns approximates null attrition model 1 with extensive percent cover of a rich herbaceous ground layer. The pattern of functional group attrition also was strengthened compared to the baseline sample data ($r = -0.68$ compared to $r = -0.54$), and made particularly strong when data from fire treatment and fire-free reference units are combined.

Not only do spatial patterns of tree density and ground layer diversity support the species attrition hypothesis, but long-term trends data from FSB, the fire-free reference site, also show gradual species losses over time (Fig. 14). Trends for mean functional group richness show a diverging pattern from nearly equal values in the baseline samples (1989) to widely and significantly divergent means in 2006 following 17+ years of fire absence at FSB and four burns at GCB (Fig. 15).

Predominantly declining functional groups at FSB include perennial grasses (C3 and C4 species combined for a 35-percent reduction in cover), sedges, woody vines, and perennial forbs. All functional groups declined in percent cover at FSB except for tree seedlings.

Dry-Mesic Woodland - Fire treatments at BDSP resulted in a significant decrease of tree density (5.6 percent) and a significant increase in ground-layer species density (17 percent). In fire-free reference units, there were a slight increase in tree density and a significant decline in ground-layer species richness compared to baseline samples 2 years earlier. As a result of these changes, an overall pattern consistent with the species attrition hypothesis developed; however, the pattern due to regression is relatively weak and non-significant (P = 0.14). Data from fire treatment units alone, which showed a weak but nonsignificant attrition pattern with the baseline data, show a significant inverse pattern in the relationship between tree density and ground-layer species density after an additional fire treatment (r = -0.57, F = 5.7, df = 12, P = 0.03).

DISCUSSION

A chief limiting factor for ground-layer species richness and cover in savanna habitats is light availability (Bray 1958, 1960, Leach and Givnish 1999, Peterson et al. 2007). However, available light is spatially and temporally heterogeneous and accurate assessment requires specialized equipment and considerations of several abiotic and biotic variables measured over time (Neufeld and Young 2003). In this study, tree density was tested as a factor for rapid and cost-effective assessment of the influence of overstory stand structure on ground-layer species diversity in savanna and woodland habitats. Tests were conducted at different ecological scales: across multiple habitats, within habitats, and within individual sites and management units. Patterns of overstory-to-ground layer interactions were obscured in this study when multiple habitat types, including sites with widely ranging ecological conditions, were examined collectively. Based on these datasets, the most appropriate ecological scale for examining patterns of overstory and ground layer interactions relevant to the species and functional group attrition hypotheses is within habitats and sites.

Results summarized here explore overstory-ground layer interactions as an indicator of ecological condition and restoration potential. The post-fire species attrition hypothesis and functional group attrition hypothesis predict that with increasing thicketization (overstory density) during extended periods of fire absence, ground-layer species diversity and the richness of plant functional groups will decline until a stasis is reached and remaining species are relatively indifferent to shade levels (null attrition model 2). Alternatively, with increasing fire, reduced tree density, and increased ground layer diversity (e.g., Taft 2003, Bowles et al. 2007, Peterson et al. 2007), the relations between tree density and ground-layer diversity may strengthen. Eventually, a temporary stasis might be reached where shade levels are not limiting to ground layer vegetation (null attrition model 1).

These results suggest a sequence of three developmental stages with fire absence in savanna and open woodland habitats relevant to ecological condition and restoration potential: 1) null attrition model 1, where tree density has yet to alter ground layer patterns of diversity; 2) post-fire species attrition where patterns of tree density are inversely related to ground layer diversity; and 3) null attrition model 2, where tree density (thicketization) has progressed to the stage where primarily only shade-tolerant ground layer species remain (e.g, MVF, unburned portions of BDSP). While there is potential for restoration of some aspects of habitats resembling null attrition model 2, full restoration may not be possible without species augmentation at some sites due to seed bank attrition (Bond 1984). In contrast, sites fitting null model 1 and the post-fire species attrition model merit consideration as management priority sites.

The pace of thicketization with fire absence can be attributed to edaphic resource availability (Taft et al. 1995, Peterson et al. 2007). Among the flatwoods examined here, sites fitting null attrition model 2 had the greatest AWC. Likewise, savanna habitats on deep, rich soils are particularly imperiled because they have fewer resource limitations (e.g., available moisture) compared to dry-to-xeric sites and readily convert to forests with extended fire absence (Nuzzo 1986, Anderson and Bowles 1999).

Applications of prescribed fire in oak savannas and woodlands in the prairie-forest transition zone, particularly repeated burns, consistently result in reduced tree density and increased ground layer species richness and diversity (White 1983, Taft 2003, Bowles et al. 2007). Consequently, development of the species attrition pattern where formerly absent (i.e., null model 2) would indicate successful treatment. The loss of the attrition pattern with fire management where it once occurred potentially would suggest the development of null attrition model 1.

In addition to the spatial pattern of decline among ground layer species with stand thicketization, long-term trends data at a fire-free reference barrens community (FSB) suggest that species loss can be a gradual and ongoing process. Perennial grasses and sedges showed the greatest decline with long-term fire absence at FSB. Similar patterns during long fire-free intervals can be expected in savanna and woodland habitats throughout the prairie-forest transition zone. Functional groups most affiliated with lower tree density in flatwoods, barrens, and woodland habitats in descending average rank order are: herbaceous vines, cool-season perennial grasses, perennial forbs, warm-season perennial grasses, perennial sedges, and nitrogen-fixing species. (In barrens, these species are most strongly affiliated with intermediate tree-density zones.) Herbaceous vines (e.g., crested bindweed [*Polygonum cristatum*], wild yam [*Dioscorea villosa*], small passion flower [*Passiflora lutea*], and carrion flower [*Smilax lasioneuron*]) were relatively scarce in all datasets (absent in Appendix 1, pg. 38).

Based on data summarized here, ordered disassembly (Zavaleta and Hulvey 2004) of savanna/woodland ground layer with thicketization begins with declines among perennial grasses, sedges, perennial forbs including herbaceous vines, and nitrogen-fixing species. A degree of spatial heterogeneity in stand structure contributes to overall site diversity because plant functional groups are partitioned into differentially shaded zones, for example with warm-season grasses in the most open sites and forbs most abundant in semi-shaded zones (Leach and Givnish 1999, Peterson et al. 2007). Similar results are found in the present study with lower diversity most pronounced in the high tree-density zones. Such trends can help guide interpretation of site conditions and the degree of needed restoration for recovery of ground layer diversity.

The soil seed bank is an important refuge for ground layer diversity (Taft 2003), particularly where there is extensive habitat fragmentation. However, the abundance of viable seeds likely declines over time. Bond (1984) and Bond and van Wilgen (1996) described a post-fire seed attrition hypothesis that predicts seed availability following fire would decline in the seed bank depending on the time frame for action by seed predators, decay, and disease. The restoration of sedges at MVF but not grasses or legumes (Taft 2005) and the recovery of C3 grasses but not C4 grasses at GCB (Taft 2003) suggest some species and functional groups may not persist in the seed bank for extended periods.

The question has been raised as to whether it is too late to restore oak communities in the eastern United States using prescribed fire except on the driest sites (Abrams 2005). While there remain opportunities for restoring woodland and savanna-like habitats in the prairie-forest transition zone, particularly remnants on dry to xeric sites, spatial and long-term trends in these communities suggest at least structural and in some cases compositional instability in the overstory. In addition, ground layer diversity continually declines during long periods of fire absence. In a highly fragmented landscape such as the Midwest, where species immigration opportunities are limited, the time frame for restoring the full compliment of species and plant functional groups in ground layer vegetation of oak woodland and savanna-like habitats appears to be limited.

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APPENDIX 1

Dominant species examined for this study from flatwoods, barrens, and woodland habitats in Illinois. The dominant species are the top ranking 10, 15, and 50 species from canopy, shrub/sapling, and ground cover strata, respectively, from combined lists for all three habitats. Summary values for each stratum and habitat are shown. IV 200 is sum of two relative values (e.g., basal area and density for trees). FG = Functional Group, af = annual forb, pf = perennial forb, pg C3 = perennial grass (cool season), pg C4 = perennial grass (warm season), ps = perennial sedge. * = nonnative species.

	FLATWOODS	BARRENS	WOODLAND	
CANOPY STRATUM				
- Top 10 ranking species	IV 200	IV 200	IV 200	
Quercus stellata	114.8	109.9	13.5	
Ulmus rubra	3.0	-	41.3	
Ulmus alata	-	35.5	-	
Quercus velutina	14.9	0.3	20.0	
Carya ovata	10.9	4.4	16.5	
Quercus alba	5.9	0.6	23.2	
Quercus marilandica	17.3	6.9	-	
Celtis occidentalis	-	-	24.2	
Carya texana	2.5	15.1	-	
Carya glabra	0.6	14.6	-	
% of IV	**85.0**	**93.7**	**69.4**	
Summary Data				
Tree Density/ha	464.0	975.0	796.0	
Basal Area (m^2/ha)	23.1	18.1	25.8	
SHRUB/SAPLING STRATUM				
-Top 15 ranking species	IV 200	IV 200	IV 200	
Ulmus alata	-	44.7	-	
Ulmus rubra	7.0	-	30.4	
Rubus alleghiensis/pen.	29.6	-	0.5	
Sassafras albidum	20.4	-	8.1	
Quercus stellata	7.8	17.8	-	
Carya ovata	13.3	7.2	3.7	
Prunus serotina	15.5	4.6	3.8	
Fraxinus americana	2.9	10.3	9.9	
*Rosa multiflora**	2.0	-	20.4	
Quercus velutina	11.1	10.6	0.3	
Celtis occidentalis	2.5	-	19.5	
Carya texana	7.1	13.6	-	
Symphoricarpos orbiculatus	-	13.7	7.0	
*Lonicera maackii**	-	-	16.9	
Quercus marilandica	9.5	5.2	-	
% of IV	**64.3**	**63.8**	**60.2**	
Summary Data				
Shrub/Sapling Density/ha	4,004	5,323	14,196	
GROUND LAYER SPECIES				
- Top 50 species, combined habitats	IV 200	IV 200	IV 200	FG
Acalypha gracilens	2.19	3.52	-	af
Impatiens capensis	-	-	4.13	af
Pilea pumila	-	-	2.52	af

continued

Appendix 1 continued

	FLATWOODS	BARRENS	WOODLAND	
Asplenium platyneuron	-	3.21	-	fern
Helianthus divaricatus	18.39	15.12	-	pf
Sanicula odorata	-	-	22.98	pf
Eupatorium rugosum	-	-	8.49	pf
Circaea lutetiana	-	-	6.59	pf
Helianthus strumosus	-	-	5.17	pf
Polygonum virginicum	-	-	3.63	pf
Galium pilosum	-	3.50	-	pf
Antennaria plantaginifolia	-	3.41	-	pf
Geum canadense	-	-	3.37	pf
Parthenium integrifolium	3.10	-	-	pf
Viola sororia	-	-	3.08	pf
Phryma leptostachya	-	-	2.68	pf
Euphorbia corollata	-	2.40	-	pf
Solidago ulmifolia	-	-	2.25	pf
Smilacina racemosa	-	-	2.14	pf
Danthonia spicata	1.51	34.01	-	pg C3
Dichanthelium laxiflorum	-	16.39	-	pg C3
Cinna arundinacea	9.83	-	-	pg C3
Agrostis scabra	7.86	-	-	pg C3
Dichanthelium acuminatum	7.84	-	-	pg C3
Dichanthelium boscii	-	4.74	-	pg C3
Dichanthelium linearifolium	-	3.57	-	pg C3
Festuca obtusa	-	-	2.66	pg C3
Bromus pubescens	-	-	2.06	pg C3
Schizachyrium scoparium	-	21.19	-	pg C4
Sorghastrum nutans	-	3.26	-	pg C4
Andropogon gerardii	-	2.65	-	pg C4
Carex artitecta	-	12.71	-	ps
Carex pensylvanica	9.69	-	-	ps
Eleocharis verrucosa	6.76	-	-	ps
Carex festucacea	6.26	-	-	ps
Carex hirsutella	4.99	-	-	ps
Carex blanda	-	-	2.36	ps
Carex radiata	-	-	1.92	ps
Rubus allegheniensis	13.59	-	-	shrub
Rubus flagellaris	7.53	-	-	shrub
Rosa carolina	-	2.63	-	shrub
Ulmus rubra	-	-	11.10	tree
Quercus stellata	3.65	1.82	-	tree
Sassafras albidum	4.45	-	-	tree
Celtis occidentalis	-	-	3.33	tree
Cercis canadensis	-	-	1.99	tree
Parthenocissus quinquefolia	30.60	12.57	24.54	w vine
Toxicodendron radicans	5.04	1.55	2.17	w vine
Vitis riparia	-	-	2.86	w vine
Smilax hispida	-	-	2.14	w vine
% of IV	**71.64**	**74.13**	**61.12**	
Summary Data				
Species Density (1/4-m^2 quadrat)	2.79	4.65	6.51	
Species Diversity (H')	2.33	2.67	2.02	

FIRE AND FIRE SURROGATE STUDY: ANNOTATED HIGHLIGHTS FROM OAK-DOMINATED SITES

Daniel A. Yaussy[1] and Thomas A. Waldrop[2]

[1]Supervisory Research Forester, U.S. Forest Service, Northern Research Station, Delaware, OH
[2]Team Leader, U.S. Forest Service, Southern Research Station, Clemson, SC
DAY is corresponding author; to contact, call (740) 368-0101 or email at dyaussy@fs.fed.us

Abstract.—The National Fire and Fire Surrogate (FFS) study was implemented to investigate the ecological impacts of prescribed fire and mechanical operations to mimic fire in restoring the structure and function of forests typically maintained by frequent, low-intensity fires. Two of the 12 sites were located in oak-dominated forests, one in Ohio and another in North Carolina. This paper summarizes results from these two sites that have been published in peer-reviewed literature, covering fire history, fuels and fire behavior, entomology, soils and belowground processes, wildlife, and vegetation. We concluded that the FFS treatments did little harm to this ecosystem, benefit many ecosystem components, and promote oak and hickory regeneration. These effects could be transient, however, and need to be studied over the long term to determine sustainability of the ecosystem.

INTRODUCTION

The National Fire and Fire Surrogate (FFS) Study was established in 2000 to compare ecological and economic impacts of prescribed fire and mechanical fuel-reduction treatments in forest types that developed under low intensity, high frequency fire regimes (Youngblood et al. 2005). These treatments were chosen to restore ecosystem structure and resiliency to forests that have experienced fire suppression for several decades. Twelve independent study sites across the United States (seven in the West and five in the East; Fig. 1) received identical treatment (prescribed fire and mechanical fuel reduction treatments) and measurement protocols. Core variables collected at each of the 12 sites encompassed several ecosystem components: vegetation composition and structure, fuel loading and fire behavior, soils and forest floor physical and chemical properties, wildlife, entomology, pathology, and utilization and economics. Five western sites are dominated by ponderosa pine (*Pinus ponderosa*) and two are classified as mixed coniferous forests. Eastern sites consist of hardwood-dominated forests in the Central Appalachian Plateau in Ohio and Southern Appalachian Mountains of North Carolina, a pine-hardwood forest in the Piedmont of South Carolina, a forest dominated by longleaf pine (*P. palustris*) in Alabama, and a site dominated by slash pine (*P. elliottii*) in Florida.

Two FFS sites—the Southern Appalachian Mountain and Central Appalachian Plateau sites—are located in oak-dominated forests. Scientific information generated by these two sites has been disseminated to more than 10,000 people during field trips. More than 50 scientific articles have been published in peer-reviewed journals or U.S. Forest Service publications. Downloadable versions of many of these publications are available through the www.treesearch.fs.fed.us Website. A listing of most FFS study articles may be found at the frames.nbii.gov website (click on the FFS link, then publications). In this paper we present an annotated reference list of some of the research highlights from these two oak-dominated sites to provide the readers with a sense of the breadth of the work conducted. Land managers may wish to use this publication as a starting point to find articles concerning the effects of fire and mechanical treatments on oak-dominated ecosystem components. In some cases, results reported in earlier articles (such as Rebbeck et al. 2004, Albrecht and McCarthy 2006, Long et al. 2006, and Phillips et al. 2007) have been included or superseded by articles annotated here. Specifically, we will show that mechanical and prescribed fire treatments can restore a more open structure to oak-dominated forests of the Appalachians without causing harm to the ecosystem.

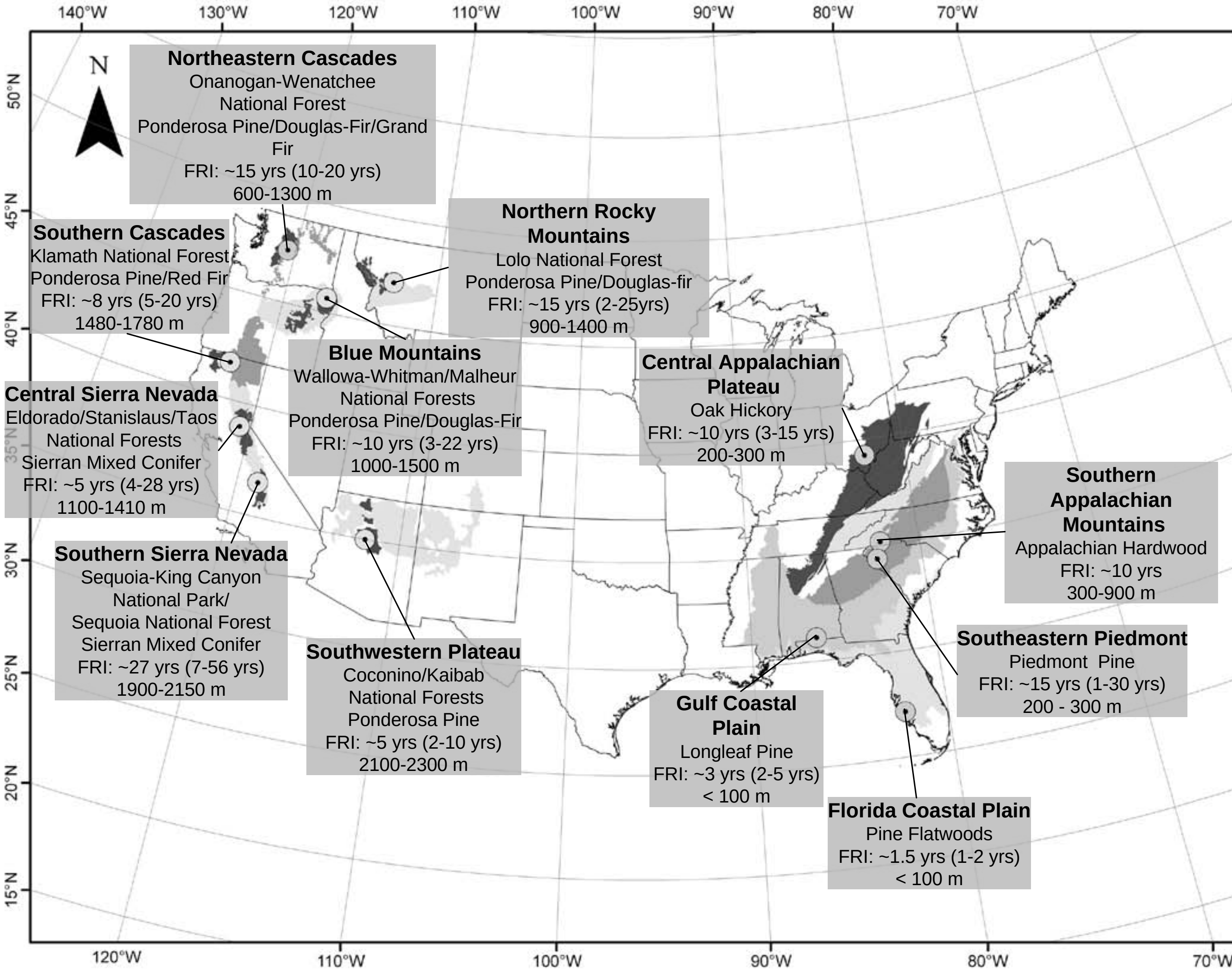

Figure 1.—Name and location of 12 Fire and Fire Surrogate (FFS) sites, showing nearest federal lands (western sites only), fire return interval (FRI), and elevational range (meters). Black shaded areas are federal lands adjacent to the western sites; lighter shading indicates 'representative land base,' or the area to which FFS results can be most directly applied for each site. Representative land bases are derived from EPA Type III Ecoregions: www.epa.gov/wed/pages/ecoregions/level_iii.htm. This paper deals specifically with the Central Appalachian Plateau and Southern Appalachian Mountain sites located in Ohio and North Carolina.

SITE LOCATIONS

The Central Appalachian and Southern Appalachian study sites each consist of three replicate blocks, with four fuel reduction treatments applied to a randomly chosen treatment unit within each block. The Central Appalachian FFS site is located on the unglaciated Allegheny Plateau of southern Ohio. The climate of the region is cool temperate with a mean annual precipitation of 40.3 in and a mean annual temperature of 52.3 °F (Sutherland et al. 2003). The forests of the region developed between 1850 and 1900, after the cessation of cutting for the charcoal and iron industries (Sutherland et al. 2003). The current canopy composition differs little from that recorded in the original land surveys of the early 1800s. The most abundant species in the current canopy were white oak (*Quercus alba*), chestnut oak (*Q. prinus*), hickories (*Carya* spp.), and black oak (*Q. velutina*); however, the midstory and understory are now dominated by species that have become common in this community only in the last few decades (e.g., sugar maple [*Acer saccharum*], red maple [*A. rubrum*], and yellow-poplar [*Liriodendron tulipifera*]) (Yaussy et al. 2003). The Central Appalachian FFS site is composed of three experimental blocks, with one each in the

Raccoon Ecological Management Area (REMA), Zaleski State Forest (ZAL), and Tar Hollow State Forest (TAR). Funding for the initial implementation on this site of the FFS study was through a grant from the USDA-USDI Joint Fire Sciences Program.

The Southern Appalachian FFS site is located in the Green River Game Land in the Blue Ridge Physiographic Province, Polk County, NC. The climate of the region is warm continental, with a mean annual precipitation of 64.5 in and a mean annual temperature of 63.7 °F (Keenan 1998). The forests of the study area were 80 to 120 years old, and no indication of past agriculture or recent fire was present, though the historical fire-return interval prior to 1940 was approximately 10 years (Harmon 1982). The most abundant species in the canopy were northern red oak (*Q. rubra*), chestnut oak, white oak, black oak, pignut hickory (*C. glabra*), mockernut hickory (*C. tomentosa*), and shortleaf pine (*P. echinata*). A relatively dense evergreen shrub assemblage was present in the understory of most of the study site, with mountain laurel (*Kalmia latifolia*) and rhododendron (*Rhododendron maximum*) the most common species. A grant from the National Fire Plan funds this site of the FFS study.

TREATMENTS AND EXPERIMENTAL DESIGN

Each of the three replicate blocks in each site is composed of four treatment units. At the Central Appalachian site, individual treatment units were 47 to 64 ac whereas in the Southern Appalachian site they were approximately 35 ac in size. A 164-ft x 164-ft grid was established in each treatment unit, and 10 sample plots of 0.25 ac were established randomly within each treatment unit.

Treatments were randomly allocated among treatment units at each site. Treatments consisted of prescribed fire (B), a mechanical treatment (M), the combination of prescribed fire and mechanical treatments (MB), and an untreated control (C). In the Central Appalachian site, the M treatment involved a shelterwood harvest. This commercial operation reduced basal area from 125 to 88 ft^2/ac. At the Southern Appalachian site, the M treatment was designed to create a vertical fuel break. Chainsaw crews removed all stems >6.0 ft tall and < 4.0 in d.b.h., as well as all mountain laurel and rhododendron stems, regardless of size. All detritus generated by the mechanical treatments was left on site in both areas.

M treatments were accomplished between September 2000 and April 2001 in Ohio and between December 2001 and February 2002 at the Southern Appalachian site. The prescribed fires were applied during March-April 2001 and 2005 in the Central Appalachian and March 2003 and 2006 at the Southern Appalachian site. The fires were applied on both the M and MB experimental units. These dormant-season fires consumed unconsolidated leaf litter and fine woody fuels while leaving most of the coarse woody fuels only charred. At the Southern Appalachian site, the fire prescription was also designed to kill ericaceous shrubs. Details of fire behavior are given by Iverson and others (2004a, b; 2008) for the Central Appalachian and Tomcho (2004) and Waldrop and others (2008) for the fires at the Southern Appalachian site.

SEQUENTIAL ANNOTATIONS FOR SELECTED REFERENCES

Fuels and Fire Behavior

Iverson, L.R.; Yaussy, D.A.; Rebbeck, J.; Hutchinson, T.F.; Long, R.P.; Prasad, A.M. 2004a. **A comparison of thermocouples and temperature paints to monitor spatial and temporal characteristics of landscape-scale prescribed fires. International Journal of Wildland Fire.** 13: 1-12.

> A method to monitor fire behavior using rigid steel thermocouple probes is described and compared to the use of temperature-sensitive paints at the Ohio site. Different paints melt to indicate the maximum temperature recorded. The thermocouple probes can be programmed to record temperature at set intervals, such as every 2 seconds, to indicate not only maximum temperature, but the duration of increased temperature. Readings from the two methods corresponded closely with each other. Deployed on a 55-yd x 55-yd grid, these sensors each give readings that may be used to calculate fire

behavior metrics such as rate of spread and Byram's fireline intensity. An illustration of prescribed fire movement through the ZAL burn treatments is included.

Graham, J.B.; McCarthy, B.C. 2006. **Forest floor fuel dynamics in mixed-oak forests of south-eastern Ohio.** International Journal of Wildland Fire. 15: 479-488.

The authors examined fuel dynamics following the fire and thinning treatments at the Central Appalachian site to determine how these treatments influence the future fuel composition and structure in eastern mixed-oak forests. They found that the M treatment to create shelterwoods, increased 100-hr, 1000-hr sound, and coarse woody debris, and decreased 1-hr fuels 3 years after treatment, whether the units were burned or not. The M treatment without burning increased litter during this time. The B treatment increased 1000-hr sound fuels and decreased 1000-hr rotten fuels. The MB treatments decreased 10-hr fuels and the duff layer. Unlike western FFS study sites where fuels accumulate over time, fuel inputs were balanced by rapid decomposition at this site.

Phillips, R.J.; Waldrop, T.A.; Simon, D.M. 2006. **Assessment of the FARSITE model for predicting fire behavior in the Southern Appalachian Mountains.** Proceedings of the 13th biennial Southern Silvicultural Research Conference. Gen. Tech. Rep. SRS-92. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 521-525.

The authors wished to evaluate FARSITE by comparing fire behavior from simulations to that from a prescribed burn and to test the effects of different fuel treatments on fire behavior in the Southern Appalachian Mountains. Fire behavior was monitored using thermocouple probes deployed on a 55-yd x 55-yd grid. Fire behavior simulations produced by FARSITE were not adequate for use in the Southern Appalachian region using standard fire behavior fuel models (FBFM) with or without calibrations. The FBFM used in this version of FARSITE did not account for the high fuel moistures of the heavier fuels. The irregular fire shape caused by a linear drip torch fire front (as opposed to point-source ignition) resulted in higher rates of spread than those predicted by FARSITE. The flammability of the ericaceous shrubs was not adequately modeled in the FBFM. FARSITE 4.1.0 (which was not available at the time of this evaluation) incorporates many improvements that may make the software more robust for this region.

Hutchinson, T.F.; Long, R.P.; Ford, R.D.; Sutherland, E.K. 2008. **Fire history and the establishment of oaks and maples in second-growth forests.** Canadian Journal of Forest Research. 38: 1184-1198.

To better understand how past fires were related to tree establishment, dendrochronological analysis was conducted on trees which were harvested to create the shelterwood stand structure on the Central Appalachian site. The authors found mean fire intervals (average number of years between fires) varied between 9.1 and 11.3 yrs on the six mechanical treatment units between 1870 and 1933. The oaks that were harvested originated between 1845 and 1900. Virtually no oaks established after 1925, close to the start of the fire suppression policies in Ohio. Harvested maples established after fire suppression. This finding supports the hypothesis that frequent, low-intensity surface fire was the ecosystem process that created the oak-dominated forests of today.

Soils and Belowground Processes

Iverson, L.R.; Prasad, A.M.; Hutchinson, T.F.; Rebbeck, J.; Yaussy, D.A. 2004b. **Fire and thinning in an Ohio oak forest: grid-based analyses of fire behavior, environmental conditions, and tree regeneration across a topographic moisture gradient.** In: Spetich, M.A., ed. Upland oak ecology symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS-73. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 190-197.

The authors present a preliminary analysis of how fire behavior, seasonal soil temperature and moisture, canopy light penetration, and oak and hickory seedling and sapling populations vary among moisture classes and before and after the first season of treatments. Thermocouple probes were deployed on a 55-yd x 55-yd grid 1 in below the soil within the C and MB treatments located on the ZAL State Forest. These probes indicated higher soil temperatures on the MB treatments at all slopes and aspects than on the C treatment. This trend was evident from April through September. The vegetation analysis presented in this paper is included and updated in Iverson et al. 2008 (see below), and will not be commented on here.

Boerner, R.E.J.; Brinkman, J.A.; Yaussy, D.A. 2007. **Ecosystem restoration treatments affect soil physical and chemical properties in Appalachian mixed oak forests.** In: Buckley, D.S.; Clatterbuck, W.K., eds. Proceedings, 15th central hardwood forest conference. e-Gen. Tech. Rep. SRS-101. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 107-115 [CD-ROM].

This paper assesses the effects of the treatments at the Central Appalachian site on soil physical and chemical properties measured immediately post-treatment and 3 years post-treatment (prior to the second prescribed fire) and evaluates the impact that any effects might have on decisions to implement one or more of these management approaches more broadly. Mineral soil exposure was still significantly higher in the treated areas than in the C at the time of the second measurement. Soil pH increased after burning and persisted at higher levels for at least 3 years. Phosphorus availability decreased and remained lower in the B and MB areas through the second measurement. The Ca:Al ratio increased immediately after burning, but the effect was greatly reduced by the second measurement. There was no significant effect on Ca, K, or Al availability or soil compaction.

Boerner, R.E.J.; Coates, T.A.; Yaussy, D.A.; Waldrop, T.A. 2008. **Assessing ecosystem restoration alternatives in eastern deciduous forests: the view from belowground.** Restoration Ecology. 16:3, 425-434.

The article investigates whether any or all of the treatments would move these ecosystems toward the N-limited ecosystems that preceded fire suppression and heavy atmospheric N deposition. The authors found that treatment effects differed somewhat between the Southern and Central Appalachian sites. At the North Carolina site, all manipulative treatments initially resulted in reduced soil organic carbon content, C:N ratio, and overall microbial activity; however, only the reduced microbial activity persisted into the fourth growing season. At the Ohio site, the M treatment was the only manipulation that resulted in a significant change in these properties, with an initial increase in organic carbon and a decreased C:N ratio. Four years post-treatment, all treatments at this site were similar to the C. Using evidence from soil properties, the researchers concluded that mechanical treatments and a single prescribed fire did not result in progress toward ecosystems present prior to fire suppression.

Coates, T.A.; Boerner, R.E.J.; Waldrop, T.A.; Yaussy, D.A. 2008. **Soil nitrogen transformations under alternative management strategies in Appalachian forests.** Soil Science Society of America Journal. 72(2): 558-565.

The authors wished to gain insight into the consequences of the management strategies represented by the FFS treatments to nitrogen availability. Treatment effects on available total inorganic nitrogen, nitrogen mineralization, and nitrification were measured at both the Central and Southern Appalachian sites, once pre- and twice post-treatment (prior to the second prescribed fire at each site). Changes in nitrogen transformations were modest and transient. Transformation rates were 2 to 10 times higher in Ohio than in North Carolina, which may be due to the inhibitory effects of the litter produced by the ericaceous shrubs present at the Southern Appalachian site.

Insects

Campbell, J.W.; Hanula, J.L.; Waldrop, T.A. 2007. **Effects of prescribed fire and fire surrogates on floral visiting insects of the Blue Ridge province in North Carolina.** Biological Conservation. 134: 393-404.

This study was conducted to determine how various groups of pollinating insects vary in abundance and species richness in response to treatments at the Southern Appalachian site. The number of pollinators captured during two summers was highly correlated to change in basal area. Reduction of basal area by any of the treatments increased light to the forest floor, stimulating herbaceous plant growth and flowering. Insect pollinators, mostly bees, were captured in significantly higher numbers in MB sites than in other treatment areas.

Campbell, J.W.; Hanula, J.L.; Waldrop, T.A. 2007. **Observations of *Speyeria diana* (Diana fritillary) utilizing forested areas in North Carolina that have been mechanically thinned and burned.** Southeastern Naturalist. 6(1): 179-182.

The authors suggest that forest management practices that include fire with other disturbances may enhance food resources for the Diana fritillary. The Diana fritillary is considered a species of concern by the U.S. Fish and Wildlife Service. This lepidoptera is thought to be indicative of undisturbed communities. During the data collection for the pollinator study above, several males of this species were observed in two consecutive years on flowering sourwood trees (*Oxydendrum arboretum*) in the MB treatment areas only. Further study of the habitat preferences of this butterfly is needed.

Lombardo, J.A.; McCarthy, B.C. 2008. **Forest management and curculionid weevil diversity in mixed oak forests of southeastern Ohio.** Natural Areas Journal. 28(4): 363-369.

The purpose of this study was to determine whether stand level treatments affect the diversity and abundance of adult weevils at the Central Appalachian site. Weevils were collected from acorns following the second prescribed fire conducted 4 years after the first treatment. The authors found no discernable treatment effects on the number of weevils, but the manipulative treatments increased weevil species diversity over the C treatment.

Wildlife

Greenberg, C.H.; Tomcho, A.L.; Lanham, J.D.; Waldrop, T.A.; Tomcho, J.; Phillips, R.J.; Simon, D. 2007. **Short-term effects of fire and other fuel reduction treatments on breeding birds in a Southern Appalachian upland hardwood forest.** The Journal of Wildlife Management. 71(6): 1906-1916.

The FFS study afforded the authors the opportunity to examine whether and how the treatments would affect bird communities or individual species. Breeding bird surveys were conducted annually for 5 years on the Southern Appalachian site starting 1 year pretreatment. Four years after treatment, species richness and density increased on the MB treatment, while only species richness increased on the B treatment. The decline in the density of three species was associated with decreased leaf litter and shrubs. Shrub cover was reduced on all manipulative treatments compared to the C treatment. Leaf litter depth was similar on the C and B, lowest on the MB, and highest on the M treatments.

Greenberg, C.H.; Miller, S.; Waldrop, T.A. 2007. **Short-term response of shrews to prescribed fire and mechanical fuel reduction in a Southern Appalachian upland hardwood forest.** Forest Ecology and Management. 243: 231-236.

This study examined the FFS treatment effects on communities of shrews, an important component of the food web in southern Appalachian oak ecosystems. Shrew populations were followed the first and second years after prescribed fire at the Southern Appalachian site. During this period, the abundance of shrews was not significantly different in the treatments than in the C; however, the MB treatment had significantly fewer shrews than the M treatment. Shrew abundance decreased significantly between the first collection and the second, which was not associated with any treatment effects. There was no difference in macroarthropods between treatments. Leaf litter depth differed, with the highest depths occurring on the M treatment and the thinnest leaf litter on the MB replications. These results imply that the difference in shrew abundance was due not to food source but to habitat structure.

Greenberg, C.H.; Otis, D.L.; Waldrop, T.A. 2006. **Response of white-footed mice (*Peromyscus leucopus*) to fire and fire surrogate fuel reduction treatments in a southern Appalachian hardwood forest.** Forest Ecology and Management. 234: 355-362.

Mice are known to be important herbivores, predators, and prey in oak ecosystems. The authors examined the effects of the FFS treatments on habitat structure and white-footed mouse populations. Seventy-nine percent of the rodents caught at the North Carolina site were white-footed mice. The number captured increased each year from pre-treatment to 2 years post-treatment on all treatments. The abundance of mice was no different on the M treatment than on the C for any year. The abundance increased on the B and MB treatments the first year after fire, but was significant only on the MB due to the high variability of collections.

Greenberg, C.H.; Waldrop, T.A. 2008. **Short-term response of reptiles and amphibians to prescribed fire and mechanical fuel reduction in a southern Appalachian upland hardwood forest.** Forest Ecology and Management. 255: 2883-2893.

Reptile populations appear to be more affected by practices which open the canopy than by prescribed fire. This study compares the effects of these two treatments, plus the combination of the two on reptile and amphibian populations. A total of 1,308 amphibians (13 species) and 335 reptiles (10 species) were captured over 4 years starting 1 year pretreatment at the Southern Appalachian site. Few significant effects were observed for families or species for the manipulative treatments. All significant effects were increases in numbers captured or in species richness; no significant negative effects were observed.

Vegetation

Huang, J.; Boerner, R.E.J. 2008. **Shifts in plant morphological traits and seed production of *Desmodium nudiflorum* following prescribed fire alone or in combination with forest canopy thinning.** Botany. 86: 376-384.

The researchers investigated the effects of the FFS treatments on *Desmodium nudiflorum*, a common perennial found across moisture and nutrient gradients of the Central Appalachian site. This long-lived perennial was chosen to represent understory herbaceous plants across the landscape. Plots were sampled during the summer on the C, B, and MB treatments at the REMA and ZAL replications four growing seasons after initial treatments and one growing season after the second prescribed fire (2004, 2005). Total plant biomass, seed mass, and seed production were significantly higher under either manipulative treatment. Specific leaf area (leaf area/leaf mass) was significantly lower under the manipulative treatments. This study found the B and MB treatments improve fitness for this species.

McCament, C.L.; McCarthy, B.C. 2005. **Two-year response of American chestnut (*Castanea dentata*) seedlings to shelterwood harvesting and fire in a mixed-oak forest ecosystem.** Canadian Journal of Forest Research. 35: 740-749.

Hybrid American chestnut seeds resistant to blight are soon to be released for outplanting to restore the species on the landscape. To quantify the forest structure most likely to produce successful plantings, 100 pure American chestnut seeds were planted in each of the replications and treatments of the Central Appalachian site prior to the second growing season after initial treatments were installed. Eleven seedling variables were measured for two growing seasons including leaf, stem, root, fine root, and total biomass; leaf area; stem height; basal diameter; and root length. All of these variables were significantly and positively affected by each of the manipulative treatments, with the exception of root length in the B treatment. The authors conclude the proper environment to reintroduce hybrid American chestnut into the Appalachian forest would include open stands, possibly with a prescribed fire regime.

Joesting, H.M.; McCarthy, B.C.; Brown, K.J. 2007. **The photosynthetic response of American chestnut seedlings to differing light conditions.** Canadian Journal of Forest Research. 37: 1714-1742.

Using a subsample of the American chestnut seedlings in the paper cited above, the authors investigated the photosynthetic capabilities of leaves in the C and M treatments of the ZAL and REMA replications of the Ohio site. The leaves of seedlings growing in the M treatment assimilated CO_2 at higher rates than leaves from seedlings in the C treatment at all but the lowest light levels. There was no significant difference in the amount of N in the leaves between treatments and the amount of N was not correlated to any photosynthetic variables or canopy openness. The authors recommend forests with higher light levels than found in undisturbed forests for successful reintroduction of hybrid American chestnut.

Schelling, L.R.; McCarthy, B.C. 2007. **Seed banks of mixed oak forest: effects of forest management on species composition and spatial pattern.** Natural Areas Journal. 27: 320-331.

Soil samples were collected 3 years following initial treatment implementation on the ZAL and REMA replications of the Central Appalachian site to investigate the effects of the FFS treatments on the soil seed bank. Species composition of the seed bank contained in these samples did not differ significantly between treatments. As other studies have found, the composition of the seed bank was significantly different from the aboveground composition on all treatments, implying that many species present aboveground do not rely on banked seeds for regeneration success.

Chiang, J.-M.; McEwan, R.W.; Yaussy, D.A.; Brown, K.J. 2008. **The effects of prescribed fire and silvicultural thinning on the aboveground carbon stocks and net primary production in an oak-hickory ecosystem in southern Ohio.** Forest Ecology and Management. 255: 1584-1594.

With the concern over the carbon sequestration potential of forests, the authors examined the potential effects of the silvicultural treatments on carbon stocks. Annual net primary production (ANPP) was calculated for the treatments of the Ohio site for the first 4 years of the study. ANPP significantly decreased in the M and MB treatments the first year following harvesting due to the reduction in basal area. However, the decrease in ANPP did not persist into the second through fourth years, implying an equivalent amount of biomass was being produced with fewer trees than on the C and B treatments. Prescribed fire had no persistent effect on ANPP.

Lombardo, J.A.; McCarthy, B.C. 2008. **Silvicultural treatment effects on oak seed production and predation by acorn weevils in southeastern Ohio.** Forest Ecology and Management. 255: 2566-2576.

To quantify the effects of the FFS treatments on mast production, acorns were collected from 72 chestnut oak and 72 black oak trees evenly distributed within the treatments of the ZAL and REMA replications of the Central Appalachian site for the five growing seasons following initial treatment implementation. Black oak acorn production did not vary among the treatments and was greater and more consistent than the masting levels of the chestnut oak. After the second prescribed fire, the chestnut oaks in the M and MB treatments produced significantly more and smaller acorns than the chestnut oaks in the C and B treatments.

Yaussy, D.A.; Waldrop, T.A. In press. **Delayed mortality of eastern hardwoods: a function of fire behavior, site, or pathology?** In: Stanturf, J.A., ed. Proceedings, 14th biennial southern silvicultural research conference. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station.

Overstory mortality was modeled using survival analysis for both Appalachian sites to investigate the effects of the FFS treatments on stand structure and composition. Mortality increased on the B and MB treatments the first growing season after prescribed fire and this increase persisted until the next fire. Mortality on the B and MB treatments increased even more after the second prescribed fire. The analysis indicated that maximum thermocouple temperature, preburn crown vigor, bark thickness, and duration of heating were the variables that explained most of the differences in mortality among treatments. Intense or slow-moving fires increase mortality of overstory trees, especially those of low vigor and thin bark, and the mortality may occur several years after burning.

Iverson, L.R.; Hutchinson, T.F.; Prasad, A.M.; Peters, M.P. 2008. **Thinning, fire, and oak regeneration across a heterogeneous landscape in the eastern U.S.: 7-year results.** Forest Ecology and Management. 255: 3035-3050.

The authors wished to evaluate the longer-term effects of the FFS treatments on canopy openness and tree regeneration. Data were collected at the grid points of the C and MB treatments at the REMA and ZAL replications of the Ohio site, pretreatment, and during the first, fourth, and sixth growing season after treatment initiation. Abundance of small trees (<4 in d.b.h.) is reported for five size classes and several species across a moisture gradient. Oak-hickory regeneration > 20 in tall, was less than 130 stems/ac on the MB grid points prior to treatment initiation. Two growing seasons after the second prescribed fire, oaks and hickories in these size classes averaged more than 930 stems/ac at dry gridpoints, and more than 685 stems/ac at intermediate gridpoints. There was little change at the mesic grid points. Each grid point was assigned an oak-hickory competition class based on density of seedlings, density of saplings, and competition. Maps are used to demonstrate the increase in oak-hickory competitiveness across the landscape as the result of moisture gradient and change in canopy openness.

Waldrop, T.A.; Yaussy, D.A.; Phillips, R.; Hutchinson, T.F.; Brudnak, L.; Boerner, R.E.J. 2008. **Fuel reduction treatments affect stand structure of hardwood forests in Western North Carolina and Southern Ohio, USA.** Forest Ecology and Management. 255: 3117-3129.

To synthesize several years of data collection, summaries of the effects of the FFS treatments on vegetation over time are reported for both the Central and Southern Appalachian sites for up to 5 years after initial study installation. Data following the second prescribed fire at the Ohio site were not available for this analysis. No attempt is made to compare the effects of treatments between the sites. The vegetation variables reported include basal area, mortality, density of regeneration and saplings (by species), shrub cover (by species), and ground layer cover (by plant type). Differences among treatments within sites can be explained by increased light availability (reduction in basal area), time since prescribed fire, and intensity of prescribed fire. Both sites indicate an increase in oak saplings on the MB treatment after the first prescribed fire.

CONCLUSIONS

The FFS study was designed as a long-term investigation of the ecosystem consequences of the reintroduction of a fire regime or its mechanical surrogate. So far, the results from these two sites have shown that the treatments do little harm to the insects, wildlife, and belowground processes and may provide some benefits to many ecosystem components such as plant biodiversity. Significant large oak regeneration has developed at both sites with the MB treatment after two prescribed fires.

Although more than 50 peer-reviewed articles have been published concerning the Southern and Central Appalachian sites of the FFS study, the results are preliminary. The treatments have changed the structure and composition of the forest, from the soils to the canopy. How long will these changes persist and what effects will a periodic burning regime have on the ecosystem? Currently, both sites are in a maintenance mode, where periodic prescribed fires are implemented every 3 to 5 years and rudimentary vegetation data are collected. Detailed fuel, wildlife, insect, and soils information will no longer be collected unless funding is obtained. The long-term effects of these restoration treatments on the ecosystem components for which the study was designed will be forfeited.

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FIRE AND THE ENDANGERED INDIANA BAT

Matthew B. Dickinson[1], Michael J. Lacki[2], and Daniel R. Cox[2]

[1] Ecologist, U.S. Forest Service, Northern Research Station, Delaware, OH
[2] Professor (ML), Adjunct Assistant Professor (DC), Department of Forestry, University of Kentucky, Lexington, KY
MBD is corresponding author: to contact, call (740) 368-0093 or email at mbdickinson@fs.fed.us

Abstract.—Fire and Indiana bats (*Myotis sodalis*) have coexisted for millennia in the central hardwoods region, yet past declines in populations of this endangered species, and the imperative of fire use in oak silviculture and ecosystem conservation, call for an analysis of both the risks and opportunities associated with using fires on landscapes in which the bat occurs. In this paper, we explore the potential direct effects of prescribed fire and associated smoke on Indiana bats. We identify the immediate effects on bats, such as exposure to smoke and displacement, when individuals are in tree roosts (under exfoliating bark or in crevices) and hibernacula (caves and mines). Radio-tracked northern long-eared bats (*Myotis septentrionalis*), an Indiana bat surrogate, flushed shortly after prescribed fire ignition in the Daniel Boone National Forest (Kentucky) on a warm spring day, confirming previously reported observations. We also consider the longer-term effects on bats of the habitat changes caused by fire use. Finally, we review National Forest Plans and ask how the available science supports their standards and guidelines. Efforts to manage Indiana bats are based on limited monitoring of the effects of habitat manipulations and a body of research that is deficient in key areas, providing a poor basis on which to either practice adaptive management or counter restrictions on growing-season burning.

INTRODUCTION

The Indiana bat (*Myotis sodalis*) is a rather small (6 to 9 g), aerially feeding (insectivorous), tree-roosting, and migratory bat with a summer distribution that encompasses much of the Midwest, and portions of New England and the south-central states (Harvey et al. 1999, U.S. Fish and Wildlife Service [USFWS] 2007a). Tree roosting occurs under exfoliating bark and in crevices in dead trees (snags) and live trees. First reproduction for healthy females occurs in their first year and females can give birth to a single pup each year for 14 to 15 years (Humphrey et al. 1977, Humphrey et al. 1977). Pregnant females and females with young aggregate in maternity colonies of tens to hundreds of individuals. The bat's distribution is believed to be associated with the prevalence of limestone caves in the eastern United States (Menzel et al. 2001). Today, hibernacula include both caves and mines.

The species was listed as endangered on March 11, 1967, after winter populations declined significantly at the majority of known hibernacula (USFWS 2007a). Early recovery efforts focused on protection and rehabilitation of hibernacula, yet declines in populations of the species continued with an estimated 350,000 individuals remaining as of the 1997 census. Through the period of declines, losses in southerly hibernacula have been partially offset by increases in northerly hibernacula (Clawson 2002). Trends since 2001 have been upward, and 2007 population estimates reached approximately 468,000 (USFWS 2008). On the negative side, mortality of Indiana bats associated with white-nose syndrome in the Northeast over the winter of 2007-2008 has been confirmed (USFWS 2009c).

Indiana bat foraging and roosting habitat includes forested areas outside of hibernacula used during the period of transition out of hibernation (staging), areas used during the warmer months of the year (summer habitat), and, again, areas outside of hibernacula used during the breeding period before hibernation (swarming). Indiana bats either stay close to hibernacula during the summer or migrate, with a maximum known migration distance of 520 km (Gardner and Cook 2002). Males and females have different roosting requirements during the summer; female Indiana bats form maternity colonies that use collections of snags and live roosts (Humphrey et al. 1977, Kurta et al. 1993, Callahan et al. 1997, Britzke et al. 2003, 2006). During the swarming period, bats breed and must gain weight rapidly to prepare for hibernation (USFWS 2007a).

Implementing the widely advocated prescription for controlled burns to restore and maintain oak ecosystems (see papers in Dickinson 2006, Yaussy et al. 2008), the U.S. Forest Service (USFS) is carrying out prescribed fire treatments on oak forests, including those that occur within the distributional limits of the Indiana bat (e.g., Alexander et al. 2006, Lyons et al. 2006). For example, on the Daniel Boone National Forest, plans call for burning upwards of 22,700 ha/year within the next decade (Mann 2006). Use of growing-season burning as a silvicultural and ecological restoration tool in mixed-oak forests has been proposed and causes concern about increased risks to bats from the direct effects of smoke, yet holds promise for improving bat habitat. Growing-season burning is an emerging issue for which scientific information is needed on attendant risks and opportunities for bats.

Widespread use of fire might have both direct (short-term) and indirect effects on bats. Direct effects include displacement, injury, and mortality. During fires, roosting bats may be exposed to heat from flames and the smoke plume. In addition, elevated concentrations of the products of combustion (e.g., carbon monoxide and irritants) occur over burn units and through the canopy. Exposure depends not only on fire behavior but also on roost characteristics, bat behavior, whether the bats are in torpor (a diurnal hibernation-like state, marked by a temporary decline in body temperature and metabolic rate), and bat gender and age. Risks from heat and gases probably would be greatest for nonreproductive bats in torpor, particularly if they are roosting near the ground or on dry landscape positions where smoke exposures are likely to be greatest. Reproductive female bats also use torpor though they appear to do so during conditions that would often not be conducive to burning. Arousal of bats from exposure to smoke in hibernacula is a concern relative to dormant-season burning. Bats arouse from hibernation periodically as a normal course of affairs, possibly because of the need to rehydrate, but each arousal is energetically equivalent to many days of hibernation (e.g., approximately 60 days for little brown bats [*M. lucifugus*] in the laboratory), and extra arousals from smoke exposure or other causes are a serious concern (Thomas et al. 1990, USFWS 2007a).

Indirect effects of fire on bats arise from fire-induced habitat change. Fire use is generally advocated as a way of improving bat habitat, through snag production, creation of more open stands preferred for foraging, and increased insect abundance and diversity (e.g., USDA FS 2003, USFWS 2007a). Fire, alone or in combination with thinning, may affect bat roost availability. Fires both create and destroy snags, with unknown long-term effects in eastern hardwood forests. Fires would be expected to reduce insect prey abundances in the short run, but their long-term effects on prey abundances are unknown.

Krusac and Mighton (2002) estimate that 5.5 million ha of lands managed by the USFS occur within the distributional limits of the Indiana bat, though not all forests contain summer habitat. The discovery in 1994 of a reproductive female bat on the Daniel Boone National Forest galvanized search and discovery of more reproductive females on other forests and, by way of the Endangered Species Act (1973), formal consultation between Forests and the USFWS. Since 1994, most National Forests within the range have revised their forest plans, including standards and guidelines intended to maintain and improve bat habitat and reduce risk to bats from management activities such as fire. State and private lands, which, combined, represent the majority of forest area in the East, presumably offer important habitat for bats, but little information is available on their importance for Indiana bats. Coordination efforts between state and private land managers and the USFWS have just begun.

Identifying the factors that have caused both the historical declines in Indiana bat populations, as well as recent increases, is central to recovery efforts (e.g., Caughley and Gunn 1995, USFWS 2007a). Authors have identified hydric habitats in the Midwest as core habitat supporting more robust maternity colonies than those that occur in upland forests, yet these habitats are severely diminished by conversion to agriculture and other land uses (e.g., Carter 2005). Research in the forested Appalachians has shown that foraging bats use riparian habitats more heavily than they use such upland sites as undisturbed stands and stands impacted by an array of forestry practices (Owen et al. 2004, Ford et al. 2005). Toxic effects of agricultural and other pesticides have potential, but unproven, negative consequences for bats (USFWS 2007a). The hibernation phase continues to be of concern because of bats' exacting microclimatic requirements; changes in hibernation patterns, perhaps owing to climate change; responses to ongoing cave disturbance; and, most recently, succumbing to white-nose syndrome during hibernation. Research and monitoring should continue to focus on the effects of bottomland hardwood habitat loss, pesticide toxicology, and hibernation problems on Indiana bat populations in order to identify the most effective management actions.

While hydric and riparian habitats are crucial to Indiana bats' life cycle, upland habitats are also used by Indiana bats for roosting and foraging during swarming, staging, and maternity periods. Furthermore, upland habitats are connected ecologically and hydrologically with riparian and hydric habitats. Because of the bat's endangered status, considerable efforts, including forest burning, are being made on Federal lands to maintain and improve summer habitat for Indiana bats, yet it is not possible to say whether these efforts are leading to gains in Indiana bat populations and, if so, why. Given low rates of upland burning relative to historical levels, it is worth considering whether fire suppression has a role in bat population declines and whether seasonal restrictions on burning are counterproductive. In this paper, as the data permit, we focus on the direct and indirect effects of fire on Indiana bats in upland mixed-oak habitats. We first report results of a field study on behavior of an Indiana bat surrogate species during and in the days after a prescribed fire and the results of a literature survey of Indiana bat roost characteristics and roosting behavior. Then, we discuss potential short-term effects of upland fires on bats and the longer-term effects of fire on bat habitat (see also Carter et al. 2002). Finally, we examine National Forest Plan standards and guidelines with relevance to fire management and adaptive management of Indiana bats.

OBSERVATIONS ON NORTHERN LONG-EARED BAT RESPONSE TO FIRES

The objectives of the following study were to document short-term behavioral response of tree-roosting bats to a prescribed fire. Longer-term responses will be reported in a forthcoming paper. Because no Indiana bats were captured as a part of this study, we chose the northern long-eared bat (*M. septentrionalis*) as a surrogate for the Indiana bat. The Indiana bat and the northern long-eared bat overlap in distribution, are similar in body mass and wing morphology, and form maternity colonies in live trees and snags during summer months (Foster and Kurta 1999, Carter and Feldhamer 2005, Lacki et al. 2008).

Methods

Study Area and Burn Unit

The Bear Waller unit is located in Red River Gorge Geological Area, Daniel Boone National Forest, Kentucky (37° 51'N, 83° 39'W). The terrain of the region is characterized by dissected valleys, steep ridges, cliffs, and rocky outcrops, with elevations ranging from 200-365 m (McGrain 1983). The forest community is typical of the Cumberland Plateau physiographic region in eastern Kentucky. Forest composition is primarily mixed mesophytic species,including several oaks (*Quercus* spp.), American beech (*Fagus grandifolia*), cucumber magnolia (*Magnolia acuminata*), maples (*Acer* spp.), tulip tree (*Liriodendron tulipifera*), white ash (*Fraxinus americana*), eastern hemlock (*Tsuga canadensis*), and several pine species (Jones 2005). The climate is moderate with average temperatures ranging from 16.6 °C to 22.9 °C from May to August and an average annual precipitation of 101 cm.

The burn unit encompassed approximately 185 ha of uneven terrain and supported second-growth forest at the time of the burn event. A prominent ridgeline traverses the unit, where a storm blow-down occurred in the recent past. Thus, much of the ridgeline was overgrown with thick stands of greenbrier (*Smilax rotundifolia*) and red maple (*A. rubrum*) prior to burning. Two wildlife ponds are located in the immediate area, one of which is situated directly within the burn unit and is surrounded by a closed canopy of upland hardwoods. This pond served as the focal point for capturing bats. The burn unit was selected in collaboration with U.S. Forest Service personnel to meet both research and management needs. The burn unit had no history of management with prescribed fire; however, reports exist of "numerous fires having burned" within the original Cumberland purchase unit prior to 1930 (Collins 1975). An 8-ha wildland fire was documented near the center of the burn unit on Oct. 30, 1994.

The experimental burn occurred on April 30, 2007 and was conducted by Daniel Boone National Forest, Cumberland Ranger District staff. The ignition pattern consisted of igniting ridgelines with a drip torch and allowing the fire to back down slopes. Flame heights ranged from 0.2 to 2 m, but were typically <1 m. At the time of burn, new seasonal vegetative growth was abundant in the under- and overstory. Scattered smoldering and mid- and lower-slope flaming on the day after the burn produced low-level smoke exposures on ridges (see below); thus, the burn can be considered an early growing-season burn. Scattered snags and downed woody debris were found smoldering on the unit as late as May 9, 2007. For smoke monitoring, carbon monoxide sensors (Sixth Sense, Inc. Eco-Sense 2e electrochemical sensors with a custom electronics signal conditioning board) were placed at 2.4 and 6.1 m above ground on towers at three ridge locations within the burn unit.

Bat Capture and Tracking

We captured bats from April 22 to April 29, 2007 using nylon mist nets (Avinet Inc., Dryden, NY) of varying widths. Nets were placed over the wildlife pond at the interior of the burn unit. We recorded gender, reproductive condition, body mass, and forearm length of each captured bat. Our netting effort in the surrounding area in summer 2006, along with 2007, resulted in no Indiana bat captures, so we chose to use the northern long-eared bat as a surrogate.

We fitted five adult northern long-eared bats (four females and one male) with 0.36 to 0.42 g transmitters (LB-2N, Holohil Systems Ltd., Carp, ONT) between the shoulder blades using Skinbon® adhesive cement (Smith and Nephew United, Largo, FL). We held bats with transmitters for 20 to 30 min to allow the adhesive to form a secure bond between transmitters and the dorsal surface of bats. Before release, we observed bats to ensure normal behavior and, thus, the safety of the animals, and to verify that transmitters were working properly. Transmitter load was 5.8 to 8 percent of the bats' body mass.

We tracked radiotagged bats to roost trees each day with TRX-1000S receivers and three-element yagi antennas (Wildlife Materials Inc., Murphysboro, IL). Tracking continued until a transmitter battery failed or the transmitter was shed by the bat. Transmitters lasted for approximately 8 to 10 days and we report behavior up to 7 days post-burn in this paper. We determined coordinates for each roost tree using a geographic positioning system (GPS). For each roost tree we recorded species, whether the tree was alive or dead, height of the tree (m), and diameter of the tree (cm). When possible, we measured height of the roost (m), noted whether the roost was beneath a plate of bark or inside a crevice or cavity, and counted the number of bats exiting the roost on the night it was first discovered.

On the day of the prescribed burn, we located roost trees of each radiotagged bat prior to ignition at 1620 EST. Afterwards, two crew members were outfitted with receivers and stationed near roost trees known to be occupied by two of the radiotagged bats. A USFS Safety Officer monitored fire and smoke conditions near the telemetry crew to ensure a safety zone and an exit route. For safety reasons, the telemetry crew was required to work close to each other; thus, behavior of only two bats was monitored during the burn. Monitoring consisted of recording time of emergence, when bats were roosting or in flight, flight patterns, time spent in flight, and any other observations deemed pertinent. We monitored behavior of these bats until 1930 EST, at which time the general roosting positions of all five radiotagged bats was determined. On the day of the burn, we could not identify the specific roost trees of all bats prior to their exiting for nightly foraging because of time constraints and inaccessibility to habitats due to fire; however, by determining general roosting areas after the burn, we were able to confirm whether individual bats relocated during the fire.

Table 1.—Sex, reproductive class, body mass, and spatial data for northern long-eared bats radiotracked on and adjacent to the April 30, 2007, Bear Waller burn on the Daniel Boone National Forest, KY

Sex/ ID no.	Female reproductive class	Body mass (g)	Home range size (ha[a])	Pre-burn % locations (# locations)		Post-burn % locations (# locations)	
				on unit	off unit	on unit	off unit
Female (B5)	nonreproductive	5.25	64.9	100 (32)	0 (0)	75 (30)	25 (10)
Female (B9)	nonreproductive	5.25	59.1	no data[b]	no data	100 (24)	0 (0)
Female (B10)	nonreproductive	6.0	-[c]	no data	no data	no data	no data
Female (B11)	Pregnant	7.25	57.5	no data[a]	no data	23 (11)	77 (37)
Male (B7)	-	5.75	56.8	86 (32)	14 (5)	100 (45)	0 (0)

[a]Conversion to acres: 1 ha = 2.47 ac.

[b]Bats B9 and B11 were captured the night prior to the burn, so no pre-burn foraging data are available.

[c]Bat B10 was not located during nightly foraging in the vicinity of the burn unit.

Home Range and Habitat Use

We used triangulation to determine location of radiotagged bats during nightly foraging. Triangulation began after bats left their respective roosts and continued until at least midnight each night. Two to three crew members were stationed at high elevation locations and recorded their position with a GPS. Simultaneous azimuths were taken and estimated foraging locations were derived by means of triangulation (White and Garrott 1990). We recorded azimuths at 3 to 5 min intervals and communicated via hand-held radios. We tracked individual bats in alternating 30-min time periods. Foraging data were collected on nights before and after the burn. We did not collect foraging data on the first night after ignition as a safety precaution.

We entered telemetry station locations and azimuths into the Locate 3.19 program to determine bat locations (Nams 2006). In all cases, two azimuths were used to determine a bat location. Studies with other animal species have shown that use of >2 azimuths does not necessarily increase accuracy or precision when radiotracking (Nams and Boutin 1991). We limited home range calculations to bats with ≥ 30 locations, although 50 locations are considered to be optimal (Seaman et al. 1999). We used ArcGIS version 9.2 (ESRI, Redlands, CA) to calculate 95-percent home ranges using Hawth's Tools extension version 3.27 (Beyer 2004). We generated pre- and post-burn estimates of home range size for bats where data were sufficient. We evaluated use of the burn unit by bats while foraging relative to timing of the burn. We compared percentage of radiolocations on the burn unit to those off the burn unit before and after the burn and across radiotagged bats using a chi square test of homogeneity (Daniel 1974).

Results

Response of Northern Long-Eared Bats to Fire

Northern long-eared bats demonstrated an average home range of 59.6 ± 1.84 (SE) ha, with a male B7 possessing the smallest home range (Table 1). One female B10 was not recorded after the night of tagging and release. We had sufficient data to derive home range estimates for two bats pre- and post-burn. Female B5 demonstrated a home range of 41.4 ha pre-burn, which increased to 74.7 ha following the burn. In turn, male B7 possessed a home range of 60.9 ha before the burn, which declined to 46.1 ha post-burn.

Use of foraging habitat was significantly different among bats ($\chi^2 = 157$, df = 9, $p < 0.005$). Female B5 and male B7 spent the majority of time foraging over the burn unit, both pre- and post-burn (Table 1). Female B9, not tagged until just before the burn, foraged exclusively over the burn unit following the burn. Only female B11, the lone pregnant female that we tracked, foraged more often off the burn unit than over the burn unit after the burn was completed.

Northern long-eared bats were tracked to nine roost trees of four species, the majority of which were oaks (Table 2). Of these, 77.8 percent were live trees with

Table 2.—Description of roosts of northern long-eared bats located on the Bear Waller burn unit on the Daniel Boone National Forest, KY

Species	Live vs. dead	Roost type (m)	Roost height size	Estimated colony (cm)	Tree diameter (m)	Tree height
Quercus prinus	live	cavity	17.7	29	30.4	22.6
	live	-[a]	-	1	57.1	27.4
	live	crevice	22.9	29	53.5	29.0
Q. coccinea	live	-	-	6	37.3	24.7
	live	-	-	1	49.5	24.4
Q. alba	live	bark	7.6	1	36.5	32.0
Acer rubrum	live	-	-	1	24.3	21.3
Unknown[b]	dead	cavity	7.6	39	18.1	8.5
	dead	crevice	10.4	18	38.8	12.2

[a]Specific roost location could not be identified.
[b]Snags were in an advanced stage of decay, preventing species identification.

bats roosting in cavities and crevices and beneath bark. Diameter of roost trees averaged 38.4 ± 4.37 cm and ranged from 18.1 to 57.1 cm. Height of roost trees averaged 24.3 ± 1.86 m and ranged from 12.2 to 32.0 m. Height of roosts aboveground averaged 13.4 ± 2.96 m and ranged from 7.6 to 22.9 m. Maximum size of maternity colonies varied from 1 to 39 bats, with an average of 15.5 ± 5.41 bats. The lone roost recorded for male B7 was a white oak and he was the only bat observed exiting the tree.

Day of the burn. We were able to locate roost trees for four bats immediately before the burn; however, the roost tree of the remaining bat was known to be within the burn unit. The male and female bats we tracked during ignition operations and the main burning period (B7 and B9) displayed similar behavior. The ignition line ran within 20 m of roost trees of each bat, and both bats exited their roosts within 10 min of ignition near their respective roosts (between 1640 and 1650 EST). The two observers, one monitoring each bat, were approximately 30 m from the respective roost tree. Ambient temperature was approximately 31 °C (88 °F). Both bats flew for about 45 min after initially leaving their roosts, then roosted for about 1 hr. They cycled through periods of flight and roosting during the burn.

Because of burning conditions and safety constraints, we could not determine whether bats returned to their pre-fire roosts, but they did return to the vicinity. Both bats continued with an alternating flying and roosting pattern until sunset, when they emerged to forage. The time the bats spent roosting increased as the day progressed. While in flight, bats concentrated their activity over habitat that the fire had not yet reached, such as upland drainages that were slow to burn. Moreover, both bats were originally roosting on the north side of the burn unit near the ignition line; however, they chose to fly in the area opposite of the ignition line where the backing fire and smoke had not yet reached. We assume that bats were attempting to limit their exposure to conditions created by fire. At no point did they fly outside of their typical home range area, nor did they travel far from the burn itself. Although no behavioral data were collected for the remaining radiotagged bats, these bats behaved similarly to the bats that were monitored, all having switched roosts at some point during the fire. A roost tree that had been used by female B5 before the fire fell after its base was weakened by smoldering combustion; all other roosts remained standing. All bats were located within the burn unit following the burn.

Days 1 and 2 after the burn. Fire was still spreading in some areas of the burn unit the day after the burn and smoke was present on the unit and in adjacent forest. Peak carbon monoxide (CO) concentrations of 50-190 parts per million (ppm) were measured at 2.4 m aboveground at three stations on the second day compared with concentrations of 350 to ≥400 PPM during burning the first day (CO sensors saturated at 400 PPM on one of three stations). Concentrations of CO during the night after ignition approached background. Two sensors at 6.1 m failed so only 2.4 m data are provided as an indication of relative smoke concentrations during the second day. All five bats roosted near the core of the burn unit on the day after the burn. That night we observed very few insects flying within the burned area. We observed male B7 foraging farther away from where he had been typically foraging, moving down-slope, and closer to the burn unit boundary near areas that had not burned. Female B5 foraged normally for the first hour after emergence then altered her foraging behavior from previous nights by foraging in areas where she had not been recorded. She covered a considerable distance, sometimes leaving the range of detection. She returned to her usual foraging/roosting area at 0200 hr. It rained all day the second day following the burn and no data were collected.

Days 3-7 after the burn. It rained sporadically throughout day 3. Logs and stumps remained burning, with continued smoke production and haze. Flying insects were scarce. Female B10 shed her transmitter, and no signal was received on male B7. Although Female B5 was found roosting away from the burn site, we could not find the roost tree. That evening, she exited her roost and was tracked for about 30 min before we lost her signal. She was not recorded again that evening. Steady rain fell during days 4 and 5 after the burn and no tracking was attempted.

Despite the rain, some downed logs remained smoldering on day 6. Flying insects were more prevalent. No signal was received from female B5, but she returned on the evening of day 7 and foraged in the burn unit and surrounding habitats. Male B7 was observed foraging downslope near the burn unit boundary on the evenings of both days 6 and 7. From our observations, overall foraging behavior of all four remaining radiotagged bats during the evenings of days 6 and 7 appeared to return to pre-burn norms in terms of emergence time, length of foraging bouts, and use of the burn unit and adjacent habitats.

INDIANA BAT ROOSTING CHARACTERISTICS

We review studies which report information on roost trees in order to better quantify what constitutes a quality roost and to assess bat risk during fires relative to the aboveground height at which they roost.

Literature Review Methods

We used data available in published studies on summer habitat of the Indiana bat to examine frequency distributions of roost tree diameter and height and the height of roosting sites above ground (sources are provided in Lacki et al. 2008). We combined data from all habitats across the range of the Indiana bat, though lack of data prevented us from comparing maternity and other roosts and roosting behavior by gender. We evaluated predictive capability of roost characteristics by developing regression models of roost height as a function of tree diameter and tree height using mean values per published study and, where available, for individual roost trees (SAS Institute, Inc., Cary, NC, 2003).

Results

Too few data were available for male roosts and roosts used by both males and females to allow any comparison with female-only roosts. Hereafter, we focus on female-only roosts (primary maternity and other roosts combined). Individual roost data demonstrated that roost trees selected by female Indiana bats exhibited a modal peak at 40-cm diameter (Fig. 1a). Bats used trees of a wide range of heights, with a mode at 25 m (Fig. 1b). Height at which Indiana bats roosted demonstrated a distribution that was skewed left, with the modal peak at 10 m (Fig. 1c). The studies that report individual roost information, and which provided data for Figures 1a-c, largely included bottomland hardwoods and trees in swamps and it is not clear how relevant these data are to uplands. To avoid habitat-related bias, mean heights at which bats roosted reported in studies from across the bat's range were averaged, providing a mean roosting height of 9.12 m (N=13 studies, SD=2.14, 95% CI 4.92 to 13.3 m). From the few studies in which roosting heights were reported (Fig. 1c), the mean roosting height was 8.03 m (N=18 studies, SD=3.27, 95% CI 1.62 to

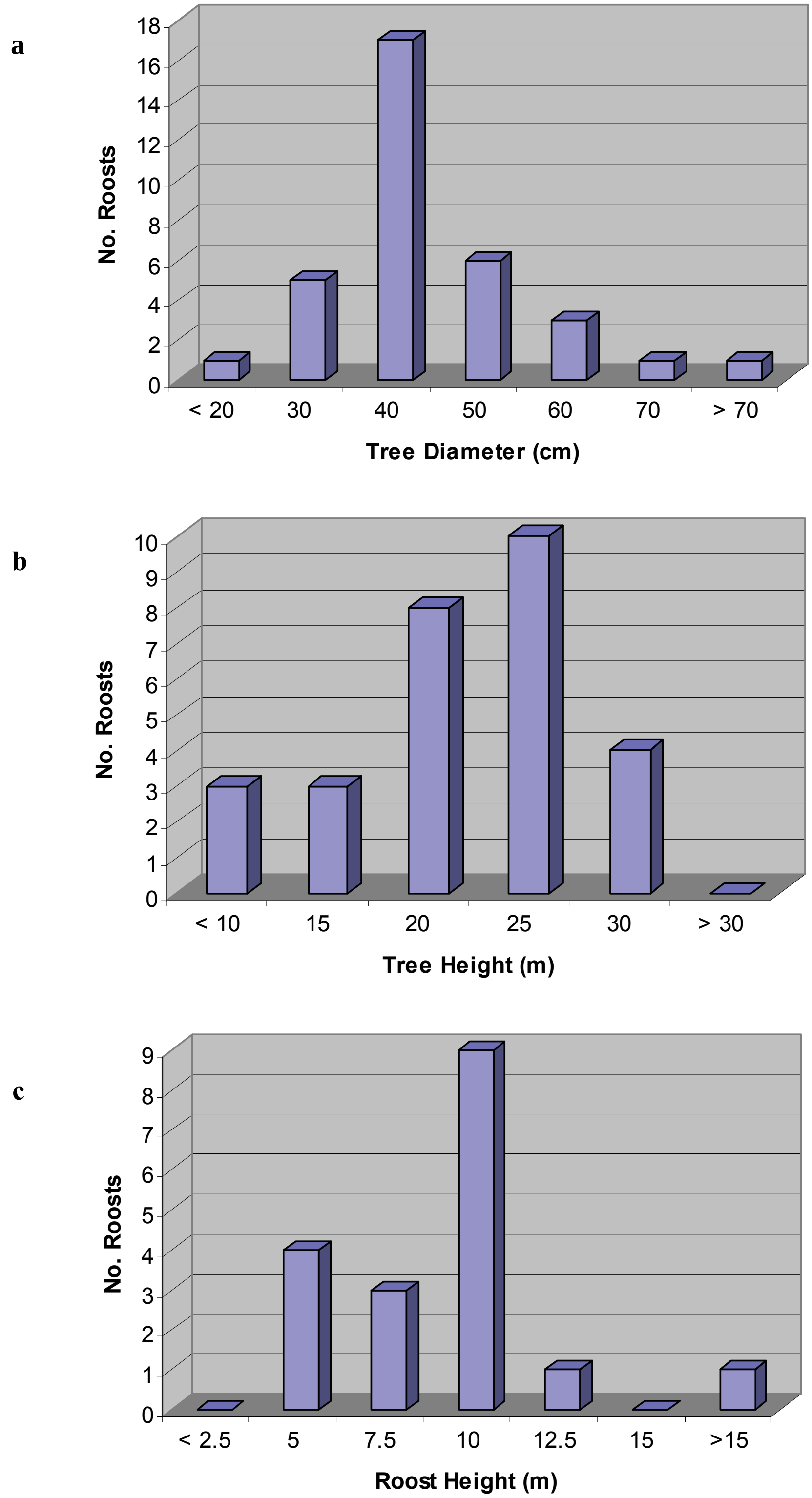

Figure 1.—Frequency distributions of female Indiana bat roost trees (maternity roosts and other) by (a) tree diameter, (b) tree height, and (c) roosting height aboveground. Sources for data are provided in Lacki et al. (2008).

Table 3.—Regression models predicting height of roosts aboveground for Indiana bats based on tree diameter (cm) and tree height (m). Models were developed using mean values per population, and data for individual roost trees where available. Published sources are from Lacki et al. (2008).

Form of data	Regression	R^2	F-value	P-value
Mean values per population	Roost height = 0.163(treediameter) + 2.75	0.27	4.15	0.07
	Roost height = 0.363(treeheight) + 1.77	0.33	5.47	0.04
Data for individual roost trees	Roost height = -0.04(treediameter) + 9.56	0.01	0.19	0.67
	Roost height = 0.135(treeheight) + 5.67	0.06	0.96	0.34

14.4 m). The lowest female roosting heights reported are 2 m (USFWS 2007b), 3.0 m (Belwood 2002), and 4.0 m (Butchkoski and Hassinger 2002).

From the few data that were available, male Indiana bat roost diameter averaged 34.0 cm (N=5, range 14-61 cm), tree height averaged 21.7 m (N=3), and roost height averaged 8.33 m (N=3). The diameter of combined male and female roosts averaged 33.2 cm (N=3) and there were no data for roost tree height and roosting height.

Regression models predicting the height aboveground of Indiana bat roosts demonstrated significant relationships with tree diameter and tree height when mean values per population were used for analysis (Table 3). Regressions based on these independent variables accounted for similar percentages of variation in roost height as indicated by R^2 values. These relationships did not hold up when data from the few studies where individual roost tree information was used for analysis, accounting for no more than 6 percent of the variation in height of roosts aboveground.

DISCUSSION

Short-term Effects of Fires on Tree-Roosting Bats

Smoke Effects in Summer Habitat

In prescribed surface fires, bats in roosts are exposed to gases and heat in the plume generated by the spreading fire. Exposures would depend on how high bats roost aboveground, bat physiological condition and flushing behavior, behavior of the fire (e.g., fuel consumption and fireline intensity), winds, terrain, and whether the canopy is leafless. If bats are in torpor or if flightless pups are present and too heavy to be carried, one can assume full exposure. Hot gases in the plume are mixed rapidly into the types of roosting sites (e.g., exfoliating bark, crevices) used by Indiana bats; these kinds of sites afford little protection (unpublished data, Guelta and Balbach 2005). Based on extrapolation from other species, incapacitation from carbon monoxide exposure would be expected if bats were exposed to $\geq$1000 PPM concentrations for 25 minutes or more and for shorter time periods if concentrations were higher (Spietel 1996). From our unpublished field data, incapacitating exposures are highly unlikely at any height above flames. Acting to reduce risk, irritants in smoke (e.g., acrolein, formaldehyde) cause an immediate reduction in breathing rate and depth (Chang et al. 1981). Elevated CO_2 levels would cause the opposite effect, but only if exposures were longer than several minutes (Purser 2002). Low breathing rates during torpor (e.g., Morris et al. 1994) would also reduce bat exposures to harmful gases.

A greater risk for bats might be external (skin) burns, which are a function of gas temperatures, their flow velocity, and skin properties (e.g., Diller et al. 1991). Skin burns could be a significant risk for bats roosting close to the ground and for those roosting at average heights above fires of intensities on the high end expected from prescribed burns (unpublished data). An ongoing project is analyzing smoke production and transport across burn units and bat toxicology for a range of prescribed fire scenarios.

From available observations, it seems reasonable to assume that bats exposed to smoke would flush if they could. All four northern long-eared bats located by radio tracking before and after a fire changed roosts during the burning period. A male and female bat tracked during the fire flushed within 10 minutes of nearby ignition, suggesting that both females and males were able to emerge from roosting sites in sufficient time (<10 min) to avoid direct impacts from heat and smoke produced by this prescribed fire conducted on a warm day. Roosting heights and roost characteristics for these northern long-eared bats were similar to those reported for Indiana bats, though most roosts were live trees (compare Table 2 and Fig. 1; see Lacki et al. 2008). Observations from other fires confirm our findings on bat movement during fires, for instance Rodrigue et al. (2001) report flushing of a roosting *Myotis* bat. Red bats (*Lasiurus borealis*) are a particular concern because they hibernate in the leaf litter during cold weather and have been shown to flush, or attempt to flush, from in front of fires (Saugey et al. 1989, Moorman et al. 1999). Rodrigue et al. (2001) also observed two red bats leaving a burn unit during a fire. Female bats are able to carry their young for some time after birth, which may reduce vulnerability (Carter et al. 2002). A drawback of flushing from fires is that bats may experience increased predation risk (Carter et al. 2002).

Risk Associated with Torpor

Risk to bats should be, in part, dependent on whether they are in torpor at the time of the burn and whether they can perceive a fire and arouse. Red bats, a species that roosts or hibernates on the ground during cold periods in oak forests, where fire is frequent, were shown to arouse in the laboratory at a 5 °C ambient temperature within 10-40 min in response to a combination of the sound of fire and smoke exposure (Scesny 2006). Little brown bats' arousal at 5 °C body temperature occurred in an average of 44 min (Thomas et al. 1990). Arousal time in a small insectivorous marsupial was found to increase exponentially as body temperatures tracked declining ambient temperatures (Geiser 1986); presumably, a similar pattern holds for bats (Carter et al. 2002).

Small mammals that use torpor often use energy available in their environment to passively maintain high body temperatures and facilitate rewarming (Hamilton and Barclay 1995, Lovegrove et al. 1999, Geiser and Drury 2003, Geiser et al. 2004). High solar exposure at Indiana bat maternity roosts aids in maintaining high body temperatures (USFWS 2007a). In contrast, male red bats in the Ouachita Mountains of Arkansas were found to prefer microsites with low solar exposure (north slopes and drainages) during hibernation in leaf litter, presumably because of the cooler and more constant temperatures they provided (Saugey et al. 1989).

It would be expected that pregnant or lactating female Indiana bats would use torpor less often than males and nonreproductive females because of a need to sustain high metabolic rates. Data from Kurta et al. (1996) demonstrated that adult female Indiana bats in Michigan sustained body temperatures of 35 °C for up to 12 hrs inside diurnal roosts and some bats sustained temperatures at that level for up to 6 consecutive days, suggesting that these individuals would be able to respond fairly quickly to an oncoming fire. On the other hand, studies on other species have demonstrated declines in body temperatures in reproductively active female bats after diet restriction, such as would happen after poor foraging success (Kurta 1991, Audet and Thomas 1997). Willis et al. (2006) demonstrated multi-day bouts of torpor in pregnant female bats during spring storms just prior to giving birth. Thus, at least periodically, maternity colonies may be at increased risk from fire because adult females may be in torpor. However, periods during which torpor is most likely would seem to coincide often with cool and/or wet periods and, thus, poor burning conditions.

Field studies found that male and nonreproductive female big brown bats (*Eptesicus fuscus*) select cooler roosts than reproductive females (Hamilton and Barclay 1995) and males enter torpor more regularly than reproductive females (Grinevitch et al. 1995). Given that male Indiana bats have a tendency to roost in smaller trees that are less exposed to solar radiation (Kurta 2005), we may assume that male Indiana bats also use torpor regularly. More data on roost microclimates and torpor dynamics for bats inside tree roosts and their relation to prescribed burn restrictions and prescriptions are needed to address issues of roost site selection and vulnerability of bats to prescribed fire (Boyles 2007).

Roost Characteristics

Roost characteristics, such as height above ground, snag condition, and landscape position, would also influence risk from fire. Because plumes radiate heat and mix rapidly with ambient air, the higher the roosting location on a snag or live tree, the lower the expected exposure to heat and gases, given similar fire behavior and ambient weather conditions. Our review of studies that provide Indiana bat roosting data (above, Lacki et al. 2008), particularly studies that compare roosts with random potential roosts (see Kurta 2005), support the expectation that reproductive and nonreproductive bats prefer larger than average, but otherwise suitable, roost trees. For fire effects, this preference is potentially important in that roosting locations in large trees tend to be higher above ground (Table 3, Kurta et al. 2002). Even though we found considerable variability in Indiana bat roost-tree height, data suggested that a minimum height of approximately 10 m is necessary before a live tree or snag is sufficiently tall to be desirable as even a secondary roost tree. Decay patterns of snags often lead to the loss of the top of the stem, reducing the height of the snag (Hunter 1990).

Fire Behavior

Smoke exposures determined by fire behavior are a realm over which the fire manager has substantial control through choice of burning conditions and firing methods. Fire managers are well versed in using ignition to influence fire behavior and fire effects on vegetation. In an ongoing fire-monitoring project on Appalachian landscapes, we documented the dramatic differences in heat and smoke release rates from ridge ignition (where the bulk of behavior is low-intensity backing spread) and a combination of helicopter ignition and strip head firing (where more high-intensity uphill runs occur). On flat ground, low heat release rates (where heat release rate is proportional to flame length and fireline intensity, kW/m) and the presence of wind will result in reduced exposures to high temperatures in plumes (Mercer and Weber 1994).

Terrain complicates exposures because of upslope flow of smoke and potential positive feedbacks between the plume and the fire. Smoke rising off burn units in a single plume core would be expected to cause increased gas and heat exposures in the canopy where the plume is centered relative to exposures resulting from smoke that is distributed among multiple plume cores (Achtemeier, this volume). In hilly landscapes, we would expect plume cores to be located along ridgelines above dry slopes and result in the greatest smoke exposures in those locations. We would expect lower exposures to daytime smoke at roosts at lower elevations and in landscape positions that provide topographic shading during the fire season (e.g., north-facing slopes during the winter and early spring).

Choice of burning weather and season are well-known methods of manipulating fire behavior and, thus, smoke exposures. Burning weather, of course, affects fuel moisture and winds, both key determinants of smoke production rates and transport. Evidence suggests that growing-season burns in mixed-oak forests are more effective as a tool for control of oak competition than dormant-season burns (Brose and Van Lear 1998), and should often be less intense because of more fuel shading, lower litter loads, and higher humidity. In addition, bats in torpor during the growing season may arouse more quickly than in the dormant season given higher ambient temperatures. An interesting feature of growing-season burning relative to smoke exposures will be the effect of the canopy, with smoke expected to disperse less readily through a leafed-out canopy in general and to exit the canopy preferentially through gaps.

Smoke Exposures in Hibernacula

Smoke intrusion into hibernacula is a concern because of the potential for inducing arousal in hibernating bats. Though no Indiana bats were present, one study documented smoke intrusion into hibernacula in Missouri, but no arousal was observed (Caviness 2003). Except at the northernmost part of the distribution of Indiana bats, suitable hibernacula require chimney-effect airflow and large cold-air traps to maintain ideal temperatures, which are below annual mean ambient temperature (Tuttle and Kennedy 2002). Chimney-effect airflow occurs when two cave or mine entrances are at different elevations and outside temperatures fall below the annual mean (which is approximated by the cave walls); air flows out of the upper entrance and in the lower entrance (Tuttle and Taylor 1998). This airflow creates the potential for smoke intrusion into hibernacula

(Carter et al. 2002) perhaps especially during cold fall, winter, and spring nights. Nighttime inversions during this period may be of particular concern and inversion climatologies, where seasonal climate and landscape characteristics are used to determine the potential for smoke accumulation (Ferguson et al. 2003), may be a good place to start for evaluating risk to individual hibernacula.

Fire and Bat Habitat

Fire may have short-term effects on bats through heat and gas exposures, but fire also affects bat habitat. Habitat effects have been assumed to include increased roost availability, facilitation of foraging from reduced clutter, and increased insect prey productivity (USDA FS 2003, USFWS 2007a). Roost availability is dependent on both the quality of individual roosts and the population dynamics of roosts. For Indiana bats, the epitome of a high-quality primary maternity roost appears to be a large dead tree, exposed to solar radiation, with large plates of sloughing bark (Kurta et al. 1993, Foster and Kurta 1999, Carter and Feldhamer 2005, Lacki et al. 2008). Alternate roosts, including live trees and other snags, are also important even though they shelter relatively few bats and seemed to be used mostly during periods of warm ambient temperatures and high precipitation (Humphrey et al. 1977, Gardner et al. 1991, Kurta et al. 1993, Miller et al. 2002). Maternity colonies may occupy one or more primary roosts. Males use smaller trees than females, on average (Kurta 2005).

Published data on summer-roosting behavior of Indiana bats, including nonreproductive males, suggest that bats select roost trees based on size (e.g., diameter) and that stands possessing trees exceeding 40 cm diameter at breast height (d.b.h.) are more likely to provide adequate maternity habitat for this species (Fig. 1). Our literature survey results are based on the roost trees that were available to bats on the landscapes they use. Given the documented preference for larger- than-average roost trees (Kurta 2005), it is likely that even larger roost trees would have been used if they had been available to bats in the published studies. Along these lines, Callahan et al. (1997) recommended promoting the development of forested stands possessing large-diameter, mature trees to provide adequate maternity habitat for the Indiana bat. Further, Carter and Feldhamer (2005) suggested that snag creation within stands of mature timber may be necessary for sustaining maternity habitat of this species in perpetuity. An implication of our results showing the importance of tree size is that, when calculating potential densities of snags and live roosts in forested stands to evaluate quality of maternity habitat for Indiana bats, managers should place the greatest weight on potential roost trees ≥ 40 cm in diameter and ≥ 10 m in height. Further, because Indiana bats prefer roosting sites beneath exfoliating bark to cavities or crevices (Kurta et al. 1993, Carter and Feldhamer 2005, Lacki et al. 2008), the presence of this habitat characteristic on snags should also be used as a criterion for counting a snag as potentially suitable.

Boyles and Aubrey (2006) found that reintroduction of prescribed fire into mixed-oak forests after decades of fire suppression resulted in a striking increase in evening bat (*Nycticeius humeralis*) roosting compared with adjacent unburned forest. They attributed the effect on this cavity-roosting bat to the high tree mortality after the first burn, which increased both the exposure of existing snags and the density of snags. MacGregor et al. (1999) found that male Indiana bats did not consistently use prescribed burned units for roosting in greater than expected proportion to their area on the landscape, though it is not clear how fires affected snag availability. Two-age shelterwoods in the same landscape in which snag and live roost retention guidelines were in place showed greater-than-expected roosting by male Indiana bats, lending support for future studies in which Indiana bat responses to shelterwood-burn treatments are analyzed.

Populations of large snags suitable for roosting are the result of a balance between canopy tree mortality rates and how long snags retain their preferred characteristics. In eastern mixed-oak forests, tree mortality rates range from 1-3 percent/year, but much of that mortality is of smaller, suppressed stems (Parker et al. 1985, Wyckoff and Clark 2002). Injury from low-intensity surface fires after long periods of fire suppression in southeastern Ohio appeared to add incrementally to the mortality of large canopy trees that were at risk before burning (Yaussy et al. 2004). Reintroduction of fire to long unburned stands can cause high rates of overstory mortality (Boyles and Aubrey 2006, Anderson and Brown 1983) as can high-intensity fires (Regelbrugge and Smith 1994, Moser et al. 1996). Although bat habitat may be improved over the short term because of these often patchy mortality events, the events tend to occur on sites that support intense fire behavior (e.g., slopes and topographically dry and exposed sites) and a return of high quality roosting habitat to those landscape positions after snags have become unsuitable for roosting would take a very long time. Avoiding high tree mortality adjacent to hibernacula would also be prudent, given ensuing microclimatic changes that may not be favorable to hibernating bats (Carter et al. 2002).

High mortality rates of large, old trees has been recognized where fire has been reintroduced into long-unburned stands of longleaf pine (*Pinus palustris* Mill., ponderosa pine (*Pinus ponderosa*), and larch (*Larix laricina*); extensive duff accumulation results in root consumption and basal heating (e.g., Varner et al. 2005, Kolb et al. 2007). Large, old trees may be inherently more susceptible to dying from a given level of injury because of their physiological characteristics (see Kolb et al. 2007). It is not clear that fire in eastern mixed-oak forests would ever lead to preferential mortality of large trees, as opposed to small trees, if only because of the lack of extensive basal duff accumulation. Efforts to use fuel management (e.g., planned felling or redistribution of tops during shelterwood operations) and ignition strategies to kill individual or small groups of potential roosts would merit attention.

Apart from the creation of snags, the longevity of suitable snags is also an important determinant of snag availability. Studies that follow snags from year to year have found that roosts are used from 2-6 years (Kurta 2005). Of recently dead trees, only a portion will develop patches of exfoliating bark suitable for roosting, though it is not known what fraction that is and what determines the propensity for bark exfoliation. Fires not only create snags by killing trees, some of which may be live roosts, but also fell snags through the structural weakening caused by smoldering combustion (Carter et al. 2002). Experience suggests that snags are drier, and smolder more readily, during late spring burns (Michael Bowden, Ohio Division of Forestry, personal communication). Snag loss may be more of a problem in late summer and fall burns after dry periods when duff is dry and consumes more readily (K. Moore and E.J. Bunzendahl Wayne National Forest (KM), Daniel Boone WF (EJB), personal communication). Bats using maternity roosts in riparian habitat, and the roosts themselves, may be least vulnerable to fire because fire intensities in these landscape positions tend to be low (Carter et al. 2002).

Snag species composition across the range of the Indiana bat suggests that a range of tree species (though not all) form suitable snags and local availability largely determines which tree species bats use (Kurta 2005, USFWS 2007a). Fire-maintained oak-hickory forests would be expected to maintain a species composition suitable for Indiana bats (USFWS 2007a). Tree species that consistently form high quality live roosts include shellbark hickory (*Carya laciniosa*), shagbark hickory (*C. ovata*), and white oak (*Quercus alba*). Oaks, particularly of the white oak group, are favored by low-intensity fire (Abrams 2005) while oak and hickory regeneration has been shown to be favored by repeated fires below open canopies (Iverson et al. 2008). Live roost trees are less ephemeral than snags and provide secondary roosts for maternity colonies and roosts for nonreproductive bats, yet most trees occupied by Indiana bats during the summer are snags (Kurta 2005).

A first step to setting targets for potentially suitable roosting habitat would be to determine adequate roost densities and spatial distribution. Unfortunately, there is a paucity of data available on these topics (Lacki et al. 2008). However, studies of Indiana bat primary and secondary roosting behavior have shown that bats show high fidelity to individual roosts. Even though they switch roosts roughly 2 days, on average, Indiana bats often return to previously used roosts, supporting efforts to protect known roost trees until they become unsuitable. Indiana bats also show fidelity within and among years to roosting areas 1 or more km in extent (Gumbert et al. 2002, Kurta et al. 2002). Because bats in a single maternity colony are dispersed among various roosts at any given time (Kurta 2005) and because of across-year fidelity to roosting areas (also see Humphrey et al. 1977, Kurta and Murray 2002), the supply of primary and secondary roosts must be maintained over areas of tens of square kilometers, an area larger than single burn or harvest units.

Fire and Foraging

Fire may affect foraging habitat in at least two ways: through effects on forest structure and through effects on insect prey productivity and community structure. Foraging habitat, and the effects of fire, may be particularly important for maternity and staging areas, where bats have high demands for insect prey (Carter et al. 2002). Indiana bats are aerial feeders, whose short, broad wings, rounded wingtips, and echolocation characteristics are suitable for foraging in forests (Norberg and Rayner 1987). Indiana bats have been observed to feed primarily around tree crowns, not within them, occasionally descending into the midstory and shrub layers (Humphrey et al. 1977). Lee and McCracken (2004) found that, in sympatry with other *Myotis* bats, Indiana bats foraged higher above ground. For these reason, it has been hypothesized that Indiana bats would prefer foraging in more open stands (e.g., USFWS 2005). Historical forest and uneven-age timber management prescriptions involving low-intensity fire that are being implemented on National Forests in the mixed-oak region would reduce understory and mid-story clutter (Arthur et al. 1998, Elliott et al. 1999, Blake and Schuette 2000, Hutchinson et al. 2005) and, if repeated over the long term, would reduce density of the upper canopy (e.g., Huddle and Pallardy 1996, Peterson and Reich 2001).

Data on Indiana bat foraging casts some doubt on the potential benefits of stand thinning by fire, though Indiana bat foraging in burned forest has not been studied directly. In Appalachian landscapes of West Virginia, Indiana bats were found to use forested riparian habitats most heavily and, where recorded in uplands impacted by a variety of forestry practices, were detected most frequently in areas with the highest canopy cover (e.g., Owen et al. 2004, Ford et al. 2005). Using sonic detectors, Titchenell (2007) found that *Myotis* bats (no identification to species, yet unlikely to have included Indiana bats) showed no difference in foraging behavior between control and shelterwood stands, yet other bat taxa foraged more intensively in shelterwoods. Loeb and Waldrop (2008) found overall preference for thinned pine stands among non-*Myotis* bats, and, again, response varied among species. Complicating matters, interspecific interactions have been shown to affect when and where Indiana bats forage (e.g., Lee and McCracken 2004).

Knowledge of the diet of Indiana bats could help provide a target for monitoring and management. From a limited number of studies, it appears that the diet of Indiana bats foraging primarily in upland forests is dominated by moths (Lepidoptera) and beetles (Coleoptera, see Brack and LaVal 1985, Murray and Kurta 2002). Lepidoptera also dominated diets of bats foraging in riparian habitats (e.g., Belwood 1979, Lee and McCracken 2004), while bats foraging over wetlands in Michigan consumed primarily the adult stages of aquatic insects (Kurta and Whitaker 1998). Diet studies which concurrently sampled both bat diets and nocturnal insect abundances indicated a preference for Lepidoptera by Indiana bats as opposed to a diet determined solely by random encounter rates (e.g., Brack and LaVal 1985, Lee and McCracken 2004).

Unfortunately, there is a paucity of information on insect prey availability for bats in central hardwood forests and its relationship with forest management activities, including fire (Rieske-Kinney 2006). A study of pollinators in oak-dominated stands in the southern Appalachians found that a combination of mechanical shrub control and fire resulted in greater abundances of beetles and butterflies (Lepidoptera) compared with burning or mechanical treatments alone or no treatment, and that these increases were related to increased herbaceous cover where canopy cover was most reduced (Campbell et al. 2007). Whether these increases in abundance would translate into greater (nocturnal) prey availability for bats is not known. Fires in mixed-oak forests have a negative effect on litter-dwelling mites and collembolans and there have been mixed results in using fire to control gypsy moths and acorn-predating weevils (see Rieske-Kinney 2006). Low- to moderate-intensity fire had no effect on palatability of two overstory tree species to gypsy moth larvae (Rieske-Kinney et al. 2002).

Wildlife managers express concern that prescribed fires in the late dormant season and spring reduce bat prey abundances during the critical period when bats

Table 4.—National Forests that consider Indiana bats in their forest plans. Presence of hibernacula in a National Forest, or within its proclamation boundary, and bat summer status were determined from the latest USFS monitoring report, USFS Programmatic Biological Analysis, or USFWS Programmatic Biological Opinion. Number of hibernacula is given in parentheses.

National Forest	State	Forest plan revision	Standards and guidelines	Hibernacula	Bat summer status
Allegheny	Pennsylvania	2007	Part 3	No	Two males captured
Cherokee	Tennessee	2004	Chapter 2	No	Three post-lactating females documented in 2006 (apparently upland roosts)
Daniel Boone	Kentucky	2004	Chapter 2	Yes (>15 w/in Forest)	Seven maternity colonies documented, 90 total records
George Washington and Jefferson	Virginia	2007 (draft)	Chapter 3	Yes (4)	No known maternity colonies, no mistnetting conducted
Hoosier	Indiana	2006	Chapter 3	Yes (1)	Two maternity colonies documented, first in 2004 (apparently upland)
Huron-Manistee	Michigan	2006	Chapter 2	Yes (1)	Two males captured during swarming
Mark Twain	Missouri	2005	Chapter 2	Yes (4)	Two known maternity colonies, male roosts identified
Monongahela	West Virginia	2006	Chapter 2	Yes (15)	One known maternity colony
Nantahala-Pisgah	North Carolina	1994	Amendment 10 (released in 2000)	No	One known maternity colony
Ozark	Arkansas	2005	Part 3	Yes (8)	>1 maternity colony, females foraging in riparian and upland habitat, various male bat captures
Shawnee	Illinois	2006	Appendix H	Yes (2)	Two known maternity colonies in bottomland hardwoods, male roosts in one cave and three mines
Wayne	Ohio	2006	Chapter 2	Yes (2)	Reproductive females captured, maternity colony/colonies assumed present

are coming out of hibernation, are migrating, and females are pregnant. Unanswered questions include the magnitude of reduction in prey abundances, how long those abundances remain depressed, and whether fire ultimately increases foraging success (through both prey availability and improved forest structure for Indiana bat foraging). Another open question is whether increased water yields from fire-maintained watersheds in the central hardwoods would translate into greater insect productivity in riparian areas with benefits to bats foraging on emerging adults (e.g., Beck et al. 2005).

In light of both the potential negative effects of fires on bats over the short term and the potential longer-term benefits, it is useful to consider the size of burn units in relation to the size of bat home ranges. There is a large range in reported mean home ranges, in one review spanning 81 to 668 ha (USDA FS 2007). Indiana bats may travel long distances to foraging areas, so roosting areas may be separated from foraging areas. Burn unit sizes are on par with mean home ranges and, if burning is done near known maternity colonies, it would be prudent to locate burn unit boundaries so that entire home ranges are not burned over in a single year or to conduct the burns in a way that creates a patchwork of burned and unburned areas (e.g., using ridge ignition, which leaves mesic areas unburned).

Fire and Bats on National Forests

We reviewed the current forest plans for the National Forests that lie within the distributional limit of the Indiana bat for information pertaining to standards and guidelines relative to the Indiana bat (Table 4). Forest Plan standards are attainments that must be reached or courses of action that must be followed to mitigate the effects of land management activities while guidelines are expected to be followed in most circumstances. Forest plans are required by the National Forest Management Act and are revised periodically and amended as needed. Individual projects must go through National Environmental Policy Act analysis, including biological assessments for project effects on Indiana bats, where present. Burn plans further specify how burns will be conducted to minimize risk to bats, where applicable.

Since 1994, nearly all National Forests within the range of the Indiana bat have requested formal consultation with the USFWS relative to their forest plans, resulting in the issuance of non-jeopardy biological opinions and associated incidental take statements (Krusac and Mighton 2002). Formal consultation is a negotiation between the land management agency and the USFWS that is intended to result in a balance between the need to conduct land management activities and the need to minimize, but not eliminate, "take" of Indiana bats (e.g., mortality and disturbance). Thus, forest plan standards and guidelines are not as restrictive as they are when the objective is to eliminate all "take." Many forest plans have been revised recently and often include extensive consideration of Indiana bats (Table 4). Because forest plans are reflective of local conditions and the particular interaction between Forest and USFWS field-office staff, generalization is difficult.

Private and other public landowners can also enter into negotiations with the USFWS to develop Habitat Conservation Plans (HCPs) that would allow incidental take statements to be issued. The state of Indiana's Division of Forestry is currently developing an HCP for Indiana bats and is expected to be the first non-USFS land management entity to receive an incidental take statement.

A sense of bat abundance on Forests can be obtained from monitoring and evaluation reports and biological opinions and assessments (Table 4). Monitoring data are not collected or reported in a standardized fashion across the range and information is often qualitative, so comparisons are difficult. Several Forests have documented only the presence of male bats and others have identified only a few maternity colonies. Maternity colonies occur in both uplands and bottomland/riparian habitat on National Forests. It is apparent from monitoring and evaluation reports that there has been little attempt to follow sampling designs that would allow a Forest to determine whether standards and guidelines were leading to improved Indiana bat performance. Regardless, the relatively few bats documented in many forests would make it difficult to detect effects of habitat modification.

Forest plans in and of themselves do not preclude growing-season burning outside of occupied roosting habitat on most of the National Forests within the range of the Indiana bat. Growing-season burning is well established on the Ozark National Forest in Arkansas. Most Forest Plans prohibit burning in the few known maternity areas (areas that include roosts and associated foraging habitat) during the roosting period. The roosting period, when defined, spans the period from May 1 (earliest date is April 15) to late summer or fall (the earliest end-date is July 31; the latest is Nov. 15). Some Forests require surveys to determine whether bats are present before burning can be done in potential roosting habitat during the roosting period. The most restrictive prescription appears to be that in place for the Shawnee National Forest, where no upland burning is allowed from May 1 to September 1. Even the May 1 restriction allows burning during late spring bud burst. The Monongahela National Forest (West Virginia) appears to be restricted to 120 ha of burning per year, an inadequate amount of acreage if fire is to play a role in oak forest management (see incidental take statement, USDA FS 2006). Given current restrictions, growing-season burning past the bud-burst stage would appear to be possible on a handful of forests, at least on an experimental basis.

Fire practitioners have expressed concern that date restrictions on live tree and snag removal in recent biological opinions (e.g., Whitebreast Creek and Fort Drum Connector Projects (USFWS 2009b) have been set back to March 31 from mid-April and that burn restrictions may also be tightened to avoid incidental "take" of Indiana bats. Concerns about date restrictions highlight the need for focused research on both the short-term and long-term effects of fires on bats and their habitats so that the negative effects of growing-season burning can be balanced against any positive effects of such burning on habitat.

Forest plans establish standards and guidelines for preventing smoke intrusion into hibernacula. The Wayne National Forest Plan prohibits prescribed burning within a defined hibernacula zone while the George Washington and Jefferson National Forests prohibit burning in a hibernacula zone unless it could be assured that no impact would occur. The Hoosier, Huron-Manistee, and Mark Twain National Forests prohibit burning during swarming and staging periods. The Hoosier, Mark Twain, Ozark-St. Francis, Shawnee, and Wayne National Forests call for best smoke management practices to be used relative to hibernacula. No mention of prescribed burning and hibernacula is made in the Monongahela and Daniel Boone National Forest plans. In these Forests, consideration of smoke management around hibernacula is left to project planning and burn plans. Forests with no known hibernacula (Allegheny, Cherokee, and Nantahala-Pisgah National Forests), of course, make no mention of hibernacula in reference to burning.

Standards and guidelines governing snags and potential live roosts differ among Forests and among silvicultural treatments within Forests. Live roost retention (e.g., shagbark and shellbark hickory and other trees with suitable bark characteristics) varies from retention of all shagbark and shellbark hickories (e.g., Allegheny and George Washington and Jefferson National Forests) to their retention in the context of the availability of other live roosts and the silvicultural system being employed. The regeneration system of most interest relative to burning in mixed-oak forests is some form of shelterwood (e.g., Brose et al. 1998, 1999), though there is some potential for using fire in clearcuts to benefit oaks (Michael Bowden, personal communication). Shelterwood standards and guidelines, where provided, include provisions for future snags and provide targets for snag densities by size class. Often the lower size limit on snags is between 10 and 25 cm d.b.h., a size that is considerably smaller than the median Indiana bat roost (Fig. 1).

Apart from forest plans, forestwide (programmatic) biological assessments (e.g., USDA FS 2005) and their associated biological opinions (e.g., USFWS 2005) are available for Forests and are good places for state and private land managers to find information on the Indiana bat and land management as they develop their own programs. The Indiana Bat Recovery Plan (USFWS 2007a) has rangewide scope and is also a good source of information. The USFWS website dedicated to the Indiana bat contains a variety of documents and other useful information (USFWS 2009a).

CONCLUSIONS AND RECOMMENDATIONS

Prescribed fires cause roost-switching behavior in tree-roosting bats that would reduce their exposure to smoke. Extensive use of torpor by roosting males and nonreproductive females would increase their risk of smoke exposure, though use of torpor and arousal times under typical burning conditions are unknown. Reproductive females are generally expected to maintain high body temperatures and, thus, be able to respond quickly to fires. However, use of torpor by pregnant female bats during spring storms has been demonstrated. Forest managers can reduce risk to tree-roosting bats by reducing fuel consumption, which determines the quantity of smoke produced, and fireline intensity, which drives smoke transport. Burning under relatively high ambient temperatures, for example, from late April through May, after bats have dispersed to their summer habitat, may also reduce risk, though data on the use of torpor are needed. Given demonstrated roost switching behavior, the critical risk period for bats may not be when maternity colonies are formed, but later, when flightless young are present. Further research may show that early growing-season burning, as opposed to burning during the vegetative dormant season, may be done at lower intensities during warmer weather and may result in more desirable fire effects on vegetation with manageable risks for bats.

Smoke exposures in hibernacula would be expected to pose problems if they cause extra arousals, though arousal thresholds for hibernating bats are unknown. Given treatment of hibernacula as smoke-sensitive targets, and the smoke management efforts that this designation entails, it is unlikely that toxicity from high gas concentrations themselves will be problematic, only arousal from exposures to relatively low smoke concentrations. Smoke intrusion into hibernacula of species other than Indiana bats has been documented. For Indiana bat hibernacula, smoke intrusion is most likely when hibernacula are exchanging air under conditions when nighttime temperatures drop below the annual mean. A better understanding of risk requires increased knowledge of arousal response to smoke by hibernating Indiana bats and information on air flow in individual hibernacula.

Fires in upland mixed-oak forests are expected to improve roosting and foraging habitat for Indiana bats by increasing the availability of suitable snags, reducing canopy clutter, and increasing long-term insect prey availability. Unfortunately, the data basis for these expectations is poor to mixed. For instance, the long-term tradeoff between snag creation and snag loss in mixed-oak forests under burning regimes is unknown. Data on foraging activity show Indiana bat preference for relatively closed-canopy stands, casting doubt on the benefits to foraging that would arise from the stand thinning caused by forest burning. Explicit studies on the benefits of forest burning on Indiana bat foraging habitat are needed.

Fire has been recommended on National Forests in two relatively incompatible contexts: as a tool for well regulated oak silviculture and as an imperative for oak ecosystem restoration and maintenance. Given Indiana bat fidelity to roosting and foraging areas, upland maternity areas might serve, on an experimental basis at first, as focal areas for oak ecosystem restoration and maintenance where burning is used to try to increase local roosting populations and their reproductive success. A regional approach to oak ecosystem conservation and monitoring on National Forests has been proposed (Yaussy et al. 2008) and, if designed properly, has the potential to add to our understanding of Indiana bat response to forest thinning and burning. Currently, adaptive management relative to Indiana bats is not possible because monitoring programs are poorly funded and not designed to assess the effects of land management activities on bat populations. Furthermore, few research projects have addressed the central questions about fire and Indiana bat habitat. The current state of knowledge is a poor foundation on which to base upland management activities and to determine whether existing, or any further, date restrictions on burning on National Forests are counterproductive for Indiana bat conservation.

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MODELING TOOLS TO SUPPORT FIRE MANAGEMENT AND PLANNING

PERFORMANCE OF FIRE BEHAVIOR FUEL MODELS DEVELOPED FOR THE ROTHERMEL SURFACE FIRE SPREAD MODEL

Robert Ziel[1] and W. Matt Jolly[2]

[1]Fire Management Specialist, Michigan Department of Natural Resources, Marquette, MI
[2]Ecologist, U.S. Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory, Missoula, MT
RZ is corresponding author; to contact, call (906) 249-1497 or email at zielr@michigan.gov

Abstract.—In 2005, 40 new fire behavior fuel models were published for use with the Rothermel Surface Fire Spread Model. These new models are intended to augment the original 13 developed in 1972 and 1976. As a compiled set of quantitative fuel descriptions that serve as input to the Rothermel model, the selected fire behavior fuel model has always been critical to the resulting modeled fire behavior. Fuel characteristics affect both the heat source and the heat sink factors in the spread model. While the original 13 models emphasized peak fire-season fuel combinations, the new set establishes a greater role for live fuels. Use of live fuel as a variable produces a broad range of modeled fire behavior related to seasonal vegetative development, especially for those fuel models that include herbaceous fuel loads. Intending to represent "greenup" and late season "curing," the new fuel models allow the user to transfer herbaceous fuels from live to dead. As fuel load transferred increases, the influence of moisture of extinction and wind limit produce dramatic changes in modeled spread and intensity. The new models present an important opportunity to model fire behavior under a wider range of fuel conditions, including fire use. However, the user needs to be aware of the live fuel components in the selected models and manage those inputs carefully.

INTRODUCTION

Wildland fire behavior is frequently described by the spread rates and burning intensity as it burns along the surface. The primary factors that influence this fire behavior are grouped into three categories: weather, topography, and fuels. The Rothermel (1972) surface fire spread model uses fuel moisture and wind to represent weather's influence, slope, and elevation to characterize the topographic factor, and a collection of characteristic inputs called fire behavior fuel models as variables describing fuel complex characteristics. In support of firefighter safety and risk management, the original 13 fuel models created by Rothermel (1972) and Albini (1976) were created to predict fire behavior under peak burning conditions. In these "worst-case scenarios," fires predicted are considered to be influenced primarily by dead fuels and their characteristics. Though four of these 13 models include live fuels, the condition of those fuels is generally of secondary importance overall. With the increasing interest in fire use under more moderate conditions during the growing season and the imperative placed on designing fuel management practices to protect values in the wildland/urban interface, accurate estimates of fire behavior with increased live fuel influence and modified fuelbeds require some additional fuel model choices. The comments in this paper are intended to provide the reader with insights into these 40 new fuel models, the opportunities they provide, and the effects of some underlying model relationships that they bring out.

PURPOSE OF 40 ADDITIONAL MODELS

Many of the more moderate conditions associated with fire use involve seasons and situations where live fuels can serve to reduce fire spread and intensity to varying degrees. To reflect these conditions, 27 of the 40 new fuel models developed by Scott and Burgan (2005) include live fuel loads, which result in a greater range of potential fire behavior for any given set of weather conditions.

Throughout the United States, most wildland fuelbeds need to be fairly dry to support active fire spread. However, in areas like the southeastern U.S., some fuels will ignite and continue to burn under much higher moisture regimes. Whereas the original 13 included only two fuel models that emphasized this characteristic, the new set of 40 include 14 models designated as humid-climate fuels with significantly higher moisture of extinction (ME) parameters. ME is the dead fuel moisture above which fires cannot actively spread.

In addition to creating two new categories of carrier fuels (grass-shrub and timber understory), the new set of fuel models includes from four to nine model choices within each carrier fuel category. The increased number of choices allows for more accurate reflection of the range of potential fire behavior among similar fuelbeds. In forested situations, models of crown fire potential and canopy scorch/mortality use surface fire intensity as a primary factor. Without accurate predictions of surface fire behavior, users will be unable to accurately evaluate the effectiveness of fuel reduction projects that are intended to reduce the probability and extent of damage to the forest canopy.

With these additional choices, users are more likely to find an appropriate fuel model within the carrier category that can be related to the vegetative cover on their site. More consistent and predictable relationships between cover and fuels will allow for more effective use of remote sensing techniques and automated updating of fuel classifications. The full potential of landscape assessment tools such as FARSITE, FLAMMAP, and FSPro will not be realized until classifications of fuels are accurate across entire landscapes.

FUEL MODEL CHARACTERISTICS

Table 1 lists all 53 fuel models and the characteristic model parameters for each. It can be used to make comparisons between familiar models among the original 13 and alternatives that may be considered among the 40 new models. Intrinsic characteristics of the fuel particles include the dead and live heat content and the dead fuel ME. Extrinsic characteristics are based on the size, shape, quantity, and arrangement of those fuel particles. They include the bed depth as well as the fuel loading and surface area to volume ratio (SAV) for each fuel size class (1-hr, 10-hr, 100-hr) and type (herbaceous and woody).

One new parameter has been added to designate fuel models that are dynamic; i.e., with herbaceous fuel loads that can be transferred between live and dead categories. Fuel models without herbaceous fuels are considered static. Figure 1 demonstrates the increased effect of the fuel load transfer between live and dead on resulting spread and intensity. While the original fuel model FB2 (timber grass and understory) shows only a limited influence of herbaceous fuel moisture, that influence in the new models (GR2, GR4, GR7, GR9) can be dramatic.

Each model has a characteristic ME. The 14 new models designated for humid climates have a ME of 30 percent or higher, allowing fuels to burn under much higher moisture content. Figure 2 shows an example of the difference between two fuel models with otherwise similar characteristics; GR4 has a ME of 15 percent, as compared to 40 percent for GR5. At 15 percent 1-hr fuel moisture, GR4 exhibits no spread while GR5 shows spread rates of 40 chains/hour and flame lengths between 8 and 9 feet.

INTERACTIONS WITH THE ROTHERMEL SURFACE FIRE SPREAD MODEL

Figure 3 depicts the Rothermel (1972) surface fire spread model. In the equation, the numerator, identified as the "Heat Source," represents the heat available to ignite adjacent fuels, effectively promoting fire spread. The denominator, labeled the heat sink, estimates the heat needed to remove moisture and raise fuel particles to the ignition temperature. The calculations are fairly straightforward for wildland fuelbeds that contain predominantly dead fuels.

Fire modeling needs to recognize that the moisture content of live fuels is higher and varies over a wider range than that of dead fuels. Live fuels can burn at a higher moisture content but also can serve to impede spread. Rothermel's model accounts for these differences by estimating the reaction intensity and heat of pre-ignition separately for live and dead fuels, then combining them.

Table 1.—Fuel model parameters for use with the Rothermel surface fire spread model; dynamic models are shaded

Carrier	FM #	FM Code	Fuel Model Name	1hr Load	10hr Load	100hr Load	Herb Load	Woody Load	Total Load	Dynamic	1hr SAV	Herb SAV	Woody SAV	Bed Depth	Moist Extinct	Dead Heat	Live Heat
GR	1	FB1	Short grass	0.7	0.0	0.0	0.0	0.0	0.7	static	3500	9999	9999	1.0	12	8000	8000
GR	2	FB2	Timber grass and understory	2.0	1.0	0.5	0.5	0.0	4.0	static	3000	1500	9999	1.0	15	8000	8000
GR	3	FB3	tall grass	3.0	0.0	0.0	0.0	0.0	3.0	static	1500	9999	9999	2.5	25	8000	8000
GR	101	GR1	Short, sparse dry climate grass	0.1	0.0	0.0	0.3	0.0	0.4	dynamic	2200	2000	9999	0.4	15	8000	8000
GR	102	GR2	Low load dry climate grass	0.1	0.0	0.0	1.0	0.0	1.1	dynamic	2000	1800	9999	1.0	15	8000	8000
GR	103	GR3	Low load very coarse humid climate grass	0.1	0.4	0.0	1.5	0.0	2.0	dynamic	1500	1300	9999	2.0	30	8000	8000
GR	104	GR4	Moderate load dry climate grass	0.3	0.0	0.0	1.9	0.0	2.2	dynamic	2000	1800	9999	2.0	15	8000	8000
GR	105	GR5	low load humid climate grass	0.4	0.0	0.0	2.5	0.0	2.9	dynamic	1800	1600	9999	1.5	40	8000	8000
GR	106	GR6	moderate load humid climate grass	0.1	0.0	0.0	3.4	0.0	3.5	dynamic	2200	2000	9999	1.5	40	9000	9000
GR	107	GR7	High load dry climate grass	1.0	0.0	0.0	5.4	0.0	6.4	dynamic	2000	1800	9999	3.0	15	8000	8000
GR	108	GR8	High load very coarse humid climate grass	0.5	1.0	0.0	7.3	0.0	8.8	dynamic	1500	1300	9999	4.0	30	8000	8000
GR	109	GR9	very high load humid climate grass	1.0	1.0	0.0	9.0	0.0	11.0	dynamic	1800	1600	9999	5.0	40	8000	8000
GS	121	GS1	low load dry climate grass-shrub	0.2	0.0	0.0	0.5	0.7	1.4	dynamic	2000	1800	1800	0.9	15	8000	8000
GS	122	GS2	moderate load dry climate grass-shrub	0.5	0.5	0.0	0.6	1.0	2.6	dynamic	2000	1800	1800	1.5	15	8000	8000
GS	123	GS3	moderate load humid climate grass-shrub	0.3	0.3	0.0	1.5	1.3	3.3	dynamic	1800	1600	1600	1.8	40	8000	8000
GS	124	GS4	high load humid climate grass-shrub	1.9	0.3	0.1	3.4	7.1	12.8	dynamic	1800	1600	1600	2.1	40	8000	8000
SH	4	FB4	chaparral	5.0	4.0	2.0	0.0	5.0	16.0	static	2000	9999	1500	6.0	20	8000	8000
SH	5	FB5	brush	1.0	0.5	0.0	0.0	2.0	3.5	static	2000	9999	1500	2.0	20	8000	8000
SH	6	FB6	dormant brush	1.5	2.5	2.0	0.0	0.0	6.0	static	1750	9999	9999	2.5	25	8000	8000
SH	7	FB7	southern rough	1.1	1.9	1.5	0.0	0.4	4.9	static	1750	9999	1500	2.5	40	8000	8000
SH	141	SH1	low load dry climate shrub	0.3	0.3	0.0	0.2	1.3	2.0	dynamic	2000	1800	1600	1.0	15	8000	8000
SH	142	SH2	moderate load dry climate shrub	1.4	2.4	0.8	0.0	3.9	8.4	static	2000	9999	1600	1.0	15	8000	8000
SH	143	SH3	moderate load humid climate shrub	0.5	3.0	0.0	0.0	6.2	9.7	static	1600	9999	1400	2.4	40	8000	8000
SH	144	SH4	low load humid climate timber-shrub	0.9	1.2	0.2	0.0	2.6	4.8	static	2000	1800	1600	3.0	30	8000	8000
SH	145	SH5	high load dry climate shrub	3.6	2.1	0.0	0.0	2.9	8.6	static	750	9999	1600	6.0	15	8000	8000
SH	146	SH6	low load humid climate shrub	2.9	1.5	0.0	0.0	1.4	5.8	static	750	9999	1600	2.0	30	8000	8000
SH	147	SH7	very high load dry climate shrub	3.5	5.3	2.2	0.0	3.4	14.4	static	750	9999	1600	6.0	15	8000	8000
SH	148	SH8	high load humid climate shrub	2.1	3.4	0.9	0.0	4.4	10.7	static	750	9999	1600	3.0	40	8000	8000
SH	149	SH9	very high load humid climate shrub	4.5	2.5	0.0	1.6	7.0	15.5	dynamic	750	1800	1500	4.4	40	8000	8000
TU	10	FB10	timber litter and understory	3.0	2.0	5.0	0.0	2.0	12.0	static	2000	9999	1500	1.0	25	8000	8000
TU	161	TU1	light load dry climate timber-grass-shrub	0.2	0.9	1.5	0.2	0.9	3.7	dynamic	2000	1800	1600	0.6	20	8000	8000
TU	162	TU2	moderate load humid climate timber-shrub	1.0	1.8	1.3	0.0	0.2	4.2	static	2000	9999	1600	1.0	30	8000	8000
TU	163	TU3	moderate load humid climate timber-grass-shrub	1.1	0.2	0.3	0.7	1.1	3.3	dynamic	1800	1600	1400	1.3	30	8000	8000
TU	164	TU4	dwarf conifer with understory	4.5	0.0	0.0	0.0	2.0	6.5	static	2300	9999	2000	0.5	12	8000	8000
TU	165	TU5	very high load dry climate timber-shrub	4.0	4.0	3.0	0.0	3.0	14.0	static	1500	9999	750	1.0	25	8000	8000
TL	8	FB8	compact timber litter	1.5	1.0	2.5	0.0	0.0	5.0	static	2000	9999	9999	0.2	30	8000	8000
TL	9	FB9	hardwood litter	2.9	0.4	0.2	0.0	0.0	3.5	static	2500	9999	9999	0.2	25	8000	8000
TL	181	TL1	Low load compact conifer litter	1.0	2.2	3.6	0.0	0.0	6.8	static	2000	9999	9999	0.2	30	8000	8000
TL	182	TL2	low load broadleaf litter	1.4	2.3	2.2	0.0	0.0	5.9	static	2000	9999	9999	0.2	25	8000	8000
TL	183	TL3	moderate load conifer litter	0.5	2.2	2.8	0.0	0.0	5.5	static	2000	9999	9999	0.3	20	8000	8000
TL	184	TL4	Small downed logs	0.5	1.5	4.2	0.0	0.0	6.2	static	2000	9999	9999	0.4	25	8000	8000
TL	185	TL5	high load conifer litter	1.2	2.5	4.4	0.0	0.0	8.1	static	2000	9999	1600	0.6	25	8000	8000
TL	186	TL6	moderate load broadleaf litter	2.4	1.2	1.2	0.0	0.0	4.8	static	2000	9999	9999	0.3	25	8000	8000
TL	187	TL7	Large downed logs	0.3	1.4	8.1	0.0	0.0	9.8	static	2000	9999	9999	0.4	25	8000	8000
TL	188	TL8	long-needle litter	5.8	1.4	1.1	0.0	0.0	8.3	static	1800	9999	9999	0.3	35	8000	8000
TL	189	TL9	very high load broadleaf litter	6.7	3.3	4.2	0.0	0.0	14.1	static	1800	9999	1600	0.6	35	8000	8000
SB	11	FB11	light slash	1.5	4.5	5.5	0.0	0.0	11.5	static	1500	9999	9999	1.0	15	8000	8000
SB	12	FB12	medium slash	4.0	14.0	16.5	0.0	0.0	34.6	static	1500	9999	9999	2.3	20	8000	8000
SB	13	FB13	heavy slash	7.0	23.0	28.1	0.0	0.0	58.1	static	1500	9999	9999	3.0	25	8000	8000
SB	201	SB1	low load activity fuel	1.5	3.0	11.0	0.0	0.0	15.5	static	2000	9999	9999	1.0	25	8000	8000
SB	202	SB2	moderate load activity or low load blowdown	4.5	4.3	4.0	0.0	0.0	12.8	static	2000	9999	9999	1.0	25	8000	8000
SB	203	SB3	high load activity fuel or moderate load blowdown	5.5	2.8	3.0	0.0	0.0	11.3	static	2000	9999	9999	1.2	25	8000	8000
SB	204	SB4	high load blowdown	5.3	3.5	5.3	0.0	0.0	14.0	static	2000	9999	9999	2.7	25	8000	8000

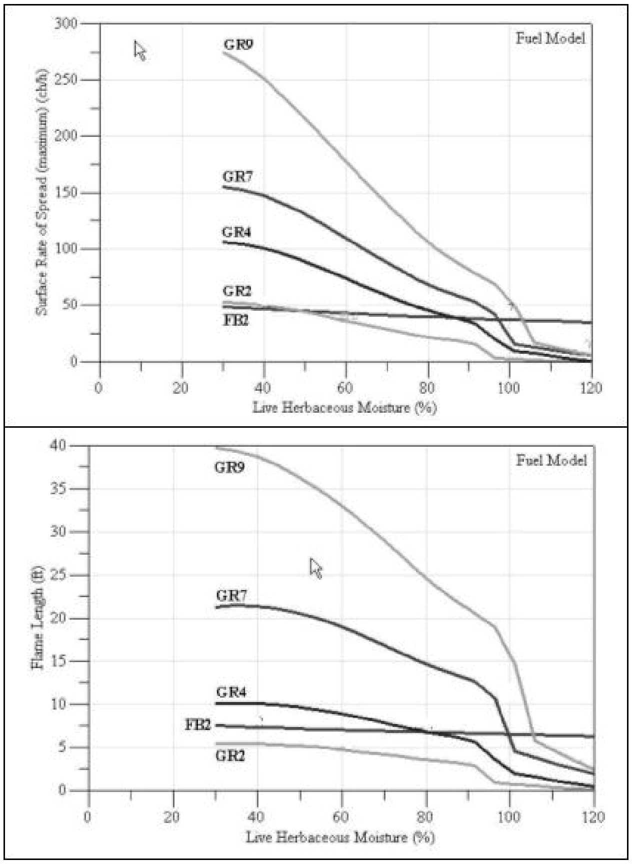

Figure 1.—Effect of dynamic fuel load transfer on surface fire behavior.

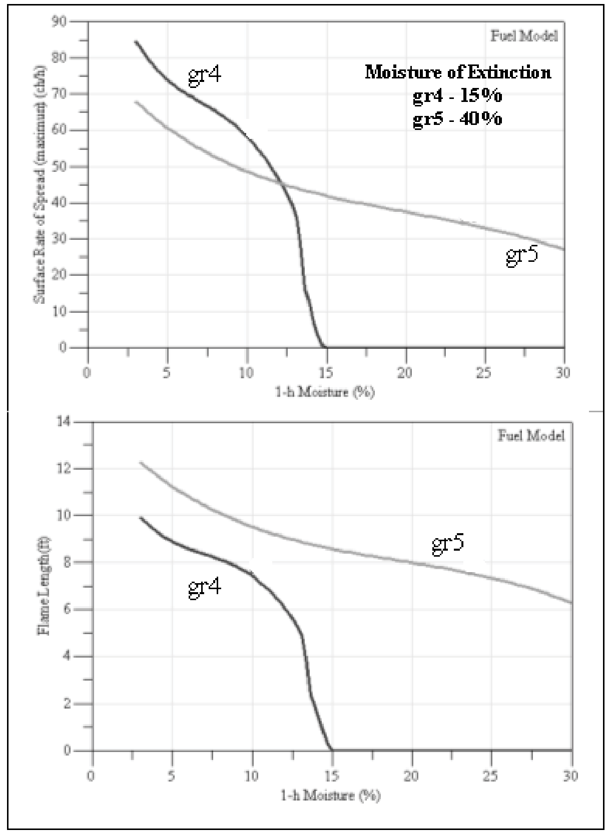

Figure 2.—Moisture of extinction effect on fire behavior.

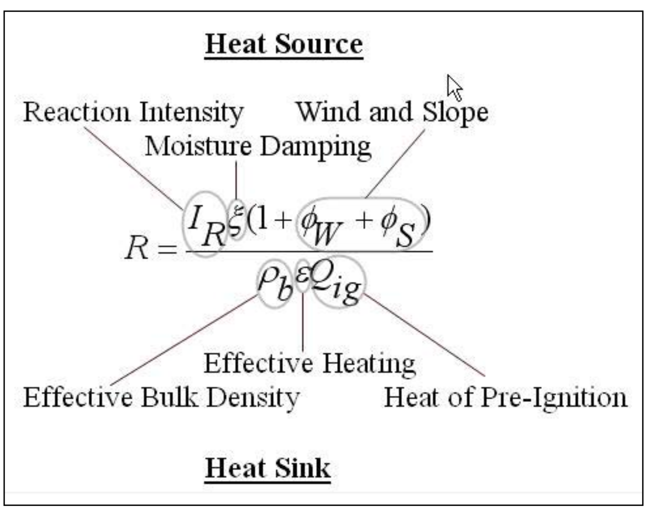

Figure 3.—Rothermel surface fire spread model.

HERBACEOUS FUEL LOAD TRANSFER

Grasses and forbs are highly variable as fuels, changing continuously from the beginning until the end of each fire season. They begin the year as largely dead surface fuels remaining from the previous season's growth, develop significant live fuel loading as they grow, and then become dead fuels through seasonal curing, drought, or frost damage. Until the new fuel models were developed, all live fuels were considered living, with no way to enter fuel moistures as low as those reached by dead fuels. The only way to capture the range of conditions described above was to utilize different fuel models at different stages in the season. For modeling efforts that span only short periods, fuel model classifications did not need to change. However, as landscape fire behavior models become more widely used and managers demand predictions over longer periods (weeks or months), fuel models must be responsive to these growing-season changes.

Table 2.—Surface fire behavior sensitivity to herbaceous load transfer

Rate of Spread (chains per hour)

Fuel Model	Herbaceous Fuel Moisture %									
	95	96	97	98	99	100	101	102	103	104
FB2	38	38	38	38	37	37	37	37	37	37
GR7	46	43	39	34	28	20	16	15	14	14
GR9	72	70	67	64	61	56	50	42	32	20

Flame Length (feet)

Fuel Model	Herbaceous Fuel Moisture %									
	95	96	97	98	99	100	101	102	103	104
FB2	6.6	6.6	6.6	6.5	6.5	6.5	6.5	6.5	6.5	6.5
GR7	12	11	10	9	7.5	5.4	4.5	4.4	4.2	4.1
GR9	20	19	19	18	17	16	15	13	9.8	6.3

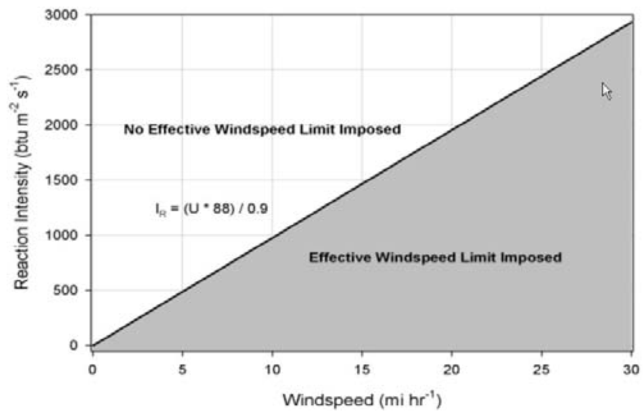

Figure 4.—Reaction intensity and windspeed, with and without windspeed limits.

With the original 13 fuel models, the effects of fuel characteristics and fuel moisture were related but separate, with fuel moisture affecting the moisture damping and heat of pre-ignition factors in the model. Seventeen of the 27 new models with live fuel loads are considered "dynamic." They include an herbaceous fuel load that the user can designate as live, dead, or in proportionate combinations of live and dead. Of the original 13 models, only FB2 (timber grass and understory) has herbaceous loads. It does not accommodate herbaceous load transfer between live and dead fuels.

This dynamic feature was originally designed to operate much the same way that it does in the National Fire Danger Rating System (NFDRS), with the fuel load transfer determined by the estimated herbaceous fuel moisture. Using this method of transfer, the entire herbaceous fuel load is considered live when the herbaceous fuel moisture (HFM) is at least 120 percent and entirely dead at an HFM of 30 percent. Between those two HFM values, the load is distributed proportionately between live and dead.

Andrews et al. (2006) refer to numerous examples of grass fuels in Australia and New Zealand as evidence that the relationship may not be valid. Despite such criticism, most of the software systems that use these dynamic fuel models transfer the load automatically based on HFM. Table 2 highlights the sensitivity of spread and intensity to small changes in HFM. In this example using GR7, if HFM is changed from 101 percent to 98 percent, spread rate more than doubles from 16 to 34 chains per hour.

As this method of fuel load transfer is used in the dynamic models, the effects of fuel moisture extend to other parts of the equation and may have some unanticipated effects on predicted fire behavior. Several of these effects are discussed here.

Windspeed Limit

The most significant impact of the fuel load transfer is that associated with the windspeed limit applied by the model. Figure 4 (equation from Rothermel [1972]) illustrates this limit. For each combination of inputs, resulting reaction intensity is compared to the value represented by the limit. If it falls below the limit value for the input windspeed, the effective windspeed used in the spread equation will be based on the wind limit function. If it exceeds the limit, the actual windspeed is used.

For dynamic models that have relatively high herbaceous fuel loads and/or relatively low 1-hr fuel loads, calculated reaction intensities for most windspeeds will fall below the wind limit. As herbaceous fuel is transferred from live to dead, the calculated value will approach the limit value. Once the limit is exceeded, the windspeed influence increases and results in the rapid shift in spread rate and intensity seen in many of the dynamic models.

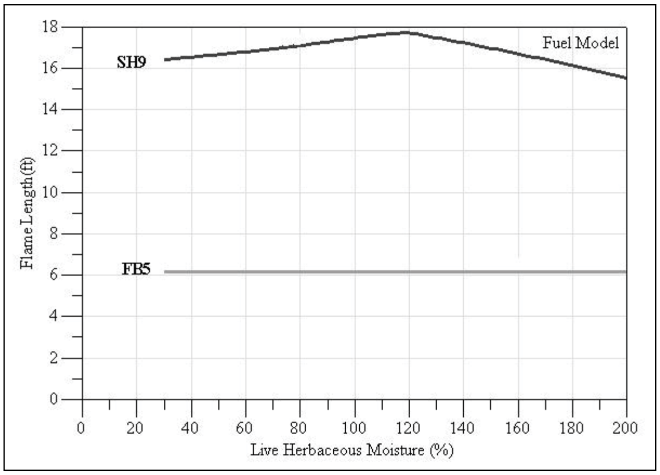

Figure 5.—Linked effect of Live Fuel Moisture on dead fuel characteristics.

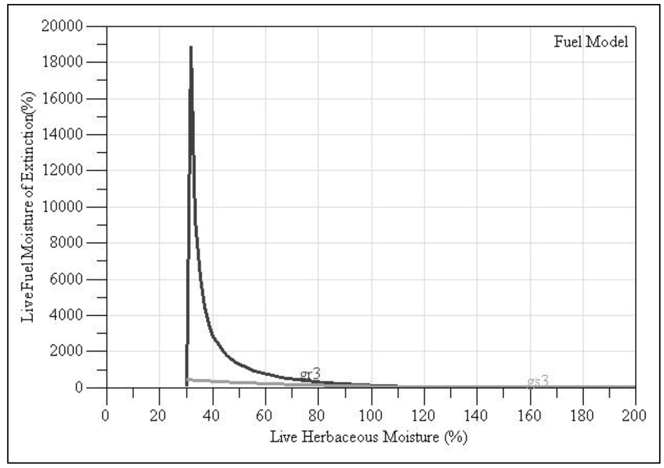

Figure 6.—Live Fuel Moisture and Live Fuel Moisture of Extinction.

Figure 1 illustrates this relationship, which cannot be supported by measurement in the field.

Dead Fuel Characteristics

As herbaceous fuel is transferred from the live category to the dead category, its characteristic SAV is retained. Fuel models that include significant loads of both 1-hr and herbaceous fuels that have different SAV values can have their combined dead SAV altered significantly as herbaceous load is transferred. In the case of SH9 (Very high load, humid climate, shrub), the effect is counter-intuitive, with the modeled spread leveling off and flame length actually decreasing as the herbaceous fuel moisture falls below 120 percent. Figure 5 demonstrates this unrealistic result.

Live Fuel Moisture of Extinction

As with the reaction intensity, separate moisture damping coefficients are calculated for live and dead fuels. Separate values of live and dead Reaction Intensity are calculated and then added together to produce an overall reaction intensity. Though the calculation is the same, there are some important differences. While the ME for dead fuels is a fuel model parameter, the live fuel ME is calculated from the dead fuel moisture, the dead fuel ME, and the ratio of live fuel load to dead fuel load. If the HFM determined the fuel load transferred, the ratio of live to dead load would decrease with decreasing herbaceous fuel moisture. The result can be rapidly increasing live fuel ME and increased Reaction Intensity contributions from live fuels. Figure 6 demonstrates this effect with GR3 (low load, very coarse, humid climate, grass).

Manual Fuel Load Transfer

Version 4 of BehavePlus allows the user to separate herbaceous fuel load transfer from HFM by creating an additional input called fuel load transfer portion if herbaceous fuels are present in the selected fuel model. With this option, the user can directly identify how much of the herbaceous fuel load is transferred from live (with a higher range of fuel moistures) to dead (with much lower fuel moistures possible). Figure 7 shows the relationship between two models of similar fuels, the static FB3 (tall grass) and the dynamic GR6 (moderate load, humid climate, grass). Using dead fuel moistures of 5 percent, 6 percent, and 7 percent, herbaceous fuel moisture of 100 percent, manual transfer of herbaceous load, midflame windspeed of 5 mph, and no slope, we can compare modeled spread rates and flame lengths. While the two fuel models have similar total fuel loads, FB3 loads are entirely in the 1-hr dead fuel class and GR6 loads are largely in the herbaceous fuel category. Modeled fire behavior converges only as the herbaceous fuel load in GR6 is nearly all converted from live to dead.

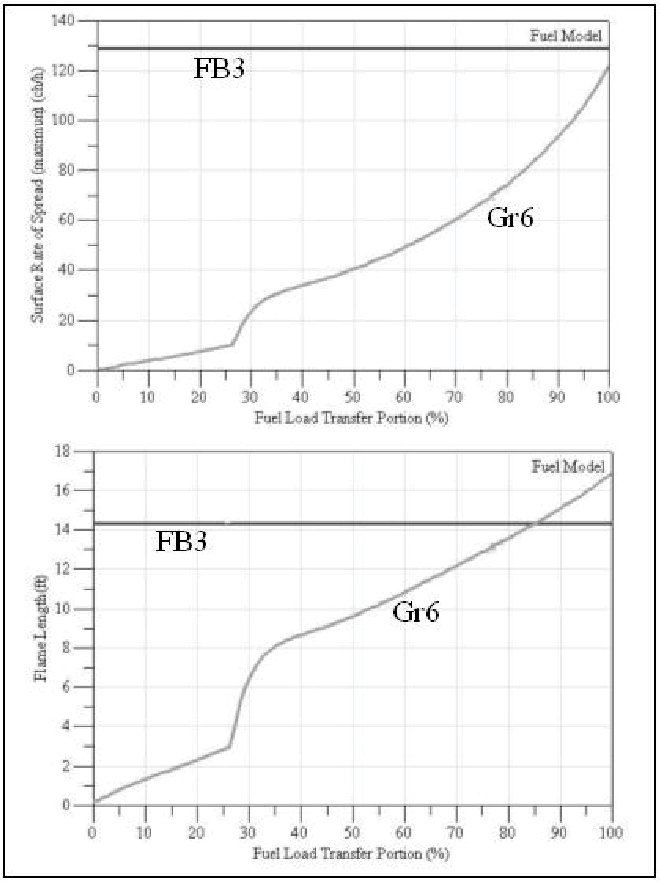

Figure 7.—Manual transfer of herbaceous fuel.

FUEL MODEL DESCRIPTIONS

Within each of the six fuel carrier categories, there are several important distinctions among the fuel models. We provide the following classification to help users select a model within each fuel carrier category:

Grass Fuels (GR)

Static

Dry	*Humid*
FB1-Short grass	FB3-Tall grass
FB2-Timber grass understory	

Dynamic

Dry	Humid
GR1-Short & sparse, dry climate, grass	GR3-Low load, very coarse, humid climate, grass
GR2-Low load, dry climate, grass	GR5-Low load, humid climate, grass
GR4-Moderate load, dry climate, grass	GR6-Moderatae load, humid climate, grass
GR7-High load, dry climate, grass	GR8-High load, very coarse, humid climate, grass
	GR9-Very high load, humid climate, grass

Grass-Shrub (GS)

All Dynamic

Dry	*Humid*
GS1-Low load, dry climate, grass-shrub	GS3-Mod. load, humid climate, grass-shrub
GS2-Mod. load, dry climate, grass-shrub	GS4-High load, humid climate, grass-shrub

Shrub (SH)

Static

Dry	*Humid*
FB4-Chaparral	FB7-Southern Rough
FB5-Brush	
FB6-Dormant brush	
SH2-Mod. load, dry climate, shrub	SH3-Mod. load, humid climate, shrub
SH5-High load, dry climate, shrub	SH4-Low load, humid climate, timber-shrub
SH7-Very high load, dry climate, shrub	SH6-Low load, humid climate, shrub
	SH8-High load, humid climate, shrub

Dynamic

Dry	*Humid*
SH1-Low load, dry climate, shrub	SH9-Very high load, humid climate, shrub

Timber Understory (TU)

Static

Dry	*Humid*
FB10-Timber litter and understory	
TU4-Dwarf conifer with understory	TU2-Mod. load, humid climate, timber-shrub
TU5-Very high load, dry climate, timber-shrub	

dynamic

Dry	*Humid*
TU1-Low load, dry, timber-grass-shrub	TU3-Mod. load, humid, timber-grass-shrub

<u>Timber Litter (TL)</u>

All Static

All have higher ME (25-30 percent).

Conifer	*Hardwood*
FB8-Compact timber litter	
TL4-Small downed logs	
TL7-Large downed logs	
TL1-Low load, compact conifer litter	FB9-Hardwood litter
TL3-Mod load, conifer litter	TL2-Low load, broadleaf litter
TL5-High load, conifer litter	TL6-Mod. load, broadleaf litter
TL8-Long needle litter	TL9-Very high load, broadleaf litter

<u>Slash Blowdown (SB)</u>

Original 13	*New 40*
FB11-Light slash	SB1-Low load activity fuel
FB12-Medium slash	SB2-Mod. load activity or low load blowdown
FB13-Heavy slash	SB3-High load activity or mod. load blowdown,
	SB4-High load blowdown

SELECTING APPROPRIATE FUEL MODELS

With 53 fuel models to choose from, making an individual selection may seem overwhelming. At this point, conventional wisdom suggests that users should keep the two sets (the original 13 models and the new 40) separate. Somewhat differently than outlined by Scott and Burgan (2005), we suggest that users consider the following issues to narrow the choices:

1) First, determine the primary carrier (grass, grass/shrub, shrub, timber understory, timber litter, or slash/blowdown). If at all possible, use the category that matches the vegetative cover type.

2) Evaluate the need to model herbaceous fuel load transfer. Such a model would be recommended for analyses that span longer periods or that include different seasons.

3) Consider moisture of extinction, especially for the grass, grass/shrub, and shrub categories. If the fuels that are burning continue to spread at high dead fuel moisture levels, the humid climate fuels should be considered. However, these fuel models may not accurately recognize the typical cessation of spread during the nighttime hours.

4) Match fuel load distribution, total loading, and bed depth. There may be several choices that represent a range of fire behavior, and these choices are likely to present an appropriate range for both spread and intensity outputs.

5) Finally, evaluate the resulting ranges of spread rate and fireline intensity based on expected weather (represented by wind and dead fuel moisture) and live fuel moisture condition. Comparing these results with expected/observed fire behavior will often make the choice clear. Users should examine model outputs for the range of possible temperature, relative humidity, and wind and slope conditions likely to be experienced. For fuel models with live fuels, examine the effect of live fuel moisture conditions for likely combinations of weather and terrain inputs to insure that results are as expected.

CONCLUSIONS

With the introduction of 40 additional fuel models, users of fire behavior prediction systems are confronted with new opportunities and the increased importance of several concepts. More fuel models are offered with high moisture of extinction distributed across the carrier categories. There are additional models with live fuel loads, especially herbaceous fuel loads that can be transferred from live to dead. This facility, whether handled automatically through linkage to the HFM or by directly entering a fuel load transfer portion, effectively allows the user to create new fuel models as the fuel load transferred changes. By selecting a "dynamic" fuel model, users who have never created or used custom fuel models will be assuming that responsibility. They should do so knowing the effects of that choice. Users should make fire behavior predictions for a range of anticipated or forecasted conditions to insure that the outputs effectively represent observed and potential fire behavior. BehavePlus can effectively be used to evaluate these relationships, especially Version 4, which allows for user designated transfer of herbaceous fuel loads.

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SMOKE MODELING IN SUPPORT OF MANAGEMENT OF FOREST LANDSCAPES IN THE EASTERN UNITED STATES

Gary L. Achtemeier[1]

[1]Research Meteorologist, Center for Forest Disturbance Science, U.S. Forest Service, Southern Research Station, Athens, GA
To contact, call (706) 559-4239 or email at gachtemeier@fs.fed.us

Abstract.—The impact of smoke from forest burning on air quality is a threat to the use of prescribed fire to manage woodlands in the eastern United States. Population shifts from urban centers to the wildland/urban interface have increased human exposures to smoke. Tighter national ambient air quality standards restrict the amount of smoke released over an area. This article reviews smoke models available prior to 1990 and those currently in use or under development. Models have become more sophisticated with advances in computer technology. The outcomes are suites of models that can describe smoke from release to vertical transport to dispersion in ways that realistically simulate how land managers conduct their burns. The results from the models can inform land managers about weather conditions and procedures to do their burns so as to minimize downwind impacts of smoke. However, effects of errors that stem from uncertainties in variables ranging from fuel loadings to weather forecasts temper optimism about the application of these models to eastern forests.

INTRODUCTION

Prescribed fire is used extensively throughout hardwood and pine lands in eastern U.S. forests to achieve a number of land management objectives (Wade et al. 2000, Brose et al. 2005, Wade et al. 2000). One of the adverse consequences of prescribed burning is degradation of air quality (Ward and Hardy 1991, Sandberg et al. 1999, Riebau and Fox 2001). Air pollution from smoke raises the need to balance issues of human health risks, nuisance smoke, visibility impairment, and transportation hazards with issues of forest health and safety, wildlife management, ecosystem restoration, timber production, and carbon sequestration (Achtemeier et al. 1998).

Smoke impacts air quality on two scales. Exposures on the local scale are typically high concentrations of short duration (hours to a day) from a single prescribed burn. Burns are conducted to minimize smoke impacts on "sensitive targets"—schools, hospitals, nursing homes, airports, urban areas, and highways. These goals are not always achieved given local variability of weather conditions, especially wind direction. Exposures on the regional scale are typically relatively low concentrations of smoke (along with other pollutants) of long duration (months to years) from numerous sources spread over a region and over time.

Smoke management is the action by the land manager to minimize the environmental impact of smoke. The amount of smoke put into the atmosphere is determined by how much of the fuel on a landscape is consumed. The rate of transport of smoke into the atmosphere is determined by how the manager spreads fire on the landscape. Backfires release relatively small amounts of smoke per unit time in comparison with stripping, spot ignitions, and aerial ignition. How smoke is distributed vertically in the atmosphere depends on stability and winds during the course of the burn and how much heat is released from burning fuels. Where smoke goes is determined by wind direction and how far smoke goes is determined by wind speed of the air containing the smoke. All of the above are factors in a well written burn plan as land managers consider potential sensitive targets immediately downwind from their proposed burn and more distant targets such as urban areas that could be impacted by the smoke.

In addition, some states have in place smoke management guidelines designed to regulate the amount of smoke allowable within an area. Some guidelines are defined through simple estimates of the capacity of the atmosphere to carry smoke, coupled with experience gained through the number of complaints of nuisance smoke. Other guidelines regulate smoke in relation

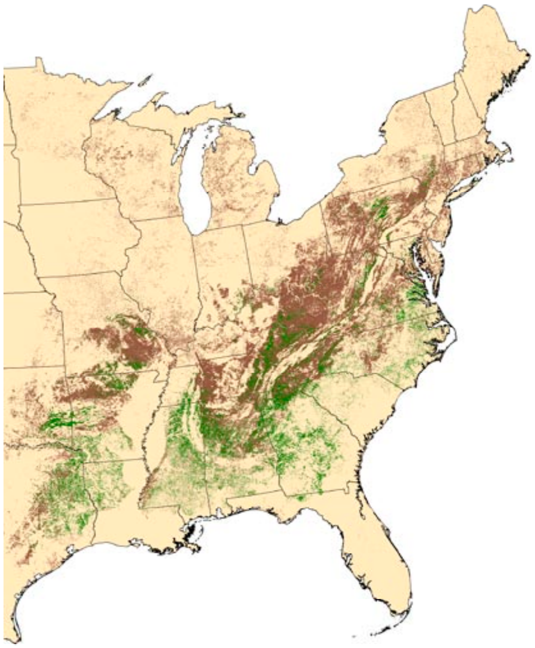

Figure 1.—Map showing eastern forests classified as oak-hickory (brown) and oak-pine (green). Map produced by Daniel Yaussy, U.S. Forest Service.

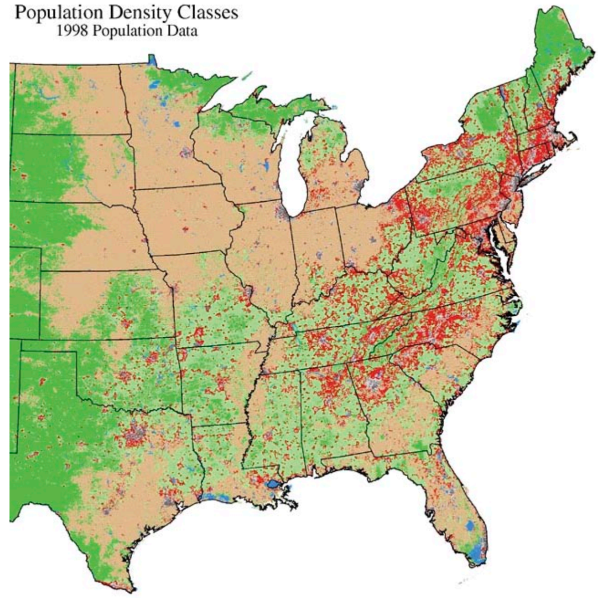

Figure 2.—Map showing the major wildland/urban interface areas in the eastern U.S., where significant human populations (red) are collocated with significant forest (or other natural vegetation) cover (green).

to the distance to downwind sensitive targets. In the future, smoke management guidelines may be linked to national ambient air quality standards, with the current particulate matter ($PM_{2.5}$) limit set by the U.S. Environmental Protection Agency in 2007 at 35 $\mu g\ m^{-3}$ in a 24-hour period on 98 percent of days. (The annual standard remains at 15 $\mu g\ m^{-3}$.)

This paper describes some of the tools that have been developed to aid the land manager and the regulator in answering the primary questions regarding smoke: 1) Where will the smoke go?; and 2) How much will get there? As the emphasis is on eastern forests, some smoke models in application elsewhere will be omitted (Sestak and Riebau 1988, Lahm 2006). Some of the early smoke management tools developed before 1990 will be reviewed. Then the development of plume models designed to simulate smoke from individual fires will be followed. Furthermore, plume models must link with regional-scale air quality models as regional-scale air quality is just the sum of local-scale contributions. Finally, managing smoke during the night differs from managing smoke during the day. This paper will examine tools designed to assist the land manager in tracking the movement of smoke at night with particular emphasis on the problem of smoke/fog impacting roadway visibility.

EARLY SMOKE MANAGEMENT TOOLS

Figure 1 shows the widespread distribution of the oak-hickory and oak-pine forest types in the eastern U.S. In all, eastern forests compose roughly 60 to 70 percent of the forest lands of the contiguous United States. Figure 2 shows population density classes for the same areas. Note the extensive wildland/urban interface (red areas) extending from the southern states through the northeastern states. Only the forests of the western Great Lakes are relatively remote from large population areas.

Because of the extensive blending of population with eastern forest lands, managing smoke for local air quality has long been a critical component of prescribed fire. This necessity is especially the case for the southern states—roughly the states south of the Ohio River and from Texas eastward—where long growing seasons favor rapid fuel buildup, many ecosystems are fire adapted, and there is a fire return interval of 2 to 5 years.

Some of the early operational models for estimating smoke impacts on air quality were influenced by the work of Reiquam (1970), who published a "box model" for air pollution. Reiquam divided the lower atmosphere into boxes with height defined by the top of the boundary layer (mixing height) and length and width set large enough to keep the model computationally feasible but small enough to carry an easily defined mean wind that is representative of the wind throughout the box. The concentration of particulate matter within a box was a ratio of the sum of emissions within the box and particulate matter transported into the box from surrounding boxes with the flux of air passing through the box.

The box model was considered unrealistic for estimating pollution from prescribed fires because of the assumption that emissions were uniformly distributed throughout the box. Because most particulate matter released in smoke is carried to near the top of the mixing layer and then mixed down farther downwind, it was judged that the box model overestimated smoke concentrations until about 100 km from the source of the emissions (Pharo et al. 1976).

Nevertheless, the concept of the ratio of emissions to volume flux of air passing over a burn site took hold. The ventilation index (VI) is defined as the product of the mixing height and the mean wind speed through the layer of the atmosphere from the surface to the mixing height. The crosswind width of the box is a unit dimension needed to maintain the concept of volume flux.

The transport wind speed and direction, mixing height, and VI are routinely transmitted by the National Weather Service in its fire weather forecasts. South Carolina smoke management guidelines classify VI into five "category days" for regulating prescribed burning. The amount of land permitted to be burned is restricted in each class. No sound scientific data exist to define burning limits for the category days. Therefore, these limits are arbitrary, set by the number of complaints of smoke by the public.

Despite the compelling logic of the ventilation index, preliminary results from the Southeastern Smoke Project, a 5-year effort measuring fire activity data, $PM_{2.5}$ concentrations from a network of up to 22 samplers, and concurrent weather data for 56 prescribed burns are showing little to no correlation between smoke concentrations and the VI. Reasons for the lack of correlation remain unclear, but three possibilities stand out. First, for operational considerations, a 24-hour prediction of the VI was used for the analyses. Mixing height and transport wind speed can be difficult to predict. Second, smoke plume dynamics may be far more complex than assumed in the calculation of the VI. Third, VI may be valid at distances from the burns that are greater than distances of data collection.

Plume rise equations developed for stack emissions (Briggs 1969, 1971, 1972) were tested for applicability to prescribed fire by Pharo et al. (1976). One conclusion drawn from Briggs' equations was that plumes from most southern prescribed fires are unable to penetrate a modest inversion of 1 °C at or above 100 m elevation. This conclusion reinforced Reiquam's use of the mixing height as a lid for his box model.

In addition, Pharo et al. (1976) recognized that not all smoke is drawn up into the plume even during the hottest portion of convective lift phase in southern prescribed fires. They suggested a ratio of 60-percent rise to 40-percent no rise for a typical burn.

LOCAL SMOKE MODELS – DAYTIME

VSMOKE

The above information was incorporated into a dispersion model based on Turner's (1970) Gaussian dispersion theory. The outcome was VSMOKE (Lavdas 1996), a smoke screening model that has been adapted for Forest Service applications in the Southeast (Bill Jackson, U.S. Forest Service, Asheville, NC, 2008, personal communication).

Figure 3 shows the fire activity data needed to input emissions data into VSMOKE. Land managers provide the area expected to be blackened (not the area to be burned) and estimates of the fuels that will be consumed (not total fuel loading). These data are input into a combustion model that estimates the amount of fuel consumed. Then a second model estimates the emissions factor (a fraction of the fuel consumed converted into fine particulate matter). Fine particulate emissions are distributed over the course of the burn by a simple emissions production model. The emissions data are needed by VSMOKE to calculate downwind concentrations as a function of distance from the burn. (Note that the processing of fire activity data into emissions production estimates is required by all smoke models that calculate concentrations. Thus plume rise models could be substituted for VSMOKE on the last line of Fig. 3.)

Figure 4 shows an example of a VSMOKE-simulated Gaussian $PM_{2.5}$ concentration pattern for the Brush Creek prescribed burn in eastern Tennessee on March 18, 2006 (Jackson et al. 2007). As smoke spreads downwind (lines diverging from the burn site), mass conservation requires reduction in concentration. Therefore the highest concentrations of smoke and the greatest threat to air quality are found close to the location of the burn (dark maroon colors). VSMOKE gives land managers a rough estimate where smoke will go and how much will get there, given planned fire activity and prevailing weather.

VSMOKE technically is not a plume rise model. The user specifies how much smoke is released at the ground and how much smoke is released near the top of the mixing layer. In addition to Pharo et al.'s (1976) suggested 60-percent rise to the top of the mixing layer, a 75/25 percent ratio is also used. Therefore, although the assumption that all smoke is confined to the mixing layer is workable for the small prescribed fires for which the model was developed, it is known that plumes from large prescribed fires and wildfires rise above the mixing height and sometimes by several thousand meters. Consequently, much fine particulate matter can be transported above the boundary layer and away from ground-level sensitive targets. Furthermore, VSMOKE is a steady-state model. It does not account for vertical wind shear or for changes in wind conditions during the course of the burn. The model assumes a spatial steady state and therefore is not valid for smoke plumes over complex terrain. The outcome is that VSMOKE can overestimate smoke concentrations.

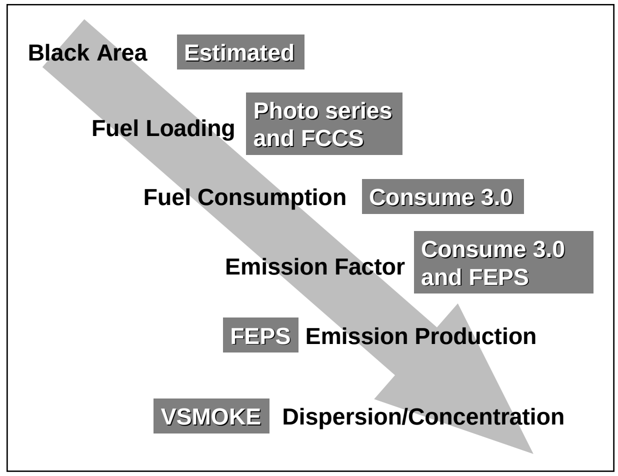

Figure 3.—Modeling framework required for the input of emissions data into VSMOKE.

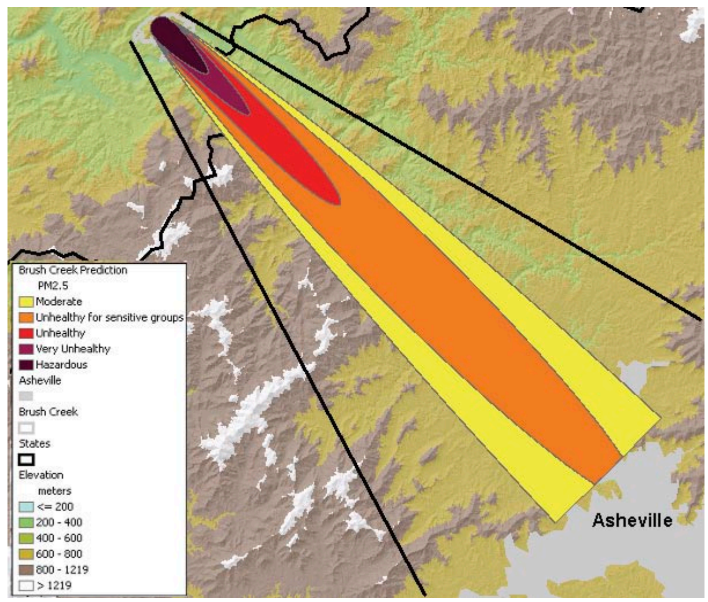

Figure 4.—Example of a VSMOKE-simulated concentration pattern from the Brush Creek prescribed fire in eastern Tennessee, March 18, 2006 (from Jackson et al. 2007).

CALPUFF

Of the many plume rise models that have been developed over the past several decades, CALPUFF is described here because it was selected as the plume rise model for the U.S. Forest Service regional-scale smoke production, transport, and dispersion framework, BlueSky. (More will be presented on BlueSky in the section on long-range transport.) CALPUFF removes many of the assumptions that limit VSMOKE (Scire 2000). CALPUFF is a nonsteady state puff model, meaning that emissions are injected into the atmosphere as self-contained volumes defined as “puffs.” The puff grows as it entrains ambient air. Thus, as the puff volume increases, the concentration of particulate matter contained in the puff decreases as the puff drifts downwind. Each puff is independent from all other puffs. This assumption makes CALPUFF ideal for simulating smoke transport and diffusion in nonsteady flows, including temporally and/or spatially varying flow fields due to influences of complex terrain, non-uniform land-use patterns, and coastal effects. CALPUFF also works for stagnation conditions characterized by calm or very low wind speeds with variable wind directions.

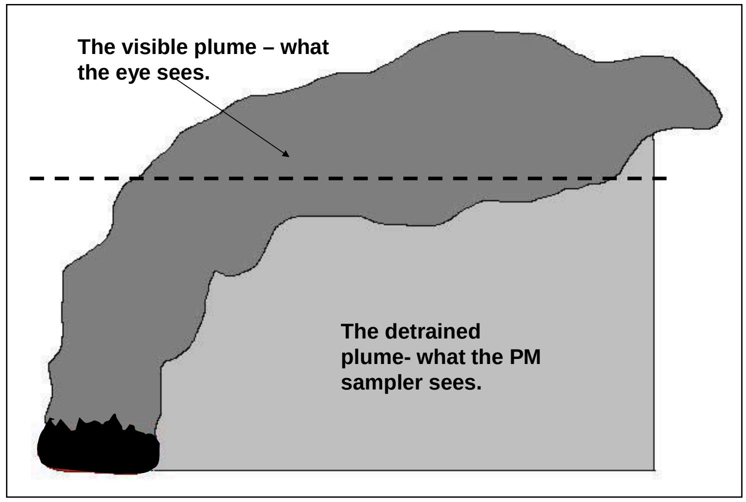

Figure 5.—A conceptual model of a “typical” smoke plume (darkened area) and a trailing “pall” of detrained particulate matter. The top of the mixing layer is given by the dashed line.

Daysmoke

Like CALPUFF, Daysmoke removes many of the assumptions that limit VSMOKE. Daysmoke is an extension of ASHFALL, a plume model developed to simulate deposition of ash from sugar cane fires (Achtemeier 1998). As adapted for prescribed fire, Daysmoke consists of four models: an entraining turret model, a detraining particle model, a large eddy parameterization for the mixed boundary layer, and a relative emissions model that describes the emission history of the prescribed burn.

From photogrammetric analysis of video footage of smoke plumes from burning sugar cane, Achtemeier and Adkins (1997) determined that a rising smoke plume could be described by a train of rising turrets of heated air that sweep out a three-dimensional volume (the plume pathway) as they expand through entrainment of surrounding air. The plume pathway can be envisioned as an inverted cone bent by the prevailing winds. The plume pathway ends where the plume vertical velocity falls below a prespecified value (currently 0.5 m s^{-1}).

Daysmoke does not contain all smoke within the plume pathway. A stream of particles representing smoke is released through the plume pathway. Each particle represents a mass of smoke. For example, for current applications of Daysmoke, each particle represents 1 kg of $PM_{2.5}$ particulate matter. As the particles make their way up the plume pathway, turbulent mixing represented by stochastic equations in the model may place some particles outside of the plume. These particles are assumed to have been detrained from the plume and are subject to circulations within the free atmosphere. Thus, a Daysmoke plume can be represented schematically by Figure 5. For each plume pathway, three outcomes are possible. Smoke may exit the top of the mixing layer (dashed line), smoke may remain as a plume within the mixing layer, or smoke may be detrained.

Additional time-dependency is built in through the relative emissions model. An accurate emissions production model must estimate emissions as they change during the ramp-up, maximum combustion, and ramp-down phases of the burn. Timing of emissions production must match the diurnal evolution of the depth of the boundary layer so that the discharge of particulate matter within and above the mixing layer can be simulated with accuracy.

Given emissions and heat production, Daysmoke updates the plume pathways every 10 minutes. Furthermore, the relative emissions model permits multiple plume pathways simultaneously. This feature of Daysmoke allows for multiple-core updraft plumes. In comparison with single-core updrafts, multiple-core updrafts have smaller updraft velocities, are smaller in diameter, are more impacted by entrainment, and are therefore less efficient in the vertical transport of smoke. Single-core updraft plumes would be expected to transport smoke

Figure 6.—Plume from the Brush Creek prescribed burn on March 18, 2006.

higher, more likely penetrating the mixing layer and injecting smoke into the free atmosphere, where it can be transported downwind with minimal impact on ground-level targets. Multiple-core updrafts, as less efficient transporters of smoke, would be expected to grow less high and deposit most of the smoke within the mixing layer—where it can more readily impact sensitive targets.

The importance of multiple-core updraft plumes in the vertical transport of smoke is demonstrated with the Brush Creek prescribed burn on March 18, 2006 (Liu et al. 2006, Jackson et al. 2007). This burn caused a smoke incident at Asheville, NC, about 50 km from the burn. Figure 6 shows the multiple-core updraft structure of the plume. Two updraft cores are easily visible in the image. An additional one to three updraft cores can be deduced from the shape of the surrounding plume. Daysmoke was run 10 times for each updraft core number out to 10 updraft cores. Figure 7 shows the mean and distribution for each updraft core number compared with the maximum hourly $PM_{2.5}$ concentration (dashed line) observed at Asheville. A one-core updraft plume would have produced $PM_{2.5}$ concentrations of 45 µg m^{-3} at Asheville. By contrast a ten-core updraft plume would have produced $PM_{2.5}$ concentrations of approximately 240 µg m^{-3} at Asheville. The ten-core concentrations are five times greater than the one-core solution. The four-core solution produced $PM_{2.5}$ concentrations of approximately 140 µg m^{-3} at Asheville—the maximum hourly amount observed.

Daysmoke also has informed us on the role of the wind speed on the vertical distribution of smoke. When the wind speed is relatively strong, the plume pathway will be bent over and more smoke will be discharged in the mixing layer in comparison with relatively light wind speeds. Regarding light wind speeds, heat of combustion can organize the plume to stand more erect with a greater chance that smoke will be injected into the free atmosphere above the mixing height. Figure 8 shows the vertical distribution of smoke for two Daysmoke simulations for one-core updraft plumes. As Daysmoke was coupled with the Community Multiscale Air Quality (CMAQ) model for these simulations, the vertical coordinate is given by CMAQ layer. The mixing height is given by the dashed line and was held constant at 1 km during the simulations. The left panel shows that 33 percent of the smoke was injected above the mixing height. Smoke within the mixing layer increased toward the ground in response to the contribution of weaker plumes during the ramp-up and ramp-down phases of

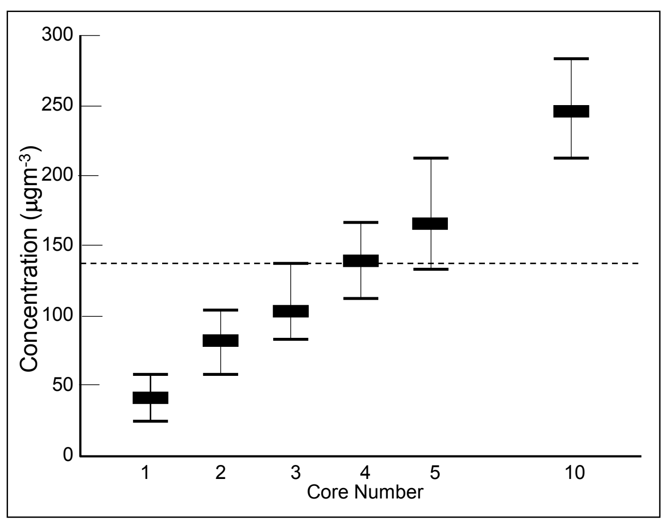

Figure 7.—The means and distributions for each updraft core number compared with the maximum hourly $PM_{2.5}$ concentration (dashed line) observed at Asheville, NC.

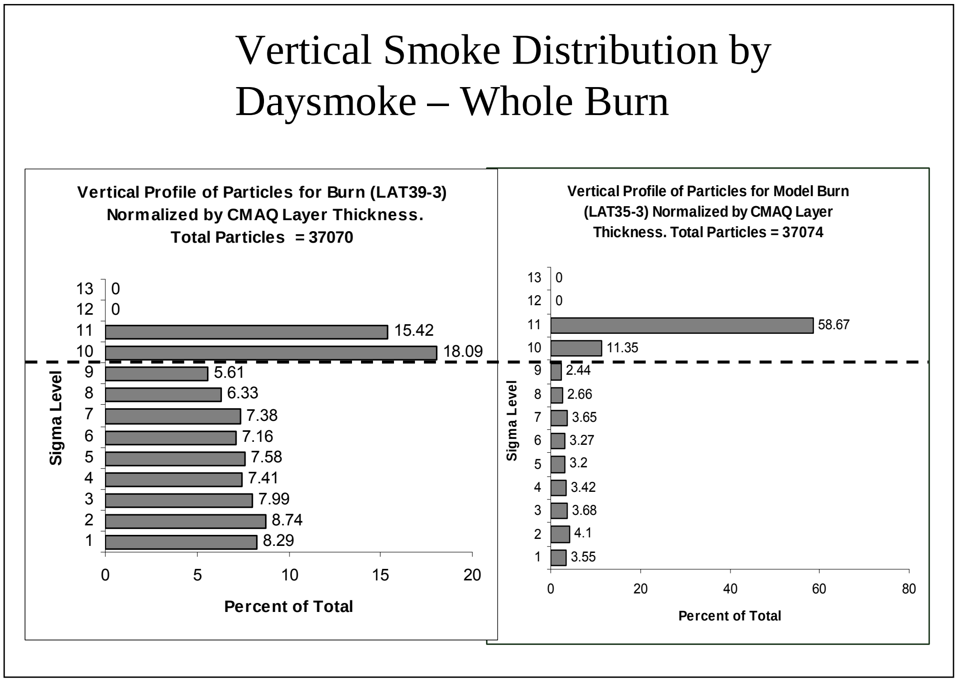

Figure 8.—Left panel: Whole burn distribution of smoke by CMAQ layer for a one-core updraft simulation. Right panel: Same as for the left panel except with the wind speed cut in half.

the burn. When the wind speed was cut in half, roughly 70 percent of the smoke was deposited above the mixing height (right panel). The caveat is that smoke injected into the free atmosphere during light winds will not disperse much and can cause major air quality problems should weather conditions change to bring the plume back to the ground.

REGIONAL SMOKE MODELS

BlueSky

BlueSky was developed as part of a multi-agency effort to simulate and predict smoke from approved or planned prescribed fires, agricultural fires, and wildfires (O'Neill et al. 2003). It couples off-the-shelf weather, fuels, consumption, emissions, and dispersion models in a modular framework in order to produce these real-time predictions. By gathering and using information on all fire activity in a region, BlueSky not only predicts the smoke $PM_{2.5}$ impacts from a single fire, but also predicts cumulative impacts from multiple fires. BlueSky cannot be separated from highly complex weather models, as it is totally dependent on this input to develop smoke simulations for operational application. CALPUFF is the plume rise model currently preferred for BlueSky.

Table 1 shows potential uses and users of BlueSky. BlueSky is a decision-support tool that predicts smoke impacts from wildland fire and delivers these forecasts to managers through the Web. Like any complex computer-automated decision support system, BlueSky must have data and infrastructure to support its operation. In this case, the tool requires significant fire activity data and computer resources to operate, and expertise to produce meaningful assessments of products. Table 2 shows implementation locations through the U.S. Forest Service Fire Consortia for Advanced Modeling of Meteorology and Smoke (FCAMMS) centers. Versions of the modeling framework have been available through the Web at all FCAMMS sites since 2005. However, BlueSky is constantly being upgraded and versions may vary from center to center. An example of a BlueSky run from the FCAMMS-EAMC (Eastern Area Modeling Consortium) is given in Figure 9.

A project was undertaken in 2005 to validate BlueSky for wildfires (Riebau at al. 2006). One of the outcomes was that BlueSky seems likely to be underestimating near-field smoke concentrations and potentially overestimating far-field smoke concentrations. Subsequent conversations

Table 1.—Potential uses and users of BlueSky (after Riebau et al. 2006)

Potential Uses and Users of BlueSky	Fires			Users								
	Rx	WFU[a]	Wildfire	Fire Mgrs	GACC[b]	IMT[c]	Regulators	Air Agencies	Health Agencies	Local Officials	Public	Researchers
Decision-making Support (e.g., go/no-go, WFU classification)												
Fire-fighting Tactics (e.g., effects of burnouts)												
Fire-fighting Resource Deployment (e.g., aviation)												
Improved Knowledge about Emissions, Climate												
Location of Air Quality Monitors												
Public Safety/Transportation												
Single and Multiple Fire Impacts (integrate across boundaries)												
Notify Public (health/nuisance/visibility)												
Regulatory – emission inventory/ apportionment/background												
After-action Assessments and Evaluations, Exceptional Event Analysis												

[a] Wildland fire use
[b] Geographic Area Coordinating Center
[c] Incident Management Team

Table 2.—BlueSky implementation locations (after Riebau et al. 2006)

Domain	Resolution	FCAMMS[b]	Station[c]	Wildfire	Rx fire	Ag fire	RAINS	In Operation Since
Northwest	12 km 4 km	NWRMC	PNW			*		2002
N. California	4 km	CANSAC	PSW					2004
S. California	4 km	CANSAC	PSW					2004
Northeast	12 km	EAMC	NRS		M			2004
Rocky Mountain	6 km	RMC	RMRC		M			2004
Southeast	12 km 4 km	SHRMC	SRS		M		a a	2005
West (demo)	36 km 12 km	NWRMC / RMC	RMRC		M			2005

M = manual only.
[a] in progress as of March 2006.
[b] Fire Consortia for Advanced Modeling of Meteorology and Smoke. NWRMC = Northwest Regional Modeling Consortium; CANSAC = California and Nevada Smoke and Air Consortium; EAMC = Eastern Area Modeling Consortium; RMC = Rocky Mountain Modeling Consortium; SHRMC = Southern High Resolution Modeling Consortium.
[c] U.S. Forest Service Research Stations, PNW = Pacific Northwest; PSW = Pacific Southwest; NRS = Northern Research Station; RMRS = Rocky Mountain Research Station; SRS = Southern Research Station.

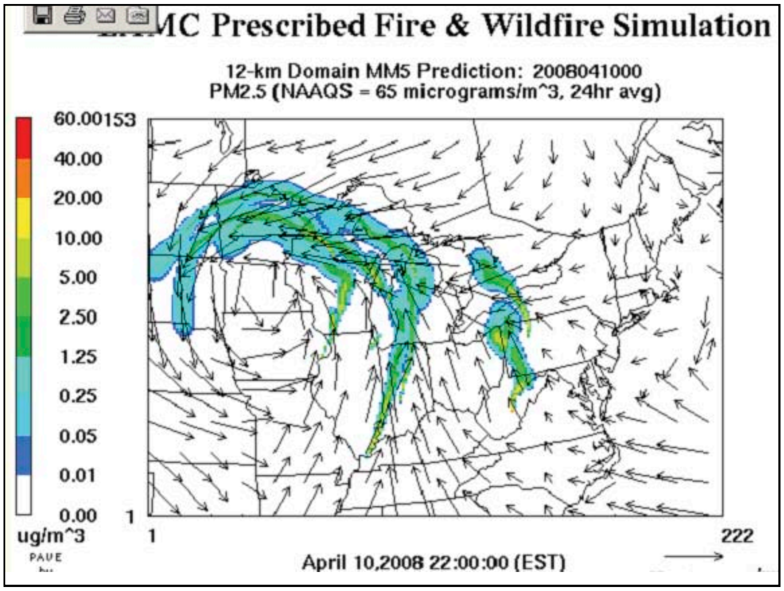

Figure 9.—Example of BlueSky output for April 10, 2008.

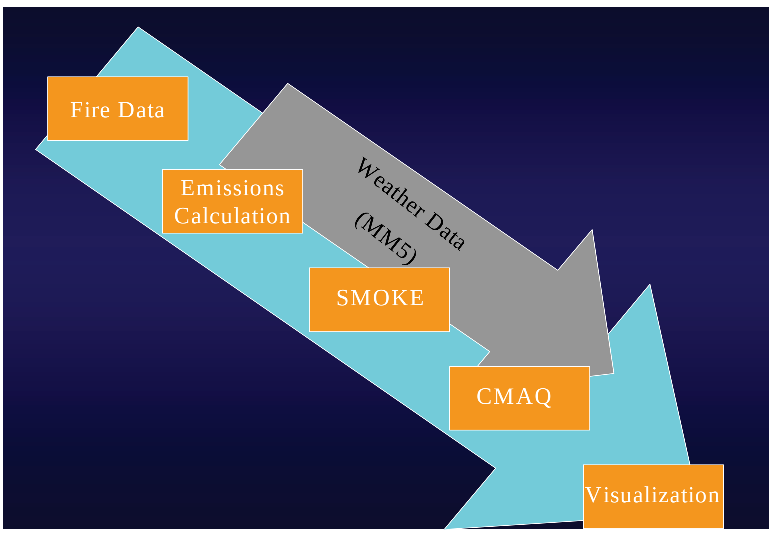

Figure 10.—An overview of the 4S framework.

with one of the authors of the study revealed that much of the error can be explained by the lack of accounting for multiple-core updraft plumes described with reference to Daysmoke.

Southern Smoke Simulation System (4S)

The Southern Smoke Simulation System couples the contribution of smoke from wildland burning with the overall air pollution budget over the southeastern United States. An overview of 4S is shown in Figure 10. Each box along the blue arrow represents steps needed to accomplish the objective of including emissions from wildland fires in regional-scale air quality models. The first box, Fire Data, gets the Southern High Resolution ModelingConsortium (SHRMC)-4S started. Information on the size of the tract of land to be burned, the date and time of the burn, the location of the burn, plus pertinent data on the kinds and state of fuels is supplied by the land manager. Fire activity data are processed through the software CONSUME and EPM (as does BlueSky) to produce emissions inventories for the burns (the Emissions Calculation box). The outputs—(the fire products)—are hourly productions of heat and the masses of gases and particulate compounds. The Sparse

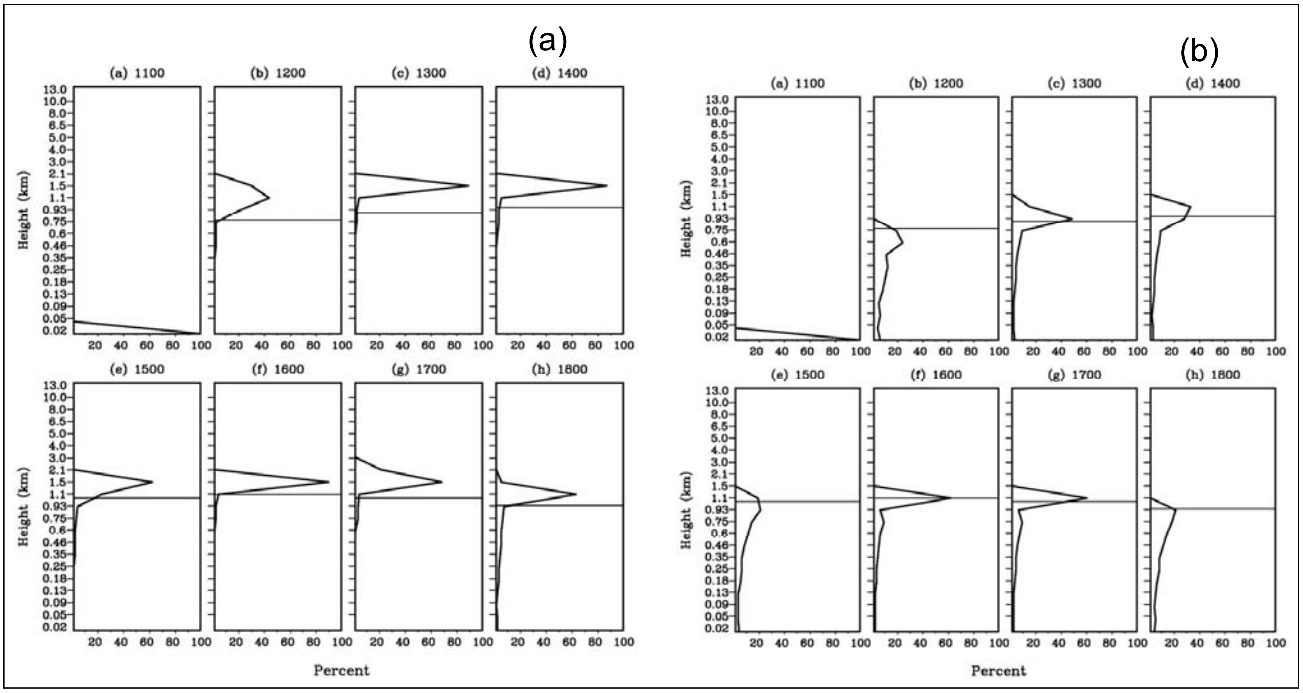

Figure 11.—The vertical distribution of smoke particles (in %) at the hours from 1100 throughout 1800 LST (local standard time) calculated using Daysmoke; (a) one-core updraft, (b) ten-core updraft (from Liu et al. 2006). The light horizontal lines indicate the top of planetary boundary layer.

Matrix Operator Kernel Emissions Modeling System (SMOKE) (Houyoux et al. 2002) processes emission data and provides initial and boundary chemical conditions for the CMAQ model (Byun and Ching 1999) for chemical modeling (Fig. 10, fourth box). A visualization method for illustrating modeling results is the last step. The National Center for AtmosphericResearch (NCAR)/ Penn State Mesoscale Model (MM5) (Grell et al. 1994) is used for providing meteorological conditions for emission calculation and SMOKE and CMAQ simulation.

Daysmoke, as the smoke injector simulator for 4S, allows for the representation of multiple-core updraft plumes. Figure 11 shows the heights of the smoke plume (plume rise) and vertical profiles of smoke $PM_{2.5}$ simulated with Daysmoke for one-core and ten-core updrafts for the course of the Brush Creek burn (Jackson et al. 2007). For the one-core updraft (left panels of Fig. 11), plume rise is about 1.5 km from 1200 to 1600 LST (local standard time) and increases to 2.1 km at 1700 LST. For the 10-core updraft (right panels of Fig. 11), plume rise is about 0.75 km at 1200 LST with the largest percentage occurring at about 0.6 km. Plume rise gradually increases to 1.1 km at the next hour and remains there until 1700 LST. It decreases to 0.92 km at 1800 LST.

These results show two differences between modeling smoke with one-core and ten-core updrafts. First, compared with single-core updrafts, multiple-core updraft plumes do not rise as high. Thus, more smoke mass is distributed at lower levels in the atmosphere. Second, plume rise simulations for one-core updrafts place the most smoke higher above the mixing layer than do simulations for ten-core updrafts. This discrepancy results in significant impacts on the ground concentrations when Daysmoke smoke profiles are linked to CMAQ.

Figure 12 shows time-height cross sections of $PM_{2.5}$ concentrations over Asheville as simulated by Daysmoke/ CMAQ run on a 12-km grid. The plume reaches Asheville after 1500 LST. Both simulations show two peaks in concentrations (an outcome of a 1-hour lapse in aerial ignition), the first arriving at 1600 LST and the second arriving at 1800 LST. The main difference between the one-core and ten-core updraft simulations is in the vertical distributions of smoke.

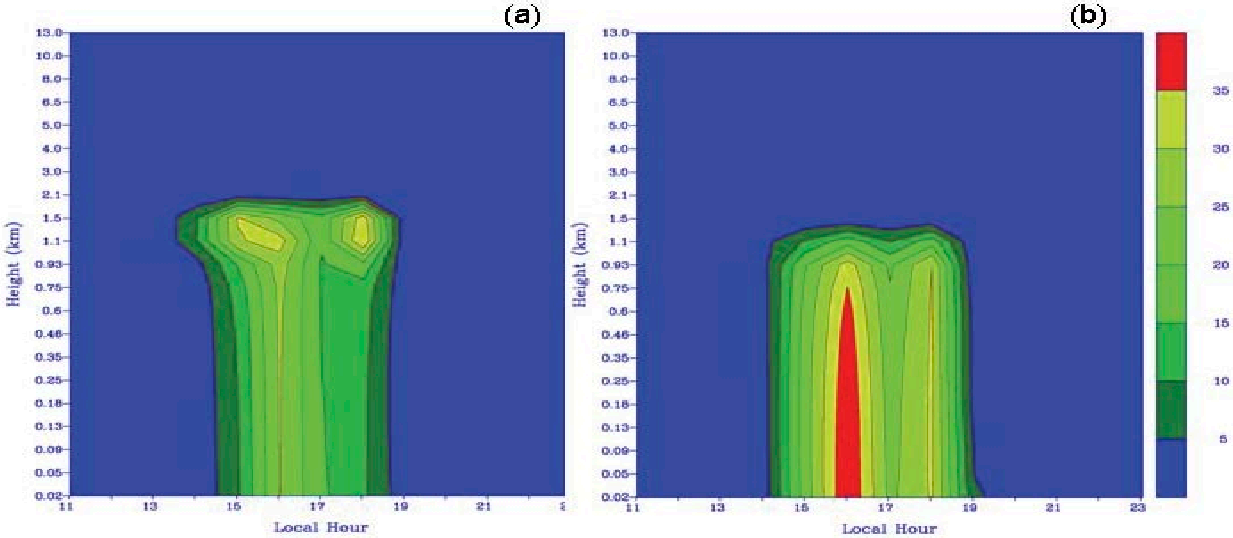

Figure 12.—Time-height across section of PM2.5 concentration (µg m-3) at Asheville simulated with CMAQ with plume rise and smoke particle vertical profile specified with Daysmoke; (a) one-core updraft, (b) ten-core updraft (from Liu et al. 2006).

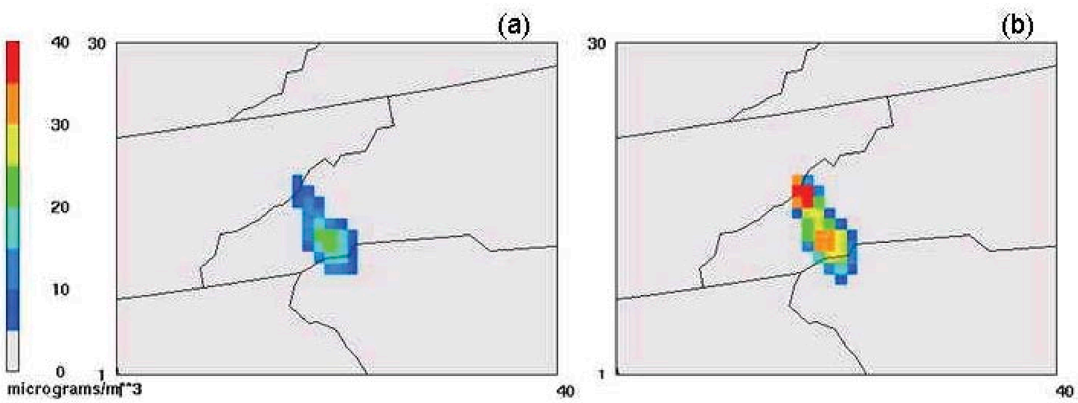

Figure 13.—Spatial distribution of ground $PM_{2.5}$ concentration (µg m^{-3}) at 1700 LST simulated with CMAQ using plume rise and smoke particle vertical profile specified with Daysmoke; (a) one-core updraft, (b) ten-core updraft (from Liu et al. 2006).

Large concentrations are found between 1.1 and 1.5 km above ground for the one-core updraft, and within about 1 km above ground for the ten-core updraft. The one-core simulation placed most smoke far enough above the planetary boundary layer (PBL) that few particles were transported to the ground (Fig. 12a). As Figure 12b shows, most particles are found within the PBL for the ten-core updraft simulation and they are nearly uniformly distributed from the ground to the top of PBL by strong turbulent mixing.

In comparison with concentrations produced directly from Daysmoke (Fig. 7), ground-level concentrations were much reduced. Figure 13 shows the horizontal distribution of ground-level $PM_{2.5}$ at 1700 LST, when largest concentrations were observed at Asheville. The smoke plume spreads from the burn site south-southeastward to the North Carolina-South Carolina border. The under-prediction by CMAQ is primarily due to the 12-km resolution of the model domain, which spreads the plume too widely. The magnitudes of the concentrations for the ten-core updraft are about two to three times that for the one-core updraft. That ratio is five times in Figure 7.

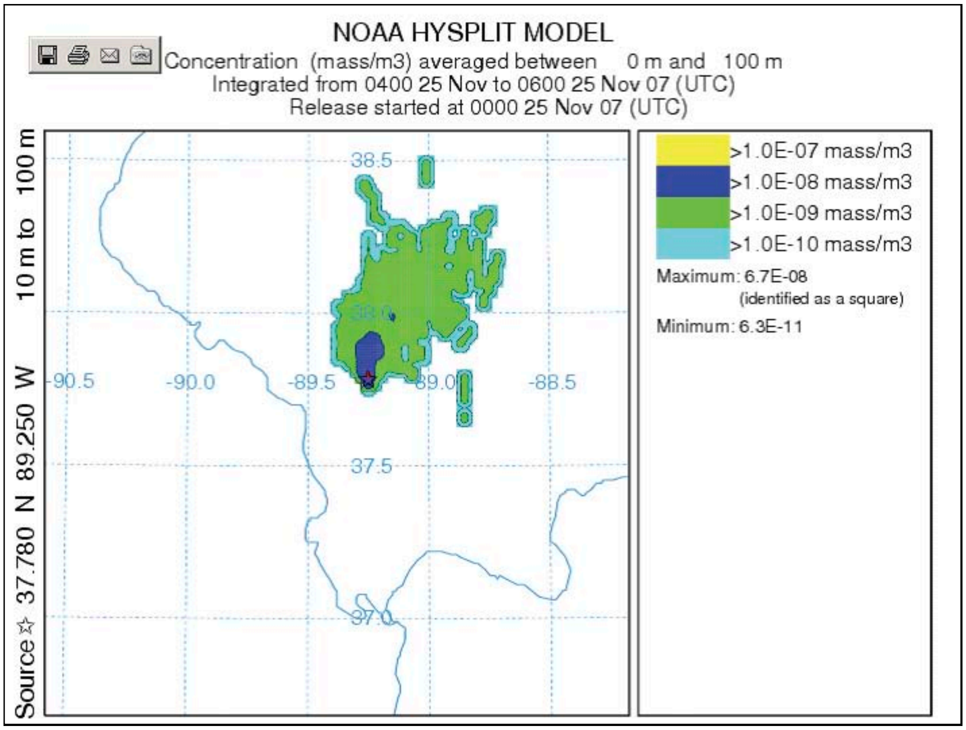

Figure 14.—An example of a HYSPLIT simulation of a hypothetical release of pollution at Carbondale, IL, on Nov. 25, 2007.

HYSPLIT

The HYbrid Single-Particle Lagrangian Integrated Trajectory (HYSPLIT) model (Draxler and Rolph 2003, Rolph 2003) is a complete system for computing simple air parcel trajectories to complex dispersion and deposition simulations. A joint effort between the National Oceanic and Atmospheric Administration and Australia's Bureau of Meteorology, the model has recently been upgraded to include modules for chemical transformations.

The dispersion of a pollutant is calculated by assuming either puff or particle dispersion. In the puff model, puffs expand until they exceed the size of the meteorological grid cell (either horizontally or vertically) and then split into several new puffs, each with its share of the pollutant mass. In the particle model, a fixed number of initial particles are advected about the model domain by the mean wind field and a turbulent component. The model's default configuration assumes a puff distribution in the horizontal and particle dispersion in the vertical direction. In this way, the greater accuracy of the vertical dispersion parameterization of the particle model is combined with the advantage of having an ever-expanding number of particles represent the pollutant distribution.

The model can be run interactively on the Web. Figure 14 shows a sample HYSPLIT run for Nov. 25, 2007 for a pollutant release from Carbondale, IL. HYSPLIT currently does not contain a plume rise model. Therefore, releases are done with the assumption of neutral buoyancy, but the user can specify the altitude of a release.

Full Physics Models

Plume rise models such as CALPUFF and Daysmoke represent a class of empirical models in which complex physical processes occurring in nature are approximated or parameterized by a set of simplifying equations. Some of these equations require pre-assigned coefficients. Liu (Yongqiang Liu, U.S. Forest Service, Athens GA, personal communication) performed a Fourier Amplitude Sensitivity Test (FAST) to identify which of the 13 parameters in Daysmoke impact plume rise. Table 3 shows the parameters and the ranges of values assigned.

Table 3.—Ranges of values assigned for the 13 parameters of Daysmoke and for two environmental parameters

Parameter	Meaning	Range
• Cp	Plume turbulence coefficient	0.05-0.1
• Cu	Air horizontal turbulence coefficient	0.1-0.2
• Cw	Air vertical turbulence coefficient	0.05-0.1
• Kx	Thermal horizontal mixing rate	1.0-1.5
• Kz	Thermal vertical mixing rate	1.0-1.5
• Wc	Plume-to-environ. cutoff velocity	0.2-0.8
• w*	Air induced ash downdraft velocity	0.01-0.02
• Wr	Maximum rotor velocity	1.0-1.5
• Pk	Entrainment coefficient for plume	0.1-0.5
• W0	Initial plume vertical velocity	5.0-15.0
• TD	Initial plume temperature anomaly	5.0-15.0
• Fd	Effective diameter of flaming area	-25%-25%
• Nc	Number of core	1-20
• Tm	Stability	-25%-25%
• Vm	Wind	-25%-25%

The analysis included two ambient variables (stability and wind speed) that are not pre-assigned coefficients. The input parameters, assumed to be mutually independent, are varied simultaneously through the ranges of values. If Daysmoke were run for 10 values within the range for each input parameter, it would take 10^{15} runs to evaluate the impact of the parameters on the model. However, only 1,027 runs were required using FAST.

Figure 15 show the relative importance of the 15 parameters to plume rise in Daysmoke. The first eight parameters, related to in-plume and atmospheric turbulence, are not important in determining plume rise. Only four model parameters substantially impact plume rise: entrainment coefficient (Pk), the number of updraft cores (Nc), the effective plume diameter (Fd), and the initial plume temperature (TD). The initial plume vertical velocity (Wo) showed little impact on plume rise.

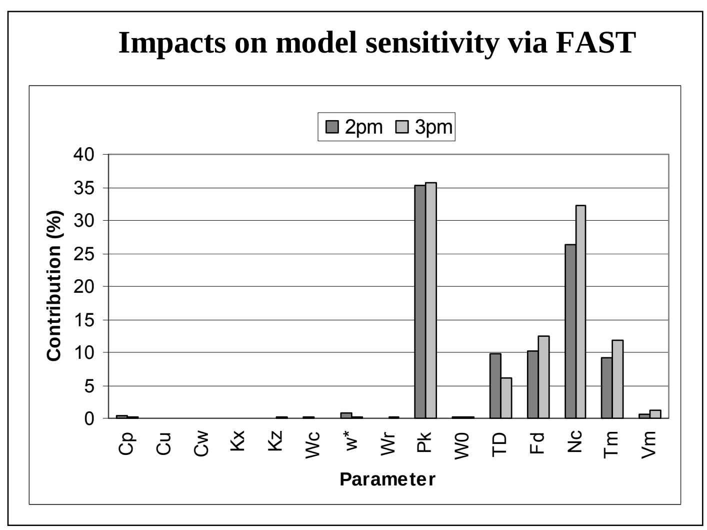

Figure 15.—The relative importance of 13 parameters and two environmental variables to plume rise in Daysmoke. (See Table 3 for definitions of the parameters.)

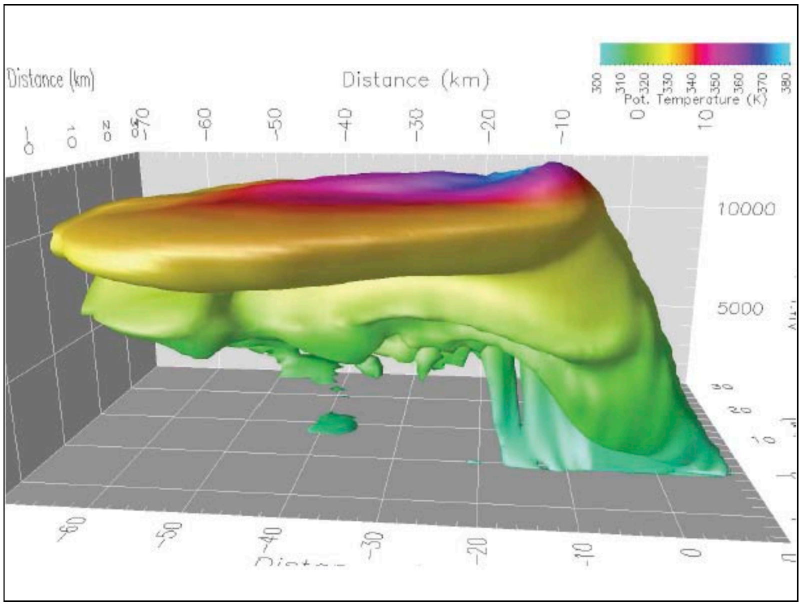

Figure 16.—The smoke plume generated by the ATHAM model for a wildfire in Alberta, CAN, on May 29, 2001 (after Luderer et al. 2006).

Full physics models have the advantage of few to no pre-assigned parameters. Specification of initial plume diameters, temperatures, or vertical velocities are not required because full physics models develop plumes above areas of heat produced by combustion as fire spreads through model landscapes.

A full physics plume model that has been applied to wildland fire is the active tracer high-resolution atmospheric model (ATHAM) (Herzog et al. 1998, Oberhuber et al. 1998). Trentmann et al. (2002) simulated a prescribed burn in northwestern Washington which closely approximated measured elevations and concentrations of smoke. Furthermore, Trentmann et al. (2006) and Luderer et al. (2006) showed how meteorological dynamics coupled with a large wildfire in Alberta, Canada, could generate a pyrocumulus that reached an altitude of about 13 km. Figure 16 shows the 100-μgm^{-3} iso-surface of aerosol concentration after 40 min. of integration. The figure gives an example of the detail obtainable with full physics models.

One disadvantage of the full physics models is the enormous computation time required. These models currently are unsuitable for simulating plume rise for hundreds of fires daily as required for operational use. However, if the increase of computational speed and power continues at the rate of the past decade, it is conceivable that full physics models may eventually find operational application.

LOCAL SMOKE MODELS – NIGHT: ROADWAY VISIBILITY AND SUPERFOG

The environmental impact of smoke is often measured in terms of regional-scale concentrations, impacts on sensitive targets, and individuals' respiratory function. However, a more deadly smoke impact, in terms of lives lost and personal injury, is from local smoke transport at night and its effects on roadway visibility. Smoke entrapped near the ground in nocturnal inversions can drift into populated areas and impact residents, particularly those with respiratory problems. Smoke-laden air masses can drift across roadways and contribute to poor visibility. Smoke and associated fog have been implicated in multiple-car pile-ups that have caused fatalities, numerous physical injuries, and heavy property damage (Mobley 1989).

A particularly deadly smoke/fog incident occurred during the early morning of Jan. 9, 2008, when smoke from a 1,200-ha wildfire flashed into dense fog and drifted across Interstate 4 between Orlando and Tampa, FL. The resulting 70-vehicle pile-up killed five and injured 38.

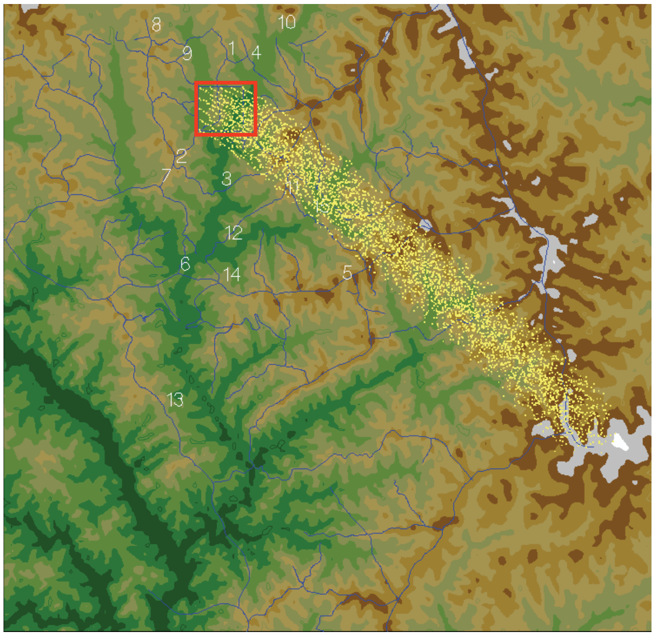

Figure 17.—PB-Piedmont simulated ground-level smoke plume for 1800 EST Jan. 15, 2003, approximately 15 minutes after sunset. PB-P map is 12.6 km x 12.6 km.

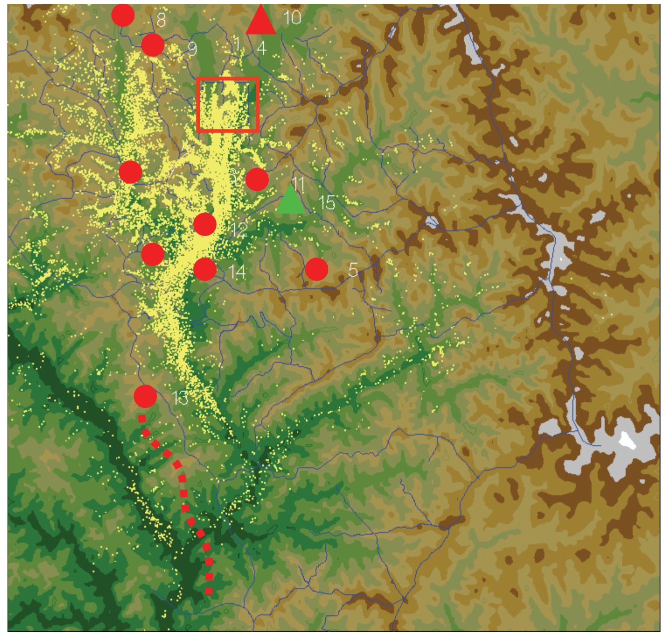

Figure 18.—PB-Piedmont-simulated smoke at 0800 EST Jan. 16, 2003.

The issue of smoke/fog on the highway is a two-part problem. The first is predicting where smoke goes. The second is understanding the dynamics of superfog – fog that reduces visibility to less than 3 m.

PB-Piedmont

Planned Burn-Piedmont (PB-P) (Achtemeier 2005) is a very high-resolution meteorological and smoke model that answers the question, "Where does smoke go at night?" (PB-P does not predict concentrations.) The model has been operational in parts of the South since 2005 and as of 2008 was modified for application anywhere in the United States. The model can be used predictively or diagnostically to simulate near-ground smoke transport at night over complex interlocking ridge-valley systems typical of landforms over much of the eastern United States.

Results from a PB-P simulation for smoke movement from a prescribed burn conducted at the Piedmont National Wildlife Preserve (PNWR) in central Georgia on Jan. 15, 2003 are shown in Figures 17 and 18. Winds were blowing from the northwest during the burn. Figure 17 shows the PB-P ground-level smoke plume (yellow dots) superimposed on local terrain and drifting southeastward from the burn site (rectangle) at 1800 EST. The difference between highest elevation (white areas) and lowest elevations (dark green) is approximately 100 m. The size of the area shown is a square 12.6 km on a side.

Weather patterns changed during the night of Jan. 15-16 so that by morning, winds had shifted to blow from the southeast. The combination of southeast winds and drainage flows created the complex distribution of smoke shown in the PB-P simulation for 0800 EST (Fig. 18). Burn crews collected observations of smoke as they drove over roads surrounding the burn site. Red circles indicate where PB-P correctly located smoke. Triangles (red-smoke; green-no smoke) show where the model failed.

Figure 19 shows how PB-P performed for 20 prescribed burns at the PNWR. Field crews took observations of smoke at various times during the night after the burns. They reported smoke by following subjective criteria: no smoke, smoke smell, smoke haze, very light smoke, light smoke, moderate smoke, and dense smoke. These reports were combined into "smoke" and "no smoke" categories and were compared with smoke maps generated by PB-P. If smoke (yellow dots) simulated by PB-P was located close to an observation, that point was designated a "hit" (see Fig. 18). Smoke hits and misses and no-smoke hits and misses were tabulated into a smoke matrix shown

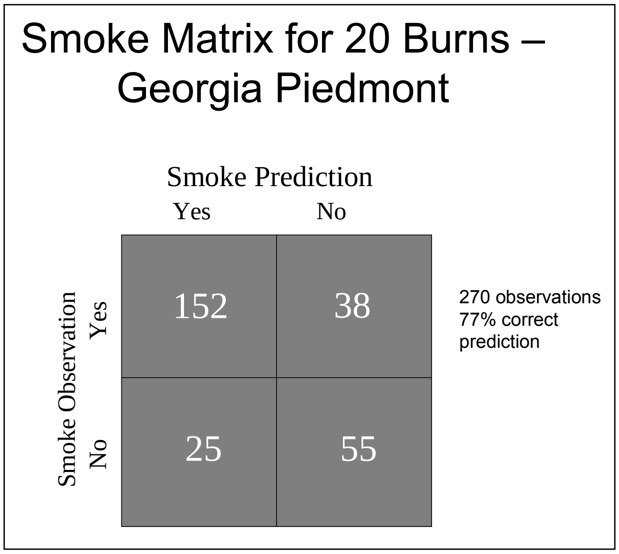

Figure 19.—Summary table for PB-Piedmont validation.

Figure 20.—Superfog over Interstate 4 on the morning of January 9, 2008. The plume of black smoke is from burning tractor trailers. (Photo courtesy of the local media.)

in Figure 19. These results show a 77-percent correct prediction by PB-P.

Superfog

Kunkel (1984) and Kokkola et al. (2003) have shown that heavily polluted conditions can favor the formation of dense radiation fogs consisting of large numbers of relatively small droplets. Pollutants act in two ways to decrease visibility: 1) by increasing the number of particles, which increases the extinction coefficient for a given liquid water content (LWC); and 2) by decreasing droplet size, which decreases mean terminal velocity and thus minimizes the fallout of liquid water.

Natural fog LWC seldom reaches levels large enough to sustain superfog. Additional moisture is required for the formation of superfog. Potter (2005) argued that water released during combustion can lead to the development of clouds, even precipitating cumulus above some wildfires. Water is a product of combustion. For 6.7 metric tons of fuel consumed per hectare, the chemistry of combustion yields 1.4 metric tons of water. To this, add in water evaporated from preheated fuels that are consumed and water evaporated from the ground and fuels not consumed. If the water equivalent released from these sources is just 0.025 cm, then the additional water released per hectare is 1.0 metric ton, for a total of 2.4 metric tons. Thus, smoke moisture can be a significant source of water for the formation of superfog.

Another factor is how smoke moisture is added to the atmosphere. Achtemeier (2006) showed that shallow layers of air passing slowly over smoldering combustion may accumulate more water per unit mass of air than do large masses of air processed through flaming combustion. The result is that smoke moisture excesses calculated from smoldering fuels can be 10 times larger than moisture excesses for flaming combustion.

It is the nongradient mixing (Gerber 1991) of two masses of humid air of widely different temperatures that makes superfog possible. Achtemeier (2008) developed a superfog model that takes into account mixing and latent heat released in the fog. If the ambient air is less humid, smoke may flash into superfog initially but then dissipate by evaporation through additional mixing, leaving a plume of particulate matter to disperse. But if the ambient air is near saturation (which is most likely just before sunrise), smoke may flash into superfog and remain superfog as the weight of suspended liquid water pulls the plume to the ground.

Figure 20 shows superfog over Interstate 4 on the morning of Jan. 9, 2008. Invisible in this image are a six-lane expressway, 70 vehicles, and several burning tractor-trailors. This fog was so dense during the night that motorists driving into it could not see the light reflected from their own headlights. The fog was described as a "black wall."

The impact of residual smoke on roadway visibility at night remains one of the most difficult smoke hazard prediction problems for the land manager. First, PB-Piedmont must provide accurate predictions of smoke transport within drainage flows over complex terrain. Second, the superfog model must accurately identify "flashpoint" temperatures—temperatures at which superfog forms rapidly. There are accounts of land managers who drove over the roads surrounding their burns into the wee hours of the morning and saw nothing save for some smoldering. Accordingly, they went home--only to learn that an accident had occurred in dense fog an hour later.

SUMMARY

The impact of smoke from agricultural and forest burning on air quality has been a major research topic for many decades not only in the United States but in many other places around the world. During this period many methods to simulate and predict the transport and dispersion of smoke have been developed. Some of these methods have never been written up; other methods languish, forgotten in obscure reports.

In this paper I have focused on those smoke models that have had, are having, and will have impacts on smoke management for eastern forests. The caveat is that the list includes those models I am aware of. There may be some excellent models I have not described.

Of the smoke models I have described, only VSMOKE is complete and operational. BlueSky and PB-Piedmont are scientific/application models that will undergo continuous modification as the science contributing to the various components improves. Both are operational, however, and can add value to a smoke management program. HYSPLIT is operational but needs a plume rise component to make it fully applicable for prescribed fire. The 4S is still under development.

Significant gaps remain. VSMOKE, designed to predict local smoke during daytime from a single prescribed burn, is limited by its assumptions. BlueSky is designed for regional-scale air quality. PB-Piedmont is designed to track smoke on the ground at night over a limited range of topographic conditions.

Plans call for Daysmoke to replace VSMOKE. Daysmoke removes the assumptions that limit VSMOKE. Furthermore, Daysmoke has shown the importance of multiple-core updraft plumes in the vertical transport of smoke. Neverthless, Daysmoke offers no pathways to predicting multiple-core updraft structures. This problem is the subject of ongoing research.

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MODELING FIRE AND OTHER DISTURBANCE PROCESSES USING LANDIS

Stephen R. Shifley[1], Jian Yang[2], and Hong He[3]

[1]Research Forester, U.S. Forest Service, Columbia, MO
[2]Post-doctoral Research Associate, University of Nevada-Reno, Reno, NV
[3]Associate Professor, University of Missouri, Columbia, MO
SRS is corresponding author; to contact, call (573) 875-5341 or email at sshifley@fs.fed.us

Abstract.—LANDIS is a landscape decision support tool that models spatial relationships to help managers and planners examine the large-scale, long-term, cumulative effects of succession, harvesting, wildfire, prescribed fire, insects, and disease. It can operate on forest landscapes from a few thousand to a few million acres in extent. Fire modeling capabilities in LANDIS are detailed, but intuitive. Modeled fires kill trees based on the fire intensity and each tree species' fire tolerance, and spatially explicit ignition probability maps can be incorporated but are not required. As the LANDIS model runs through many annual or 10-year iterations, it illustrates how and where forest vegetation is expected to change in response to succession, fire, harvesting, and other disturbances. LANDIS output can be mapped, summarized, and linked to other attributes of interest, such as wildlife habitat suitability. Although it is possible to run the LANDIS model using generic or default values, the real benefits come when the model is calibrated to reflect the unique conditions associated with a specific forest ecosystem. Applications of LANDIS include analyses of fire regimes, separately or in combination with harvesting, on the Mark Twain National Forest (Missouri), the Hoosier National Forest (Indiana), and the Chequamegon-Nicolet National Forest (Wisconsin). These analyses can guide the selection of long-term management alternatives for forested landscapes. LANDIS has also proven useful in more theoretical investigations that compare long-term effects of alternative fire regimes on the expected direction and rate of tree species composition change across an array of ecological land types.

INTRODUCTION

Comprehensive forest management requires consideration of the long-term, large-scale, cumulative effects of management activities and natural disturbances on forest commodities, amenities, and services, which include forest products (type and quantity over time), wildlife habitat (quality by species over time), water quality (usually addressed through implementation of best management practices), and biodiversity (diversity of plant and animal species, diversity of forest age and size structure, and diversity of habitats over time). Recently, increasing emphasis also has been placed on managing forests for ecological services such as carbon sequestration.

Foresters are skilled at forecasting the effects of management on products, forest size structure, and tree species composition at the stand scale. But keeping track of those details for thousands or hundreds of thousands of acres requires a landscape decision support system, which is usually in the form of a landscape computer simulation model. There are many forest landscape models available. They vary in the details of how they operate, but they all provide a means to forecast and display expected changes in forest conditions across landscapes in response to management and natural disturbances caused by wildfire, wind, insects, and disease. Forest landscape models are especially useful for forecasting expected impacts of wildfires, which, unlike harvests, are not constrained by stand boundaries, and from year to year vary considerably in location, extent, and severity. Modeling fire effects using a landscape decision support system provides a mechanism to examine average tendencies, expected variation over time, and potential impacts of fire mitigation strategies.

Examples of forest landscape models include the Forest Vegetation Simulator (Dixon 2003, USDA Forest Service 2008), HARVEST (Gustafson and Rasmussen 2005), the Landscape Management System (University of Washington 2008), LANDSUM (Keane et al. 1997, 2002), SIMPPLLE (Chew et al. 2007), and VDDT/

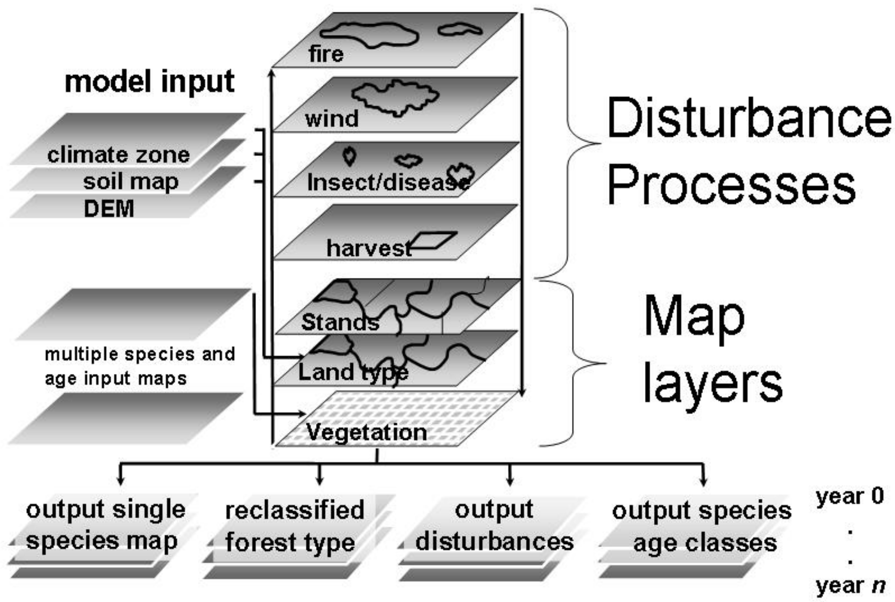

Figure 1.—Schematic of LANDIS design. This diagram illustrates the map and data layers used for LANDIS input, processing, and output. Succession and the other disturbance processes shown alter the vegetation map layer for each time period of a modeled scenario. Additional detail is available from He et al. (2005).

TELSA (Beukema and Kurz 1998). They all differ in the details of how they operate and in their relative strengths and weaknesses (Barrett 2001, Keane et al. 2004). Our experience has been with the LANDIS forest landscape model (He and Mladenoff 1999, He et al. 2005), which has been in use for about 15 years with ongoing development and applications in the United States, Canada, and Europe.

The following sections draw from previous applications of LANDIS to Midwestern forest landscapes to illustrate how LANDIS can be used to model fire effects as part of a comprehensive analysis system that also includes the effects of alternative silvicultural practices on forest growth, species succession, and wildlife habitat.

THE LANDIS MODEL

LANDIS Design

LANDIS represents a forest landscape as a mosaic of square sites (He et al. 2005) (Fig. 1). In the jargon of geographic information systems, the sites can be referred to as rasters or pixels. Although the site size can be set by the model user, we typically use sites that are about a quarter-acre (30 m on each side) or about 0.025 acre (10 m on each side). A 0.025-acre site corresponds to roughly the canopy size of a mature hardwood tree, so it is possible to create highly detailed representations of forest landscapes composed of millions of adjacent sites. LANDIS projections are made on a 1-year or a 10-year time step (iteration), and LANDIS will produce maps and summary statistics of forest conditions at each time step of a projection that may cover a few decades or a few centuries.

At each site, LANDIS tracks which tree species are present by age class. In the Midwest we used up to 18 species groups to describe vegetation in LANDIS. Sites are grouped into contiguous stands that can have complex age structure and species composition. Stands are grouped into management areas that need not be contiguous, similar to National Forest management areas used in forest planning.

When forecasting forest change, LANDIS modifies trees on each site according to a set of succession rules and equations. For example, in the absence of disturbance, the trees grow older. When trees become older, they have an increasing probability of mortality. Shade-tolerant tree species can regenerate on sites with older trees, according to a set of probabilities. When trees die, they are replaced according to a set of regeneration probabilities that differ

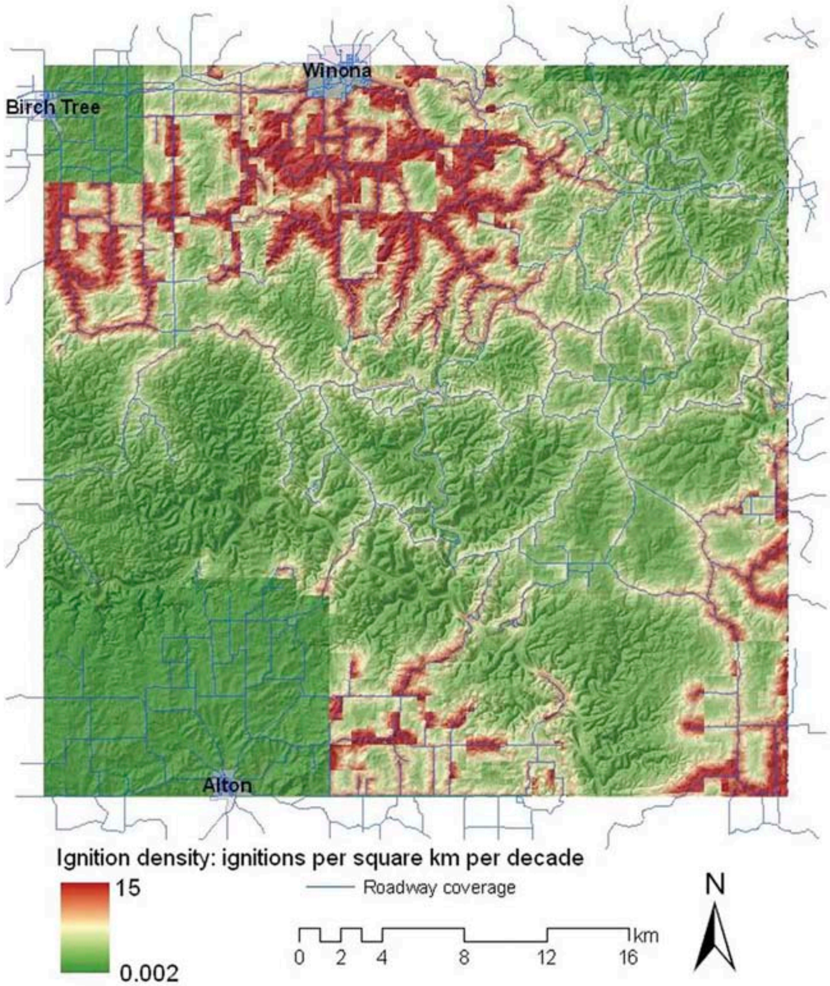

Figure 2.—Estimated probability of fire ignition based on 32 years of fire records in southern Missouri. (See also Yang et al. 2007.)

by ecological land type. Rules and probabilities can be modified to accommodate conditions for a wide array of ecosystems.

Simulated disturbances alter patterns of forest development within LANDIS. Simulated harvests can be applied to individual stands and entire management areas. Users control the type, frequency, and location of harvests based on rules that can account for stand age (e.g., oldest first), location (e.g., do not harvest adjacent stands), desired rotation length, desired species, and desired tree age or size. Windthrow is modeled according to probabilities indicating the likelihood of damaging wind events of various sizes. Usually, many small wind events and a few large ones result (Rebertus and Meier 2001), but users can modify this pattern. The location of wind events on the landscape is random, and there is greater probability of windthrow for large trees than for small trees.

Modeling Fire in LANDIS

Fire modeling in LANDIS consists of three subcomponents: fire occurrence, fire spread, and fire effect. Fire occurrences on the landscape are modeled as a two-stage process (ignition and initiation) in LANDIS (He et al. 2005, Yang et al. 2007). In the first stage, fires are ignited based on a fire ignition probability map layer. Whether an ignition can become a successful fire or not is determined by fire initiation probability in the second stage. The fire ignition probability map layer can be very simple (e.g., every location has the same probability of fire ignition), or very complex. Recent analysis of 32 years of fire records for part of the Mark Twain National Forest (Missouri) produced a detailed map showing the probability of fire ignition for each location (Fig. 2). People cause about 98 percent of fires in that region, with 75 percent of fires due to arson. The analysis of spatial patterns showed the greatest probability of fire ignition is along roads through lands owned by the Mark Twain National Forest and near local communities (Yang et al. 2007). Similar methods can be applied to other regions with long fire records. Alternatively, fire ignition probabilities for other ecosystems could be estimated by experienced observers using simple relationships based on distance from roads, ecological land type, elevation, or any other factor believed to be associated with local ignition patterns.

The quality and accessibility of historical fire ignition data is improving to the point where developing ignition probability maps for large areas is possible. Once developed, such maps are useful for many years and for purposes beyond the application of LANDIS (e.g., for distribution of fire suppression resources or communication with the public).

After an initiation is simulated, the simulated fire spreads across the modeled landscape based on fuel load, topography, and prevailing wind direction. Rates of fire spread across a site and into an adjacent site are computed using the relationships described in FARSITE (Finney 1998) and BEHAVE fire models (Anderson 1982, Andrews et al. 2005). By computing and storing the rate of fire spread for all combinations of fuel class, slope class, and wind speed for a given landscape, LANDIS can efficiently model detailed patterns of fire spread. Fires modeled in LANDIS spread until they reach a specified area or a specified time to suppression, either of which can be based on local experience when such information is available.

Tree mortality following a fire is modeled according to a set of rules that the LANDIS user can modify. Fire intensity for each burned site is classified into five categories based on the fuel load at the time of the fire. Each tree species and age class combination is categorized by fire tolerance. The youngest trees are the most susceptible to fire-caused mortality, so for a given fire intensity the tree species with lower fire tolerance will have older trees killed than will species with higher fire tolerance. Following a modeled fire, the fuel load is adjusted to a lower value that can be set to match local observations.

When the above procedures are used to model wildfires, the timing and location of the modeled fire events are based on random draws from probability distributions. Consequently, for any two separate simulation runs, the timing and location of the modeled wildfire events will differ. However, the total area affected by wildfire for the entire landscape will be similar for both runs. This element of randomness does not work well for modeling prescribed fires, where the fire location, frequency, and intensity are specified as part of a silvicultural prescription. However, prescribed fires for specific stands and years can be simulated using the flexible set of silvicultural treatments available in LANDIS fuel module (Gustafson et al. 2000, He et al. 2005).

LANDIS APPLICATIONS

In the Midwest, we have applied LANDIS to investigate the outcomes of alternative management scenarios for parts of the Mark Twain National Forest, for the entire 180,000-acre Hoosier National Forest in Indiana, and for two Ranger Districts of the Chequamegon-Nicolet National Forest in northern Wisconsin (Shifley et al. 2006, 2008; USDA Forest Service 2006; Rittenhouse 2008; Zollner et al. 2008). National Forests, like many public forest ownerships, often have digital maps, forest inventories, wildfire records, and ecological classification systems prepared for large areas. Those are valuable assets when initializing a forest landscape for use in LANDIS. Moreover, managers of public forests have a mandate to consider the cumulative effects of their management practices on the multiple commodities, amenities, and services their forests can provide.

Changes in species composition related to alternative fire regimes applied at the landscape scale can be examined over time (Fig. 3). As expected, increased fire on the landscape pushes the anticipated species composition toward fire-adapted species. While that general result is not surprising, analysis of fire effects using LANDIS provides additional insights about rates of change in species composition and age structure over time, the effects of different ecological land types (e.g., slope, aspect, hydrology) on those changes, the spatial distribution of changes, and cumulative effects over space and time.

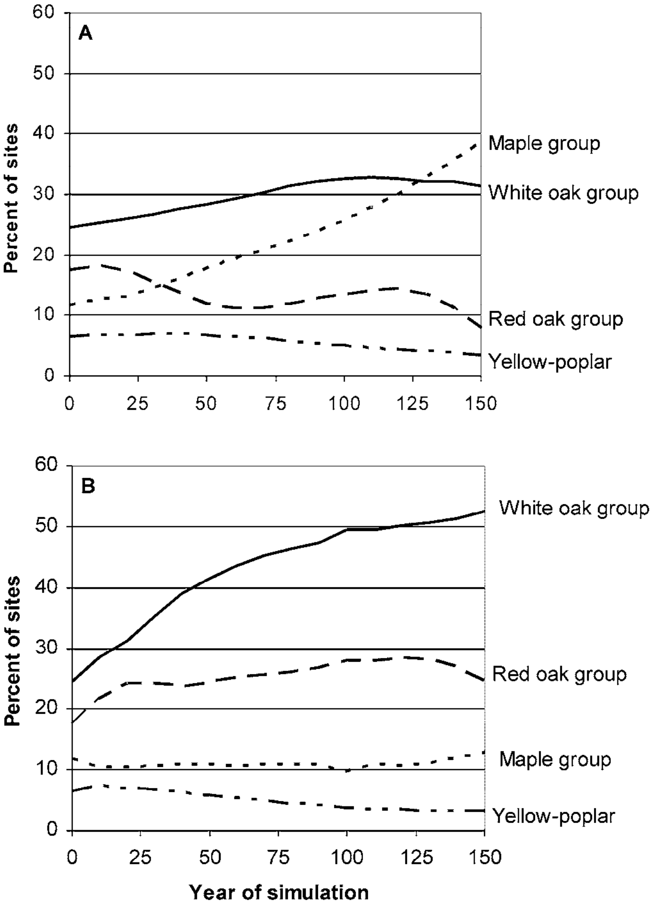

Figure 3.—LANDIS projection of the proportion of sites dominated by major species groups for two management alternatives applied to the 200,000-acre Hoosier National Forest in southern Indiana. (A) Managed with no prescribed burning and no harvest. (B) Managed with a total of 360,000 acres of prescribed burning and 52,000 acres of harvest over the 150-year projection period. (See also USDA Forest Service 2006.)

Results can be evaluated for specific landscapes, but analyses based on hypothetical landscape conditions can also provide important results to guide managers. For example, Lafon et al. (2007) created a hypothetical landscape with two elevation classes and three ecological land types representative of conditions found in the southern Appalachian Mountains. They then used LANDIS to apply two fire regimes and systematically explore the differential effects of each combination on changes in tree species composition over time. They found that when suitably calibrated, the LANDIS predictions were consistent with ecological theory. They provided evidence that (1) reintroduction of fire can reverse undesirable shifts in biodiversity; (2) species responses will differ by land type and elevation; (3) there can be interactions involving multiple factors; and (4) changes in species composition will occur gradually over one to two centuries. These results are useful to develop site-specific management plans with reasonable expectations about the likely rate of species change during restoration efforts. Other applications on real landscapes have systematically analyzed the sensitivity of LANDIS predictions to differences in fire regimes and to simplifying assumptions about seed dispersal and species establishment rates. Results for the Georgia Piedmont have shown that assumptions about seed dispersal and the effect of topographic differences on fire regimes have a relatively large effect on LANDIS forecasts (Wimberly 2004).

To investigate the long-term effects of fire suppression on central hardwood forests in the Missouri Ozarks, Shang et al. (2007) examined two management scenarios: (1) a fire suppression scenario circa 1990; and (2) a historical fire regime scenario prior to fire suppression, with a mean fire-return interval of 14 years. They found that both fuel and fire hazard increased to a medium-high level after a few decades of fire suppression. A century of fire suppression could result in more than three-quarters of the fires having medium- to high intensity levels, uncharacteristic for those Central Hardwood ecosystems. Fire suppression could also lead to distinct changes in species abundance; the pine and oak-pine forests common in the study area prior to fire suppression would be replaced by mixed-oak forests.

Forest planning is an important activity in which forest landscape models can provide valuable assistance. Using LANDIS, Zollner et al. (2008) demonstrated a way to evaluate alternative management plans and assess whether they are likely to meet the stated, multiple objectives. They predicted forest composition and landscape pattern under seven alternative forest management plans drafted for the Chequamegon-Nicolet National Forest. In most cases, the modeled results showed that multiple objectives were obtainable without conflict, but in 20 percent of the cases, land managers needed to prioritize among eight timber and wildlife management objectives. Some desired outcomes were obtainable only by mutually exclusive management activities.

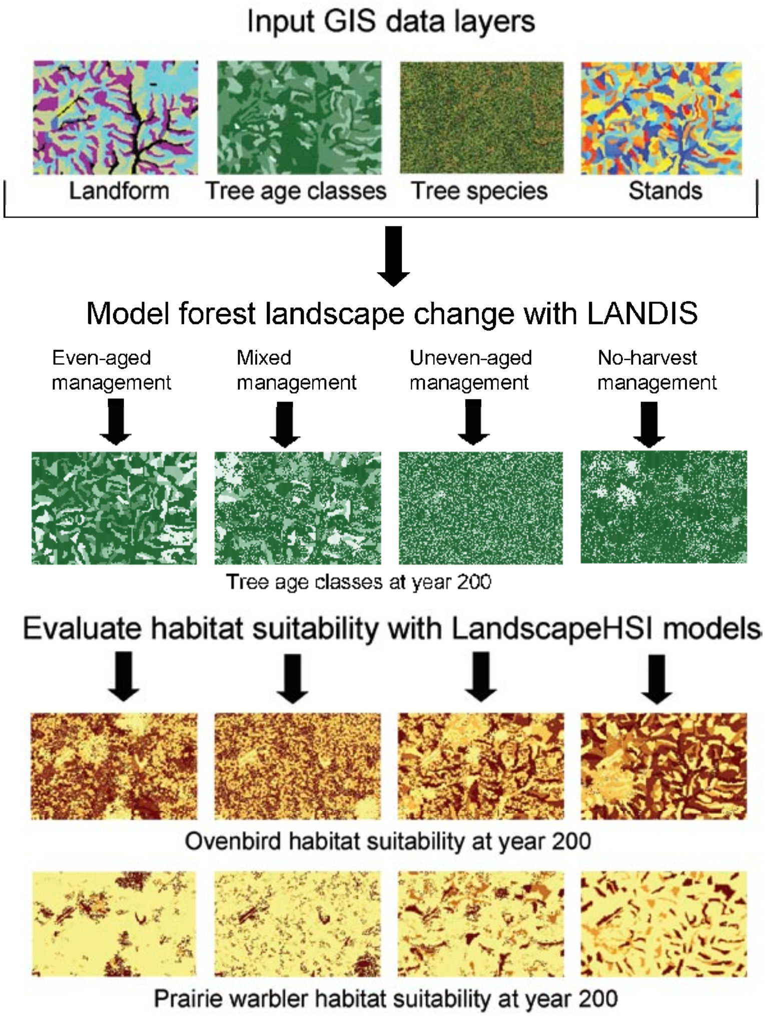

Figure 4.—Illustration of the LANDIS results for four management alternatives. Darker shades indicate older tree age classes and greater wildlife habitat suitability. Harvest practices, wind disturbance, and fire regimes affect tree species composition and tree age class, which in turn affect estimated wildlife habitat suitability. Similar maps or tabular summaries can be generated for each decade of each modeled scenario (from Shifley et al. 2008).

The predicted outcome of alternative management scenarios projected using LANDIS can be displayed graphically as well as in tabular summaries (Figs. 3 and 4). One of the most powerful attributes of LANDIS is the ability to simultaneously incorporate multiple, spatially explicit disturbance factors (fire, wind, harvest, insects, disease) and evaluate the combined effects on multiple forest attributes. Figure 4 shows a small subset of the information that can be mapped for each decade of a scenario. The even-aged management scenario illustrates the effects of regenerating 10 percent of the forest area per decade via clearcutting. The uneven-aged management scenario illustrates the impact of group selection harvesting that regenerates about 10 percent of the forest area per decade. The mixed management scenario blends those two practices to regenerate about 10 percent of the forest area per decade. In each of those scenarios the spatial distribution of vegetation age is heavily influenced by the patterns of harvest on the landscape. All scenarios included wildfires occurring with a mean fire-free interval of approximately 300 years, a value that is based on wildfire observations over the prior 30 years. The effects of wildfire are apparent in the no-harvest management scenario, where modeled fire events create the largest patches of young forest on the projected future forest landscape.

In addition to the forest characteristics illustrated in Figure 4, we can use LANDIS output to estimate, tabulate, and map habitat suitability for other wildlife species, tree species composition, harvest area, harvest volume, standing volume, snags, and coarse woody debris. Scenarios that have been modeled for the Hoosier National Forest mimic the forest management alternatives proposed as part of the formal forest-planning process. Forecasts of the cumulative effects of forest management (including prescribed fire, wildfire, and timber harvest) were factored into the selection of a preferred management alternative (USDA Forest Service 2006).

DISCUSSION AND CONCLUSIONS

LANDIS provides an extremely versatile approach for modeling the long-term, large-scale cumulative effects of forest management, disturbance, and succession on forest landscapes. Modeled landscapes can range from a few thousand to a few million acres in extent, and the approach has been used to support National Forest Planning (USDA Forest Service 2006, Rittenhouse 2008, Zollner et al. 2008).

The procedures for simulating fire ignition and spread and modeling fire effects on fuel and vegetation have been greatly expanded in the most recent revision of LANDIS (version 4.0, Yang et al. 2004, He et al. 2005). Fire spread algorithms are consistent with established fire behavior models (Anderson 1982, Finney 1998, Andrews et al. 2005) and can be adapted to accommodate a wide range of fire regimes and detailed fire effects. The only drawback of having such versatility is the need to actually specify fire ignition probabilities, fuel loads, fire spread rates, fuel treatment effects, and fire effects on vegetation. The LANDIS fire routines can be readily operated using default or generic values for these factors, but that approach fails to take full advantage of the fire modeling capabilities. For most landscapes, our ability to model detailed fire ignition, spread, and responses with LANDIS exceeds our ability to locally calibrate the model with site-specific data on fire occurrence and fire effects. To date, modeling fire effects with LANDIS has been hindered primarily due to lack of detailed information on (1) the spatial distribution of current fuels; (2) fire effects by fire intensity class on tree mortality and regeneration; and (3) effects of fire and other treatments on residual fuels. Other presentations and posters from the Third Fire in Eastern Oak Forests Conference (see the remainder of this proceedings) provides reassurance that such data are being accumulated and will become available to calibrate the LANDIS fire models for specific ecosystems.

Modeling fire effects is one reason to apply LANDIS, but LANDIS also is a framework for synthesizing many forest succession and disturbance processes that also include wind, silviculture, regeneration, and land use change (e.g., Syphard et al. 2007). Results for alternative scenarios can be summarized, illustrated, and/or mapped through time to show differences in forest vegetation structure and composition, wood volume, down wood, forest fragmentation, species diversity, and wildlife habitat suitability (e.g., Dijak and Rittenhouse 2008). The ability to analyze landscape-scale forest change provides opportunities for foresters, wildlife biologists, ecologists, and planners to collaborate and examine interactions or tradeoffs of various management alternatives on a large forest landscape.

In our experience it can take many months to initialize and calibrate LANDIS for a new, large landscape. By comparison, running various alternatives through LANDIS requires a relatively short time (a few hours or a few days), and summarizing results may take a few days to a few months depending on the complexity of the attributes of interest. For example, summarizing wildlife habitat quality for multiple species is notoriously time-consuming, but some summaries of forest vegetation require only a few minutes to summarize and map. Although substantial effort can be required to calibrate LANDIS for a large landscape, we have discovered large benefits from that up-front investment. Each time we have initialized a large landscape for use with LANDIS, we have attracted new partners with new questions that we can explore collaboratively on those same landscapes.

Application of LANDIS is best pursued through a team approach that incorporates competencies in a variety of specialties such as geographic information systems, data processing, programming, forest management/silviculture, ecology, wildlife biology, fire behavior and management, timber and markets, and planning. That collaboration makes the process efficient by drawing on a diverse array of technical specialists. It also facilitates communication across disciplines, and it helps ensure that results are practical and relevant for multiple purposes.

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Martin Spetich and an anonymous reviewer provided comments and suggestions that helped us improve an earlier version of this manuscript. William Dijak, Frank Thompson, Joseph Fan, Chadwick Rittenhouse, Michael Larson, and Josh Millspaugh helped develop material used in some of the examples used in the paper. We thank them all.

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FIRE AND FOREST MANAGEMENT

BUILDING A STATE PRESCRIBED FIRE PROGRAM: EXPERIENCES AND LESSONS LEARNED IN OHIO

Michael Bowden[1]

[1]Fire Supervisor, Ohio Department of Natural Resources, Division of Forestry, Columbus, OH
To contact, call (614) 265-1088 or email at michael.bowden@dnr.state.oh.us

Abstract.—Science continues to show the need for the use of prescribed fire in oak-dominated forests of the eastern United States. Fire is necessary to reduce competition by thin-barked species, allowing for the development of oak advance regeneration. Many agencies are beginning to apply this science by using prescribed fire to manage their oak forests. This paper examines the development of the Ohio Division of Forestry's prescribed fire management program. It outlines the experiences of the Division as it increased use of fire and began burning at the landscape level. Problems, challenges, and lessons learned during the growth of Ohio's prescribed fire program are addressed as well. Although this paper does not provide all of the answers for how to develop a prescribed fire program, it does use the experiences the Ohio Division of Forestry faced over the past 10 years to provide some idea of what expect.

FIRE AND OHIO'S STATE FORESTS

Ohio's forests, like many in the eastern United States, are made up various species of deciduous trees; oaks are the most abundant, especially in the southern and eastern portions of the state (Griffith et al. 1993). At time of settlement (ca. 1800) the state was estimated to be more than 95 percent forested. Settlement and western expansion during the 19th century resulted in a deforestation that lasted nearly 100 years. During the first decade of the 20th century only 10 percent of Ohio was forested. This rapid and significant deforestation spurred the state government to create a State forest agency in 1885.

One of the primary missions assigned to the newly created Division of Forestry was to address the deforestation and prevent a "timber famine." The Division worked on ways to increase forest lands in the State. In 1912 the state legislature amended the Ohio Constitution to allow for the creation of a forest reserve system. Shortly thereafter, the State began to purchase lands which became the first State forests. The lands the Division acquired had been heavily cut over, burned, grazed, and farmed during the late 1800s and early 1900s. These newly obtained lands were then protected from fire and grazing, which resulted in rapid regrowth to mostly oak-dominated stands.

Wildland fires continued to burn occasionally through these forests, but by the mid-1930s the Division's fire control efforts had dramatically decreased the acreage of forest land burned annually (Leete 1938). The decrease in wildfires allowed fire-intolerant species such as red maple and yellow poplar to move out of their traditional locations in the coves and bottomlands and begin to establish in the uplands (Hutchinson et al. 2008). Prior to the development of an effective fire suppression program, these species would be killed during wildfire events that occurred, on average, once every 3-10 years (McEwan et al. 2007). Without fire burning through these stands during the past 70 years, red maple and yellow poplar have developed into saplings and pole-sized stems, creating dense, shaded conditions in the understory, which prevents the development of oak advance regeneration (Dey and Fan, this volume).

In addition to maintaining forest reserves, or State Forests, the Division has a mandate to "take such measures as are necessary to bring about a profitable growth of timber." The Division utilizes many "measures" to address this charge, including harvesting and timber stand improvement practices. The high density of red maple and lack of oak advance regeneration (large seedlings and saplings) has created problems for foresters trying to regenerate oak and sustain its dominance in our State forests. Without

Figure 1.—Ohio Division of Forestry crew conducting a burn in 2005 on the Fire and Fire Surrogates Study.

advance oak regeneration present, harvests or significant disturbances (e.g., tornadoes, ice storms, or insect outbreaks) result in stands that regenerate with an oak component greatly reduced from the previous stand.

Studies throughout the eastern United States have brought to light the changes that are occurring in oak-dominated forests due to the lack of fire; this research stresses the important role that fire plays in maintaining the oak component in these ecosystems (Abrams 1992, Brose and Van Lear 1998, Dey and Fan, this volume). The Ohio Division of Forestry has been involved with this research through its partnership with the U.S. Forest Service, Northern Research Station. Northern Research Station scientists based in Delaware, OH, have led the efforts to study the interaction of fire and fire surrogates (i.e., thinning and harvesting) on the oak-dominated forests in Ohio (Yaussy and Waldrop, this volume). Two of the three replications of this long-term national study are located on Ohio State Forests, Tar Hollow in Ross County and Zaleski in Vinton County (Fig. 1).

The rapid increase in number of scientific publications suggesting the need for fire to sustain oak-dominated ecosystems was a catalyst for the Division of Forestry to expand its use of prescribed fire during the late 1990s and early 21st century, from small sites to larger landscape applications (Fig. 2). This increase in the use of prescribed fire has pointed out issues that fire managers need to be ready to address when building or increasing a prescribed fire program.

The Division of Forestry had been involved with fire suppression for more than 70 years and had been using prescribed fire for nearly 20 years prior to its expansion to the landscape level. Differences between the Division's traditional practice of fire suppression and the larger-scale application of prescribed fire quickly became apparent. The goal of this paper is to share the experiences of the Division of Forestry during the expansion of its prescribed fire program.

GROWING A PRESCRIBED FIRE PROGRAM

Preliminary Considerations

When beginning to grow a prescribed fire program, the natural resource manager should consider several things before taking action. You should identify the goals of using prescribed fire. Burning for ecosystem process restoration is different from burning to reduce hazardous fuels and will require different tools, training, and application. In addition, you need to consider how much land you envision treating annually and compare

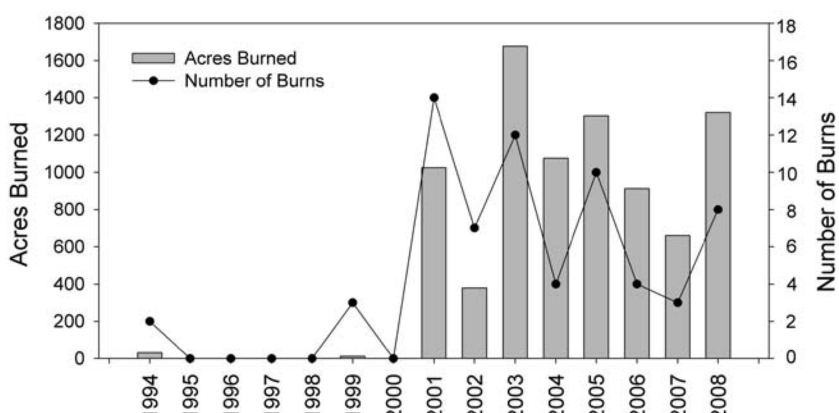

Figure 2.—Prescribed fire application by the Ohio Division of Forestry, 1994 - 2008.

that amount with your current levels of treatment. You should identify the situations, such as weather conditions, season of burning, and fuel moistures, in which you can treat the landscape with prescribed fire. What barriers may impact your implementation of prescribed burning? Can these barriers be mitigated or overcome? Finally, do you have opportunities to establish partnerships with other agencies or organizations to increase capacity to burn more acreage? Serious consideration of these issues can reduce "growing pains" and speed the growth of the program.

During the growth of our prescribed fire program, we have realized the importance of making sure that all involved, from leadership to firefighters, understand its goals. A lack of understanding results in application of fire that does not meet the objectives. Additionally, a workforce with a long history of fire suppression may not understand the goals of prescribed burning, will not fully embrace the management technique, and will be less willing to implement it.

As a program is being expanded, I recommend starting small and gradually increasing capacity. This stepwise implementation allows workers to increase their skills gradually, gives all participants an opportunity to identify practices that are efficient and effective, and prevents workers from landing in situations they are not prepared for or able to deal with. This approach will result in a more effective prescribed fire program in the long run, as recovery from a "misstep" when growing too fast is a difficult process.

Public Perception and Education

Public perception of prescribed fire will be an issue when your organization is developing and growing a prescribed fire program. It is critical to recognize the public's role and to be proactive in sharing information about the program. Educational materials such as Websites, handouts, and displays explaining what and why you are conducting burns are important tools to prepare in advance of program growth. You should plan on spending time talking with local, state, and federal elected officials about the importance and need for prescribed fire, as constituents will contact their elected representatives about your burning efforts.

As prescribed fire programs grow, they often encounter vocal minority groups that oppose the use of fire as a management tool. These opposition groups can provide a serious challenge to a program. They may base arguments on emotion rather than science, and will utilize websites, letters to the editor, and petitions to promote their viewpoint. Many of the opposing arguments focus on potential short-term impacts to plants and animals and do not consider the long-term benefits to habitat that result from the application of prescribed fire.

The majority of the general public will remain silent on the issue, even if these individuals support your efforts. However, you should expect people to speak up if a burn impacts them or others they care about, as in the case of a close-to-home fire or one that creates a smoke problem. When addressing these concerns, make sure you take time to listen and understand the concern. Frame your response to address the concern and stress the importance of the prescribed fire as a management tool. If there are ways to mitigate the concern in the future, be sure to acknowledge that you will change your practice to accommodate the concern. Not addressing an individual concern may result in the formation of a new vocal minority group that opposes prescribed fire.

As a prescribed fire program grows, it is important to recognize that just because there had been no public opposition to burning in one or more areas, this silent acquiescence may not hold true as burning begins in new areas. Additionally, as the use of fire increases, there will be a corresponding need for increasing efforts to reach out and educate the public.

Questions will also come from the public, as well as within agencies, on the question of why there is a shift from fire suppression to prescribed fire. When addressing these questions, as well as other questions on the use of fire, be sure that if you say something is "science-based," you have the support of science and scientists. If you are using adaptive management to make your decisions, say that as well. Finally, when dealing with the public and media, you or other fire management personnel may say something that can be taken out of context, portraying your agency in a bad light. It's important to think how something you might say could be used in a story or on a website.

IMPLEMENTATION: METHODS AND CHALLENGES

As a prescribed fire program grows, the complexity of burns will increase, as will the need for additional resources. Along with these changes comes the challenge of positioning personnel to perform successfully through training. Training of prescribed fire personnel should focus on both the operational level and the "state of the science." Understanding the science will better enable your personnel to meet the burning objectives because they will understand the benefits of prescribed fire more fully.

It is important to recognize that a workforce that has extensive fire suppression training and experience may lack the knowledge, skills, and abilities to perform prescribed burns that meet land management objectives. Tactically, the application of fire, or "burning out," during fire suppression is commonly the approach that experienced firefighters utilize when implementing prescribed burns. In most cases, burning out is rapidly performed in order to secure the control lines by removing the fuel ahead of the main fire. Many times use of the standard burning-out methods results in too much heat and negative fire effects (e.g., damage to the stand). Training on firing methods such as flanking fire, dot firing, and chevron patterns is helpful in addressing this issue.

A key point to emphasize to the workforce is that prescribed burns are conducted to achieve management objectives, not to achieve black acres. To achieve management objectives, prescribed fire programs may need to look at using different burning practices, methods, and tools.

Equipment needs are different for prescribed fire than for fire suppression. Instead of emphasis on equipment that can put out fire, large-scale prescribed fires require emphasis on equipment that can light fire. Efficient and effective ignition of prescribed burns requires the need for more and specialized equipment. Drip torches are the workhorse of any prescribed fire program. It is important to have a lot of these in your equipment cache. Prepositioning a large stock of torches throughout your burn unit will reduce the time spent waiting for torches to be refilled. Having more drip torches available also increases the number of igniters you can use on a burn. Having too few torches often is the limiting factor for completing the burn efficiently. In addition to drip torches, standard operating procedures should require all igniters to carry several fusees, which can be used to keep the operation moving when refueling is not possible.

Other useful firing devices are flare guns, Terra Torch®, torches mounted on all-terrain vehicles (ATVs), and plastic sphere dispensers. Flare guns and plastic sphere dispensers are used to facilitate interior ignition. Flare guns can launch a highly flammable flare approximately 100 yards to ignite inaccessible areas. Ground-based plastic sphere launchers are also used to facilitate interior ignition. These handheld launchers perform the same function as aerial-mounted plastic sphere dispensers (see below), injecting spheres filled with potassium permanganate with ethylene glycol to create an exothermic chemical reaction; instead of the ball falling from the aircraft to the ground, the ball is launched into the fuel bed. These devices are very useful when using the spot ignition technique on smaller units and when igniting areas that are difficult to reach with other ground-based tools (e.g., locations across waterways, areas of dense fuel concentrations).

ATV-mounted torches are great tools to use when firing on flat ground or where there are roads and trails that are ATV-accessible. These torches expedite blacklining and lighting from roads and trails. If the burn unit has gentle terrain and is fairly open, these devices can be used to perform interior ignition as well. The operator using an ATV to perform interior ignition needs to be an experienced ATV operator as well as an experienced igniter. In many incidents during the past several years, ATV torch operators have been trapped inside burn units because of ATV operation error or lack of good judgment on fire behavior.

Figure 3.—The Terra Torch® in use on a Division of Forestry prescribed fire.

Terra Torches® are very useful when burning larger fuels or for burning during times of higher fuel moistures (Fig 3). These devices can be used to blackline prescribed burn units due to the long residence time of the gelled fuel. These units are also helpful in ignition on roadsides and can blacken edges rapidly. The Terra Torch® requires access to both gasoline and diesel fuel in large quantities, gelling powder, and a means of transportation for the torch.

Aerial mounted plastic sphere dispensers (PSD) should be considered when burning forested areas more than 200 acres in size. The use of a PSD machine allows the prescribed burn manager to spread fire across the unit very quickly. The fire intensity can be managed through the spacing of the ignition flight lines as well as the distance between each sphere within a given line. Each sphere creates a single ignition point and hence an individual fire. The spacing of these individual fires can be used to manipulate how much of the unit is burned with heading, backing, and flanking fire intensity. The use of an aerial PSD machine has specific requirements that may be challenging for some agencies. To use aerial ignition, the prescribed burn manager needs access to a helicopter, a PSD machine, and personnel trained in the operation of the equipment. Although the cost of a helicopter is generally in the range of $400-$1000 per hour, the use of aerial ignition can dramatically reduce prescribed fire costs, especially on larger units, due to the reduction in time and personnel needed to complete the burn.

Other tools include ATVs, water units, and portable water tanks. ATVs are very useful for patrolling, monitoring, and moving supplies throughout your burning area. It is important that you use trained and skilled operators that recognize where the ATVs can travel safely. Prescribed fire managers should not push the limitations of the equipment and operators by asking them to go places that are not safe. Water units such as engines and other utility vehicles with water tanks and pumps are very useful during holding and mop-up operations. When water units are used, portable water tanks strategically placed throughout your burning area will be very helpful in providing refill locations.

As a prescribed fire management program grows, there will be successful days and days that were not as successful. It is critical to document what you did, when you did it, why you did it, and what results you got. Without these data, it will be difficult to determine what works well and more importantly, how to replicate it. When determining the effects of a burn, you need to be patient: It often takes time for those effects to become obvious. When burning, you may need to find out what fire intensity is “too hot” and what is “too cool” for your objectives and try to burn at an appropriate intermediate intensity.

After-action reviews (AAR) should be established as a standard operating procedure. These reviews can be implemented in many ways without taking too much time. The AAR should be done immediately after the burn has finished, while the crew still has the events of the day in mind. Items that come out of these reviews may seem minor, but oftentimes it is the small details that get overlooked in the planning process. Paying attention to these details can greatly improve the overall performance of the burning group. The use of AARs has caused the Division of Forestry to place greater emphasis on logistical support and planning, such as prepositioned drip torches, better communications, and better availability of supplies. Periodic program reviews are also helpful for looking at organizational issues that may need adjustment.

As a prescribed fire program grows, it is important to recognize that your people will need to grow as well.

Utilize the most skilled prescribed fire personnel in the most critical roles and locations during burns. Make sure that they have someone with them who has a mentor and is being nurtured for the future. It is important to establish partnerships with other agencies and organizations that conduct burns. It can be very educational for your personnel to work with these other groups to provide a different training experience. In addition, helping out on others' burns is beneficial as they will be more likely to reciprocate.

As a program burns more acreage, it is important to stay abreast of air quality issues. Air quality regulations are developed nationally and enforced locally. It is also important to get to know your local air quality regulators and to have them observe burns. Once they observe what you are doing with fire and understand that you know what you are doing, they will be more likely to work with you. Many states' air quality regulations are lumped into the general open burning regulations, and showing the regulators how prescribed fire is different from general open burning is important as well. In areas where smoke is a concern, the air quality agency may be able to place monitors in these locations to document background air quality as well as impacts on the air quality from prescribed burns. Smoke management training and research carried out during prescribed burns will be critical to ensure that smoke does not impact the public in undesirable ways and that burning can be allowed to continue.

Most importantly, safe burning practices must be stressed as programs grow. Escaped prescribed fire could result in legislative action that would limit or end the use of this important land management tool.

LESSONS LEARNED DURING IMPLEMENTATION

Fire Behavior and Fire Season

An early lesson that we learned was that the conditions on days the Division typically thought of as "fire days" are not the most conducive to prescribed burns. The fuel and weather conditions on "fire days" are often too dry, resulting in intense fire behavior and mortality of larger trees, which is not our objective.

The development of burn prescriptions is an art based upon science. When developing prescriptions, you should be careful not to prescribe yourself out of the opportunity to burn. Prescriptions should be developed to allow for burning on almost any day that fire will carry across the landscape.

The number of days since rain is more critical in the early spring than the late spring when vegetation is pulling moisture out of the soil; a longer rain-free period is necessary in early March as compared to mid-April for leaf litter to be dry enough to burn. During the fall, burning is successful under higher relative humidities than in the spring because autumn leaf litter is less compacted. During the spring, when the litter has been compacted over the winter, only the top layer of litter burns unless the site has experienced an extended rain-free period, generally 5 or more days.

In our region, it is very difficult to manage fire behavior on very dry sites that have not had fire exposure in several years, especially those with dense thickets of greenbriar (*Smilax rotundifolia)*. These sites burn with high intensity, regardless of the type of fire (heading, backing, or flanking), and some overstory mortality often occurs. Conversely, fire may carry across mesic sites, but with very low intensity and limited effects. Observations of fire behavior on sites that had a mixture of oak (dry sites) and yellow-poplar/red maple (mesic or wet sites) stands illustrate this difference. Fire behavior in the oak stands was moderately intense (3-ft flame lengths) while the fire behavior in the yellow poplar/red maple areas was minimal and in many places the fire went out (Fig. 4)

Early growing-season burns may be the "wave of the future" and should be considered. Research has indicated that burning during this season can provide the greatest benefit to oak regeneration by having a larger negative impact on competing species such as red maple and yellow-poplar (Brose and Van Lear 1998). Early growing-season fires are much easier to control because they have lower flame lengths, lower rates of spread, and reduced potential for escape due to lower probability of ignition (Fig. 5). Growing-season burning also extends the spring fire season by 3 to 6 weeks. In Ohio, green-up begins in mid-April, the traditional end of the prescribed fire

Figure 4.—Patchy fire coverage on a site dominated by non-oak species.

Figure 5.—An early growing-season burn conducted at the Vinton Furnace Experimental Forest, May 6, 2008.

season; burning throughout the early growing season extends the fire season until mid-May. However, early growing-season burns raise questions and concerns related to potential negative impacts on animals such as turtles, snakes, and ground-nesting birds. More research needs to be conducted to determine whether short-term negative impacts (if any) are outweighed by long-term positive changes to habitat.

Operationally, growing-season burns will be different in several ways. The prescribed fire crew is exposed to more physically taxing conditions due to higher air temperatures. During the typical dormant burning season, it is unusual to experience air temperatures over 75 °F, but higher temperatures will be common during the early growing season. Early growing-season burns will also have reduced visibility across the unit due to the leaves on the vegetation, reducing the ability to see people in the unit and spot fires outside of the unit.

Fire Effects

Many prescribed burns are being conducted in oak-dominated forests to reduce the competition to oak regeneration by red maple and/or other thin-barked trees. A single burn will topkill red maple stems, but abundant resprouting will follow. To reduce this competition, more than one burn will be needed to be successful. Our experience is that fire effects, both desirable and undesirable, take at least 2 years to become evident. It is important to continue to monitor sites for several years after burning to detect these changes.

In general, we observe that burning is reducing the low shade and topkilling the fire-intolerant species. However, if promotion of oak advance regeneration is a management goal, canopy opening will likely be required to provide more light to the oak seedlings. In locations where we have burned and opened the canopy, we are beginning to see the development of vigorous oak advance regeneration (Fig. 6).

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Figure 6.—Abundant white oak advance regeneration on a burned site in southern Ohio. This site had been burned five times from 1996 to 2004, largely eliminating the sapling layer of shade-tolerant species. Here, white oak seedlings are growing rapidly in a gap caused by the death of several dominant and codominant white oaks, part of a regional white oak decline event.

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LANDSCAPE-SCALE FIRE RESTORATION ON THE BIG PINEY RANGER DISTRICT IN THE OZARK HIGHLANDS OF ARKANSAS

John Andre[1], McRee Anderson[2], Douglas Zollner[3], Marie Melnechuk[3], and Theo Witsell[3]

[1]Ecologist (retired), Big Piney Ranger District, Ozark-St. Francis National Forest, Hector, AR

[2]Fire Restoration Project Manager, Interior Highlands Office, The Nature Conservancy, Jasper, AR

[3]Director of Conservation (DZ), and Ecologist (MM), The Nature Conservancy, Little Rock, AR

[4]Botanist, Arkansas Natural Heritage Commission, Little Rock, AR

JA is corresponding author: to contact call (870) 496-2445 or email at jandre@hughes.net

Abstract.—The Ozark-St. Francis National Forest, The Nature Conservancy (TNC), the Arkansas Natural Heritage Commission, Arkansas Forestry Commission, private landowners, and others are currently engaged in a collaborative project to restore the oak-hickory and pine-oak ecosystems of the Ozark Highlands on 60,000 acres of the Big Piney Ranger District. Frequent historical fires and native grazers (e.g., elk and bison) maintained an open canopy of woodlands and savannas, where oaks, hickories, and shortleaf pine dominated. Today, these former woodlands now contain "forest-like" closed canopies (dominated by oak-hickory, oak-shortleaf pine, and shortleaf pine) caused by changes in land-use practices, including fire suppression, domestic livestock grazing, and silviculture. Historic records indicate that pre-European settlement Ozark woodlands averaged 38-76 trees per acre. Current densities average 300-1,000 stems per acre. Oak decline and the red oak borer have affected at least 300,000 acres of the Ozark National Forest and about 1.5 million acres of the Ozark Highlands. The Big Piney Ranger District is implementing a long-term, landscape-scale ecosystem restoration project in a TNC conservation priority area. Specific project activities include the application of periodic prescribed fire and thinning by commercial and noncommercial methods. The goals of these activities are to improve forest health, restore fire-dependent woodland ecosystems, and provide for safety by reducing hazardous fuels in the wildland/urban interface. A comprehensive monitoring program documenting ecosystem responses to prescribed fire and woodland thinning treatments is currently being implemented. The monitoring program includes tracking changes in overstory, understory, and herbaceous plant species, the structure of plant communities, fuel loads, nonnative invasive species, and breeding and wintering bird communities.

INTRODUCTION

Fire has been a dominant force in establishing and maintaining woodland ecosystems throughout North America (Pyne 1982). Forest health issues in the Ozark Mountains of Arkansas mirror changes occurring in other forest ecosystems throughout the country. Altered fire regimes and past land management activities have led to a significant increase in fuel loads and tree densities, a shift in species composition, and a decline in forest health. Currently, there are substantially more degraded woodlands with closed canopies (oak-hickory, oak-pine, and shortleaf pine [*Pinus echinata* Mill.]), and less woodlands (oak-pine woodland, oak-hickory woodland, and shortleaf pine woodland) than occurred historically under a more frequent fire regime (Foti 2004). The frequent, low-intensity fire regime under which these ecosystems evolved has been disrupted (Guyette and Spetich 2003). The woodlands of the Ozark and Ouachita Mountains were heavily logged 80-100 years ago. By the 1920s and 1930s systematic fire protection had been established on National Forest and private lands, and the fire frequency was drastically reduced (McGuire 1941, Bass 1981, Strausberg and Hough 1997, Soucy et al. 2005). Historic records indicate that presettlement tree density in the Ozark Highlands (Bailey 1998) ranged from 38-76 trees per acre (Foti 2004). Current densities in much of the region range from 300-1,000 stems per acre. Increased stand density leads to increased competition for available nutrients, sunlight, and moisture. Fralish (2004) states that in addition to densification, woodland communities are shifting from a fire-adapted oak-pine dominated community to a more fire-intolerant community of red maple (*Acer rubrum* L.), sugar maple (*Acer saccharum* Marsh.), and American beech (*Fagus grandifolia* Ehrh.). During periods of drought, trees already stressed by resource competition become more vulnerable to disease and insect attack, such as the red oak borer. Fire suppression has greatly reduced the once-common oak and pine woodlands on National Forest lands; their integrity is at risk.

Dendrochronologists and other researchers have investigated the fire history of the Ozark Highland over the last 30 years. At three sites on the Big Piney Ranger District, Guyette and Spetich (2003) reported mean fire return intervals of 5-16 years from 1680-1820, 2-3 years from 1821-1880, 2-5 years from 1881-1920, and >62 years (one study site) to 80 years (two study sites) from 1921-2000. Recent work from three additional sites on the Big Piney Ranger District revealed mean fire return intervals of 1.4-5.4 years from 1810-1920 (Guyette et al. 2006). Engbring et al. (2008) reported mean fire return intervals for a site in north-central Arkansas of 1.9 years from 1820-1900, 2.1 years from 1901-1930, and 2.6 years from 1931-2003. Reduced fire frequency beginning in the 1920s reflects the implementation of effective fire suppression and prevention programs in Arkansas (Bass 1981).

The Ozark Highlands are biologically important due to the diversity provided by a system of fire-adapted, wooded, and partially wooded herbaceous plant communities. More than 150 species of animals and plants are endemic to the Ozark Highlands. Most of these species rely in some way upon healthy oak and pine ecosystems. Many socially, economically, and ecologically important species inhabit these ecosystems, such as neotropical migratory birds, white-tailed deer, black bear, turkey, and quail.

In 2001 the U.S. Forest Service (USFS) and multiple partner groups initiated the largest ecosystem restoration project in the USFS Southern Region. The project encompasses 60,000 acres of the Big Piney Ranger District, Ozark-St. Francis National Forest, in Arkansas. The goal of the project is to restore structure, function, and ecological processes that will develop and sustain oak and pine woodland ecosystems. This paper describes project planning, including modeling and quantifying desired conditions, and implementation. The use and effectiveness of collaborative partnerships at multiple levels are also described.

RESTORATION SITE LOCATIONS

Six restoration areas, ranging from 6,000 acres to 12,500 acres, were established on the Big Piney Ranger District. At the time of establishment, the areas were in the General Forest Management Area of the 1986 Ozark-St. Francis Land and Resource Management Plan. Under the Revised Land and Resource Management Plan (USDA FS 2005), the areas are in the oak woodland and pine woodland management areas. The six sites are primarily in the Lower Boston Mountains Subsection. The location of each restoration area within this subsection represents one or more of the land type associations (LTAs) (USDA FS 1997) that occur on the District.

PROJECT DEVELOPMENT

Collaborative Partnerships

Collaborative partnerships at multiple levels have formed to address the current state of declining forest health throughout the Ozark Highlands. The ecosystem restoration project described here is supported by the Oak Ecosystem Restoration Team, which includes representatives from the Arkansas Wildlife Federation, Arkansas Game and Fish Commission, Arkansas Forestry Commission, Arkansas Natural Heritage Commission, U.S. Fish and Wildlife Service, University of Arkansas Cooperative Extension Service, The Nature Conservancy (TNC), USFS, and USFS - Southern Research Station. The Oak Ecosystem Team has selected the Big Piney Ranger District's Ecosystem Restoration Project as a demonstration site for the restoration of fire-adapted woodland ecosystems in the Ozark Highlands. The partnership uses this demonstration site to host public, private, and legislative tours and to provide information on current and desired ecological conditions and strategies for achieving ecosystem restoration objectives.

The ecosystem restoration project is also part of the Fire Learning Network (FLN), a collaborative project between the USFS, Department of the Interior, and TNC. The FLN promotes the development and testing of creative,

Figure 1.—Current condition of a degraded oak-hickory woodland.

adaptive, multi-ownership fire management strategies that are compatible with the National Fire Plan goals and TNC's conservation goals. The network strives to achieve tangible results at landscape and ecoregional scales. Projects within the network have completed ecological models, developed spatially explicit maps of current and desired future conditions, developed alternative management scenarios for ecological restoration, and developed broad-based monitoring programs to track progress toward achieving desired future conditions.

Landscape Assessment

Based on an analysis of the Government Land Office (GLO) records, Foti (2004) concluded that 67 percent of the Ozark Highlands presettlement landscape was composed of open woodlands, savanna, and prairie. On the Big Piney, open woodlands occurred most frequently on ridge tops, flats, and south-facing slopes and less frequently on upper north-facing slopes, toe slopes, and bottoms. Mid and lower north-facing slopes were more densely forested. Oak, oak-pine, and pine woodlands occurred on about 80 percent of the land area of the district during presettlement times.

In 2002 oak and pine woodlands covered less than 6 percent of the land area. The Big Piney Ranger District and its partners identified 60,000 acres in six areas (Middlefork-12,200 acres, Rotary Ann-12,500 acres, Piney-12,200 acres, Oak Mt.- 6,000 acres, Eastside-11,300 acres, Southfork- 6,000 acres) for a long-term ecosystem restoration project. Project area boundaries adjoin about 14,000 acres of private property; almost 1,100 acres of private property are inside the designated restoration areas, providing opportunities for public-private partnerships.

Ecological Models

To develop an understanding of how the ecosystems within the project area function, the project team and partners collaboratively developed and quantified conceptual ecological models using the Vegetation Dynamics Development Tool (Beukema et al. 2003). This process expanded the partners' ecological understanding and aligned alternative treatment options with the desired condition (plant community structure and species composition) envisioned for the landscape.

Quantifying and Mapping the Current Condition

We used the NatureServe (www.natureserve.org) classification system to identify seven cover types: *Oak-Hickory Closed Canopy, Oak-Hickory Woodland, Oak-Pine Closed Canopy Forest, Oak-Pine Woodland/Savanna, Shortleaf Pine Closed Canopy Forest, Shortleaf Pine Woodland/Savanna,* and *Sandstone Glade.* We used these cover types and three data sets (land type-associations, land types, and USFS Natural Resource Information

Figure 2.—Desired condition of a restored oak-hickory woodland.

System FSVeg cover types) to define and map current ecological conditions. LTAs (USDA FS 1997) were used to qualitatively map the plant communities that occur within the project area: the Mesic Morrow Mountain Uplands, Bloyd Mountain Valleys, Lower Atoka Hills and Mountains, and the Arkansas Valley Hills. Each LTA was mapped and overlaid with the current cover types derived from ground-truthed orthophoto quads and FSVeg data. A land type (subdivision of LTA) combines a suite of ecological characteristics (soil type, aspect, slope, and climate conditions) that distinguish it from other land types. We classified land types as riparian, toe slope, south slope (lower, middle, and upper), ridge top, and north slope (lower, middle, and upper). This information was projected across the project area. Current cover types were quantified for the project areas (Table 1).

Desired Conditions

Information used to develop the scientific justification for the desired condition included historical accounts, dendrochronological studies, GLO records, scientific literature, and the information inherent to the Ecological Classification System. The desired condition of each cover type was quantified, described, and delineated across the project area using geographic information systems. Cover types were mapped to depict their locations under the historical fire regime. Once this process was completed, the desired conditions were presented to the project partners in a collaborative, peer-reviewed process (FLN 2002b) to refine the product (Table 1).

PROJECT IMPLEMENTATION

We developed a 10-year implementation plan that strategically identifies five key areas crucial to the project's success: (1) prescribed fire; (2) public outreach; (3) noncommercial mechanical treatments; (4) commercial timber sales; and (5) monitoring. The purpose of the 10-year work plan was to set annual implementation targets

Table 1.—Quantification of current and desired conditions

Cover Type	Current Condition (%)	Desired Condition (%)
Oak-Hickory Closed Canopy Forest[a]	75	16
Oak-Hickory Woodland	1	35
Oak-Pine Closed Canopy Forest	8	0
Oak-Pine Woodland/ Savanna	4	38
Shortleaf Pine Closed Canopy Forest	10	0
Shortleaf Pine Woodland/Savanna	1	10
Glade/Prairie	1	1

[a]Includes degraded Oak-Hickory Woodland currently classified as Forest.

Table 2.—Restoration Treatments for 2004-2014

Project Area	Prescribed Fire	Thinning
Middle Fork	12,200 acres	4,300 acres
Piney	12,000 acres	6,705 acres
Rotary Ann	12,500 acres	4,478 acres
South Fork	6,000 acres	2,830 acres
Oak Mountain	6,000 acres	4,122 acres
East Side	11,300 acres	5,816 acres

for all staff (U.S. Forest Service and partners) working on the project and to track progress toward achieving the desired condition. Management actions with National Environmental Policy Act (NEPA) requirements were approved for the Middle Fork area in 2002. By 2004, all management action planning and NEPA requirements were completed for the remaining five restoration areas. Table 2 outlines the management actions planned for each restoration area. These actions will significantly move the ecosystems toward the desired overstory structural condition within 10 years.

Public Relations and Educational Strategies

The team developed public relations education strategies through the planning process (FLN 2002a). Defining these strategies broadens the thinking behind restoration implementation by identifying possible barriers and by exploring a variety of issues that are generally not captured with traditional planning efforts. The FLN team identified the need to develop public support for the use of prescribed fire and mechanical thinning for this ecosystem restoration project and specifically the need to deal with the public's concerns about smoke pollution. Furthermore, the team recommended addressing credibility issues with the public, enhancing collaborative relationships with partners, and developing an outreach process to include new partners.

Prescribed Burning

Prescribed fire is crucial in meeting desired ecological conditions. The desired fire regime includes a fire return interval that ranges from 2 to 5 years with the entire area being burned 2 to 4 times during the duration of the project (10 years). Both the seasonality and severity of fire are adjusted to achieve the desired ecological condition. Dormant-season fires are used in parts of the project area to reduce fuel loads. A combination of dormant- and growing-season fire is used to achieve the desired fire effects on plant community structure and composition.

Since the inception of the restoration project in 2002 through mid-2008, 100,000 acres have been burned (some units burned multiple times). Most of these large-scale burns have taken place in the dormant or early growing season (January-April). The average size of a single-day burn is about 2,000 acres. Multiple-day burn units are up to 5,500 acres. Less than 5 percent of the acreage has been burned in the summer. The largest growing-season burn spanned 10 days (June 24- July 4, 2006), due to the limited availability of a helicopter for aerial ignition. The historic fire regime is being restored on public and private lands.

Mechanical Treatments

Mechanical treatments are conducted on both public and private land. For the public lands, treatments are implemented using commercial timber sales, wildlife stand improvement thinning, public firewood cuts, and salvage sales to remove oak-decline affected trees. For private lands, the Forest Land Enhancement Program, Stewardship Incentive Program, and the Forest Incentive Program are being used. The treatments reduce overstory basal area and stem density and move the overstory structure and composition toward the desired condition. To accomplish a reduction in overstory basal area (BA), oak-dominated and mixed oak-pine stands are thinned to a BA of 40 to 60 square feet per acre. Pine-dominated stands are thinned to a BA of 50 to 70 square feet per acre. All thinnings remove the smaller diameter mid-story and co-dominant trees. Hardwood leave trees are selected to favor the historically dominant white and post oaks. As of mid-2008, 12,000 acres have been thinned to the desired condition.

Monitoring and Adaptive Management

Collaborative landscape-scale monitoring and adaptive management protocols have been developed for the restoration project. Monitoring components include fire effects on vegetation, fuel loading, and birds. The ecological monitoring program was developed by the Big Piney Ranger District, TNC, and Arkansas Natural Heritage Commission and reviewed by the Oak Ecosystem Restoration Team. The monitoring program documents and quantifies movement toward the desired condition, fuels reduction, and forest health. Specifically, the program includes monitoring goals, project success criteria, macro-plots sampling methodology, and nine individual monitoring protocols to determine whether the project is meeting success criteria defined by all partners.

The plant community monitoring component was established to answer two questions: (1) Are project activities reducing hazardous fuels within the project area; and (2) are the treatments moving the project area toward the structural and compositional desired ecological conditions? The monitoring utilizes a nested quadrat method with four, 3.25-ft x 3.25-ft herbaceous quadrats and two 11-ft radius shrub quadrats nested within one 33-ft radius tree plot. Analyses indicate treated areas (one to two prescribed fires) have fewer stems per acre, fewer shade-tolerant species, and increased herbaceous diversity.

CONCLUSION

The Oak Ecosystem Restoration and Fire Learning Network teams and Big Piney Ranger District identified 60,000 acres for a long-term woodland ecosystem restoration project in 2002. This is one of the largest, most successful restoration projects in the Forest Service Southern Region. It serves as a model for designated Oak and Pine Woodland Management Areas (252,333 acres) in the Revised Land and Resources Management Plan for the Ozark-St. Francis National Forests (USDA Forest Service 2005).

Collaborative landscape-scale fire restoration in the Ozark Highlands of Arkansas depends on a multi-partner approach, a defined project vision, quantification of current and desired conditions, information transfer, and a monitoring program to track progress toward achieving the desired condition. Partners bring expertise from a variety of backgrounds with diverse skill sets, resources, and expertise and are key to the success in landscape-scale restoration projects. A collaboratively developed project plan and vision outlining the steps needed to achieve the desired ecological outcomes are crucial for successful partnerships. This project integrates the ecological rationale for restoration with education and outreach to the public and a diverse suite of partners. A successful partnership has been formed with a common vision for sustainable restored ecosystems of oak and pine woodlands. Public support for restoring historic fire regimes and reducing hazardous fuels has strengthened. The project demonstrates the successful restoration of ecosystem structure and function in oak and shortleaf pine ecosystems within the Ozark - St. Francis National Forest and responds to the need to implement landscape-scale treatments on woodland ecosystems administered by the USFS (USDA FS 2005).

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RESTORING OAK ECOSYSTEMS ON NATIONAL FOREST SYSTEM LANDS IN THE EASTERN REGION: AN ADAPTIVE MANAGEMENT APPROACH

Gregory Nowacki[1], Michael Ablutz[1], Dan Yaussy[2], Thomas Schuler[2] , and Dan Dey[2]

[1]Regional Ecologist (GN) and Regional Silviculturist (retired) (MA), U.S. Forest Service, Eastern Region, Milwaukee, WI
[2]Research Foresters, U.S. Forest Service, Northern Research Station, Delaware, OH, Parsons, WV, and Columbia, MO
GN is corresponding author: to contact, call (414) 297-1977 or email at gnowacki@fs.fed.us

Abstract.—The U.S. Forest Service has recently completed an ecosystem restoration framework and enacted accompanying policy to help guide its nationwide efforts. The Eastern Region is in the midst of translating the general guidance set forth in these documents to actual on-the-ground restoration. We envision a set of coordinated field demonstrations that will initially focus on oak-dominated ecosystems—ecosystems that are greatly imperiled by compositional changes to shade-tolerant trees such as maple (*Acer*) and beech (*Fagus*) due to long-term fire suppression. In collaboration with National Forests and the Northern Research Station, an adaptive management approach (learning by doing) is being promoted whereby an experimental design and a set of potential silvicultural treatments will be applied to a network of sites regionwide. Land managers, researchers, and the general public will all benefit from this "networked" field demonstration, which employs a uniform science-based method for project design, treatment selection and installation, and monitoring efforts.

INTRODUCTION

Land management continues to become increasingly complex with time as our ecological knowledge, environmental awareness, and demand for resources grow. Ecosystem restoration is one of a long list of new items that land managers now have to consider. Ecosystem restoration has a disproportional influence on land management as it embodies and simultaneously benefits many other resource endeavors, including improved biodiversity, forest health, and wildlife habitat conditions; increased representation of natural communities (including old growth) and threatened and endangered species; and the reduction of nonnative invasive species. The challenge of integrating a new mandate into land management warrants a well thought-out approach. The objectives of this paper are to do just that by 1) providing a brief history of ecosystem restoration as it pertains specifically to the U.S. Forest Service (USFS); and 2) proffering the use of adaptive management principles to design a network of integrated field experiments for learning and demonstration across National Forests of the Eastern Region.

NATIONAL ECOSYSTEM RESTORATION FRAMEWORK

Public interest in ecosystem restoration has burgeoned across America, bolstered in part by catastrophic wildfires of the western United States. Growing public concern helped galvanize and propel support for political action. Legislation followed in the form of the President's Healthy Forests Initiative (2002) and the Healthy Forests Restoration Act (2003). Attempting to implement this legislation underscored the need for a more holistic, overarching approach involving ecosystem restoration. This realization, in turn, spurred the need for clear language on the subject; specifically, what is ecosystem restoration and how can it be attained?

Although the USFS has had a long history of rehabilitating degraded lands (in particular the "lands that nobody wanted" in the eastern United States [Shands and Healy 1977]), actual policy and direction on ecosystem restoration were lacking. In response, the USFS established a team to help formulate the necessary steps to guide ecosystem restoration efforts on National Forests and Grasslands (Day et al. 2006). The resulting framework adopted a pre-existing definition for ecosystem restoration (the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed [SER 2004]) and provided the following guiding principles:

1) Seek and set goals for restoration only as societal choices; public involvement is key.
2) Make operational decisions at the lowest possible levels in an organization.
3) Consider the effects of restoration at local and landscape levels.
4) Give priority to restoring ecosystem processes, such as hydrologic pulses for rivers and streams or prescribed burning for fire-dependent ecosystems.
5) Establish objectives for the long term.
6) Recognize that ecosystems are dynamic and that change is inevitable; avoid "static endpoint" thinking.
7) Use multiple sources of relevant information, such as historical records, scientific studies, practical experience, and indigenous knowledge to set targets/benchmarks for evaluating progress based on monitoring.
8) Deal with uncertainty by using adaptive approaches to restoration.
9) Design and implement monitoring as part of restoration.
10) Learn as you go—use the feedback loop not only to modify treatments but also to modify objectives to incorporate new information or changing social or ecological needs.

The framework is purposefully general for application across a wide variety of ecosystems managed by the USFS. As such, it is meant to serve as a foundation to which information and strategies of increasing resolution can be added to help National Forests and Grasslands implement ecosystem restoration on the ground. The USFS's dedication to this effort has been further demonstrated by its recent adoption of official restoration policy (USDA FS 2008).

EASTERN REGION PLANNING FOUNDATION

Land and resource management plans guide activities that take place on individual National Forests and Grasslands. It is through these management plans that opportunities for ecosystem restoration are identified and eventually implemented at the project level. The release of the ecosystem restoration framework and ensuing policy was fortuitous for the Eastern Region, where many National Forests/Grasslands had recently revised their individual land management plans (Fig. 1). As such, the Eastern Region was well positioned to begin restoration efforts with framework guidance, policy, and new management plans. A basic question ensued: "How can management and research at the Regional level best help eastern National Forests/Grasslands in their quest to restore ecosystems?"

IMPLEMENTATION OF RESTORATION THROUGH ADAPTIVE MANAGEMENT

A key role of USFS Regional Offices is to bring a broader perspective to issues, initiatives, or projects, especially those that span multiple National Forests and Grasslands or involve multiple partners or ownerships. Ecosystem restoration is a prime example – a multifaceted topic requiring broad integration for success. So, how best to embark on ecosystem restoration? One promising way is by taking an adaptive management approach (Walters and Holling 1990, Stankey et al. 2005). This approach is ideal in situations where high levels of uncertainty exist, which is indeed the case in restoring eastern U.S. ecosystems long affected by human manipulation (e.g., cutting, overgrazing, drainage, and introduction of nonnative pathogens and plants).

Adaptive management is founded on a formalized learning process that links directly to decision-making. Basically, it treats on-the-ground actions and policies as hypotheses from which we gain learning, which then provides the basis for modifying subsequent actions and policies (Stankey et al. 2005). A four-phase management cycle has been proffered that embraces learning, helping speed the acquisition and transfer of new knowledge

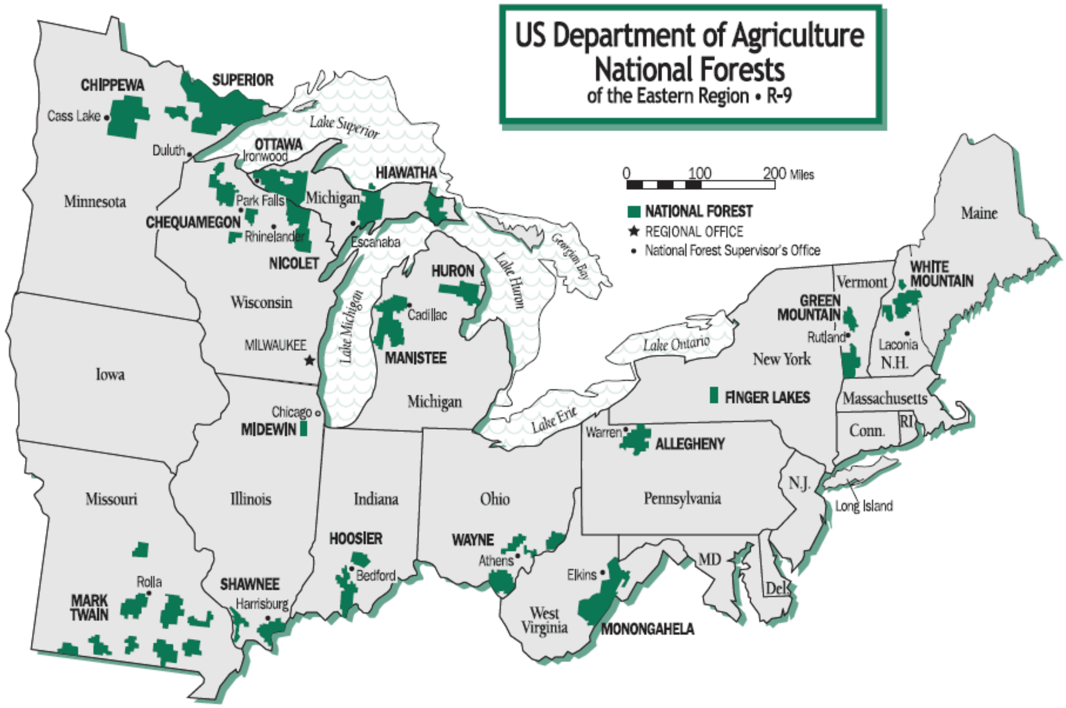

Figure 1.—The U.S. Forest Service Eastern Region and component National Forests and Grasslands.

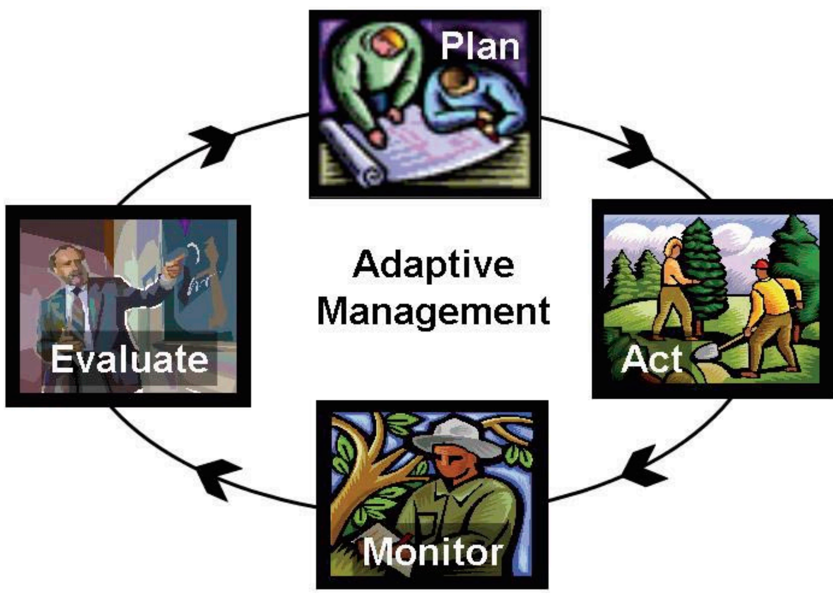

Figure 2.—The adaptive management cycle.

to management action (Fig. 2). In the first cycle, plans are constructed on existing knowledge, experience, and organizational goals. Actions are initiated on the ground and their effects monitored with scientific guidance (e.g., experimental design and layout, measured variables, appropriate data analyses). Monitoring results are evaluated in the final phase, which might trigger the cycle to reinitiate if existing practices are not meeting expectations. Practices failing to meet expectations can be dropped, reformulated, or replaced with new promising ones during the planning phase of the next cycle. Long-term commitment from all parties is essential to successfully carry out adaptive management and accrue its benefits (Bormann et al. 2007).

Employing an adaptive management approach for ecosystem restoration requires many steps, some of which run concurrently. First, the agency's intention to pursue ecosystem restoration must be conveyed to employees, potential partners, and the general public. Management opportunities for ecosystem restoration must be identified by National Forest/Grassland staff according to individual land management plans and affiliated field projects. Relevant research must be identified and interested scientists contacted to ensure best available science is used in all stages of planning, implementation, and monitoring.

Efficiencies can be maximized by developing an integrated network of study sites using a uniform experiment design and data collection methods (Yaussy et al. 2008). This unified approach allows for a preferred set of treatments to be identified and evaluated across a wide variety of conditions, forming a network of demonstration sites (for education) that meets scientific rigor (for true learning). Further efficiencies would be attained by pooling data for improved statistical analysis and trend detection. Some flexibility in treatments is acceptable, perhaps even desirable, so long as data sharing and analysis are not compromised. A strong research-land management partnership is required to ensure ecosystem restoration success, which by necessity requires a long-term commitment. Joint discussions should be held among researchers, land managers, resource specialists, and partners, whereby the above steps (and personnel carrying out those steps) are clarified to achieve ecosystem restoration.

ECOSYSTEM SELECTION FOR FIELD DEMONSTRATION

Ecosystem restoration is best demonstrated in the field. Fortunately, many individual restoration projects are already underway as National Forests/Grasslands implement their new land management plans. By linking these and future restoration projects across multiple Forests and Grasslands, we can demonstrate the advantages of taking a broader view through regional-scale analysis and evaluation. However, networking sites for data pooling and joint analysis requires focus on a single forest type or ecosystem. Selecting a representative ecosystem in which to employ this networked approach is difficult as the Eastern Region comprises a diverse number (from tallgrass prairies to sub-boreal conifer forests) with a variety of restoration needs. Indeed, all ecosystems are important and many are in dire need of recovery. For instance, red spruce forests in Appalachia have been severely affected by exploitive logging and wildfires of the late nineteenth and early twentieth centuries; only 12,000 ha of the original 600,000 ha currently remain in West Virginia (Rentch et al. 2007). Tallgrass prairies and associated oak savannas are considered the most severely degraded ecosystems on the North American continent, with virtually no original prairie land left in a pristine state (Nuzzo 1986, Packard and Mutel 1997).

Ultimately, it made sense to concentrate efforts on an ecosystem having a large geographic extent (to maximize the number of demonstration sites and applicability across the Region) with a well documented history of ecological alteration or degradation. To aid this endeavor as well as to help define reference conditions, we created a literature database of more than 500 articles that cataloged presettlement composition, structure, and disturbance regimes and post-settlement land-use impacts for the entire eastern United States. While compiling this database, which included information from historical, paleoecological, dendrochronological, fire-scar, and land survey records, we identified three broadly distributed ecosystems that stood out as having been heavily altered by European activity: 1) the loss of hemlock (*Tsuga canadensis*) and eastern white pine (*Pinus strobus*) in former conifer-northern hardwoods (Thompson et al. 2006, Schulte et al. 2007); 2) the near elimination of the tallgrass prairie-oak (*Quercus*) savanna mosaic in the Midwest (Transeau's [1935] "Prairie Peninsula," Anderson 1998); and 3) the ongoing conversion of oak-dominated systems to maple, beech, and other shade-tolerant trees (Nowacki and Abrams 2008).

The following logic was employed in ecosystem selection. In regards to the restoration of conifer-northern hardwoods, the prospects of re-establishing the conifer component look quite dim, especially for hemlock (Gustafson et al. 2007). First, thinning treatments would be needed to reduce hardwood (especially maple) competition in both the overstory and understory so that growing space would be made available for conifer regeneration. The lack of local seed sources due to preferential conifer removal by past logging and fire would require planting nursery stock in many locations—a substantial undertaking and expense. Moreover, even if conifer regeneration were successfully established, overbrowsing by white-tailed deer (*Odocoileus virginianus*) would threaten conifer growth and advancement in many areas (Alverson et al. 1988), necessitating additional control costs (e.g., fenced deer exclosures). Lastly, present-day investments into hemlock are questionable in light of the uncontrolled spread of the highly lethal hemlock woolly adelgid (*Adelges tsugae*) (Orwig and Foster 1998). In spite of these challenges, conifer restoration into northern hardwood forests is taking place on some National Forests where feasible.

Because of ownership patterns, the USFS's ability to effect change on the tallgrass prairie-oak savanna mosaic is largely limited to the Midewin National Tallgrass Prairie (Illinois) and Mark Twain National Forest (Missouri). Moreover, these units are already actively pursuing prairie-oak savanna restoration through prescribed burning.

Given the above situation, the expansive oak-dominated woodland-forest complex, which covers a sizable portion of the Eastern Region, is the logical choice for a multi-site restoration project. The benefits of restoring oak systems would be vast, greatly aiding mast-dependent wildlife populations and the rejuvenation of allied shrubs and ground flora requiring high-light conditions (Packard and Mutel 1997, McShea and Healy 2002). Research indicates that the near-universal conversion to shade-tolerant species is largely preventable through the intervention of silvicultural treatments, especially the reintroduction of fire (Brose et al. 2001; see also Dey and Fan this volume). Even in the presence of overabundant deer populations, it is ultimately the lack of understory light that causes the mortality of oak regeneration and associated ground flora (Anderson and Schwegman 1991, Oswalt et al. 2006, Yuska et al. 2008). A variety of silvicultural treatments look promising to improve light conditions and reduce competition, including thinning, prescribed burning, and herbicide treatment, either singly or in combination (Brose and Van Lear 1998, 1999; also Dey this volume).

CONCLUSION

The recently published ecosystem restoration framework and policy reaffirms the U.S. Forest Service's commitment to restoring ecosystems (Day et al. 2005, USDA FS 2008). With the recent completion of most Land Management Plans in the Eastern Region, National Forests and Grasslands are poised to initiate restoration efforts using framework guidance and policy. Since ecosystem restoration entails scientific knowledge, a broader landscape perspective, and multiple ownerships, the Eastern Regional Office has partnered with the Northern Research Station and The Nature Conservancy to facilitate these efforts. An adaptive management approach is envisioned whereby National Forests and Grasslands can collectively benefit through project networking. Specifically, by employing standardized methods of project layout and data collection, participants can test and compare different restoration techniques through shared data for a given forest type. Researchers will help in the experimental design and monitoring protocols, USFS ecologists and silviculturists will pool their knowledge to select the most promising restoration treatments, and resource specialists will implement and monitor treatments at the field level. We envision participants from both within and outside the agency learning to overcome significant challenges together as we restore oak ecosystems on the National Forests and Grasslands of the Eastern Region.

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POSTER ABSTRACTS

RESPONSE OF UPLAND OAK AND CO-OCCURRING COMPETITOR SEEDLINGS FOLLOWING SINGLE AND REPEATED PRESCRIBED FIRES

Heather D. Alexander[1], Mary Arthur[1], David Loftis[2], and Stephanie Green[3]

[1]Ph.D. Candidate (HDA) and Professor (MA), University of Kentucky, Forestry Department, Lexington, KY
[2]Project Leader, U.S. Forest Service, Bent Creek Experimental Forest, Asheville, NC
[3]Senior Land Management Specialist, Southwest Florida Water Management District, Brooksville, FL
HDA is corresponding author; to contact, call (859) 257-2806 or email at heather.alexander@uky.edu

Studies within and outside the United States indicate recurring oak (*Quercus* spp.) regeneration problems. In deciduous forests of the eastern U.S., a prevailing explanation for this trend is that fire suppression leads to high competitor abundance and low understory light. In response, prescribed fire is increasingly used as a management tool to remedy these conditions and encourage future oak establishment and growth. Within eastern Kentucky, we implemented single and repeated (3x) prescribed fires over a 6-yr period (2002-2007). Pre- and post-burn, we quantified canopy cover and oak seedling survival and growth compared to other woody seedlings deemed potential competitors, primarily red maple (*Acer rubrum* L.) and sassafras (*Sassafras albidum* [Nutt.] Nees.). Burning temporarily decreased canopy cover 3-10 percent, but cover rebounded the subsequent growing season. Repeated burning ultimately produced canopy cover about 6 percent lower than sites unburned and burned once, suggesting a cumulative effect on understory light. Red maple exhibited low survival (40 percent) following single and repeated burns, but growth remained similar to unburned seedlings. Burning had little impact on sassafras survival and led to total height and basal diameters two times greater than unburned seedlings. A single burn had no impact on red oak (*Erythrobalanus* spp.) survival and increased height and basal diameters 25-30 percent, but this positive growth response was driven by seedlings on several plots that experienced high burn temperatures and consequently high overstory mortality. White oaks (*Leucobalanus* spp.), however, exhibited twice the mortality of those unburned, with no change in growth parameters. Repeated burning negatively impacted survival and growth of both oak groups compared to unburned seedlings, and among both burn regimes, oaks with smaller pre-burn basal diameters exhibited the lowest postburn survival. Thus, despite the ability of prescribed burns to increase understory light and reduce red maple survival, neither single or repeated burns placed oaks in an improved competitive position. These findings result from a combination of highly variable yet interdependent factors including 1) the life history traits of oaks compared to their co-occurring competitors; 2) pre-burn stature of pre-existing oak seedlings; and 3) variability in fire temperature and effects on overstory structure.

FIRST-YEAR RESULTS OF A PRESCRIBED BURN IN A HIGH-ELEVATION RED OAK STAND

Peter Bates[1] and Dean Simon[2]

[1]Associate Professor, Department of Geosciences and Natural Resources, Western Carolina University, Cullowhee, NC
[2]Western Region Wildlife Forester, North Carolina Wildlife Resources Commission, Lawndale, NC
PB is corresponding author; to contact, call (828) 227-3914 or email at bates@email.wcu.edu

In March 2007, a prescribed burn was conducted on approximately 70 acres in a high-elevation red oak stand located on the Cold Mountain Game Land in western North Carolina. This burn was the first in a series of treatments designed to increase oak and hickory regeneration in this stand. Other treatments will include an herbicide application, a second prescribed burn, and a shelterwood harvest. Fifty-two, 1/10-acre, fixed-radius forest data collection plots were established prior to the burn. Thirty of these plots were within the prescribed burn area and 22 were outside of the burn. Vegetative conditions in these plots were remeasured in the year following the burn. Fuel loads were measured at each of the 52 plots immediately before the burn and in the 30 burn plots immediately after the burn. Fire intensity was estimated at each burn plot by hanging ceramic tiles with temperature-sensitive paint 20 inches above the ground. Data were collected to evaluate and compare the first-year effects of the fire (and fire intensity) on the following stand characteristics: (1) regeneration density of oaks, hickories, maples, and other species; (2) crown damage and mortality in overstory trees; and (3) hard mast production. Permanent photo points established at each plot provide a visual evaluation and comparison of preburn, postburn, and unburned plot conditions.

MODELING THE PROTECTION AFFORDED BY BURROWS, CAVITIES, AND ROOSTS DURING WILDLAND SURFACE FIRES

Anthony Bova and Matthew Dickinson[1]

[1]Physical Scientist (AB), and Research Ecologist (MD), U.S. Forest Service, Northern Research Station, Delaware, OH
AB is corresponding author; to contact, call (740) 368-0093 or email at abova@fs.fed.us

Wildland surface fires produce many toxic and irritating compounds, such as formaldehyde and acrolein, and harmful gases such as carbon monoxide. Several factors influence the degree of protection offered by animal shelters against combustion products and heat. Prominent among these variables are shelter configuration, the velocity of the prevailing wind, and time of exposure. We employed Fire Dynamics Simulator software, available from the National Institute of Standards and Technology, to model the mixing of smoke and gases in an assortment of animal shelters, including bark flaps, burrows, and tree cavities. Preliminary results are presented, and we discuss the possibility of developing a comparatively simple general relationship to estimate shelter vulnerability.

PRESCRIBED FIRE RESEARCH IN PENNSYLVANIA

Patrick Brose[1]

[1]Research Forester, U.S. Forest Service, Northern Research Station, Irvine, PA
To contact, call (814) 563-1040 or email at pbrose@fs.fed.us

Prescribed fire in Pennsylvania is a relatively new forestry practice because of the State's adverse experience with highly destructive wildfires in the early 1900s. The recent introduction of prescribed fire raises a myriad of questions regarding its correct and safe use. This poster briefly describes the prescribed fire research projects of the Forestry Sciences Lab located at Irvine, PA.

One of the primary reasons prescribed fire is becoming increasingly popular with land managers is its ability to benefit oak regeneration at the expense of other hardwood reproduction. Earlier research identified four principal factors that determine whether an advanced regeneration stem sprouts after being top-killed by a fire. Those factors are: season-of-burn, fire intensity, size of the stem's root collar, and depth of the root collar in the forest floor. The ***Post-Fire Sprouting Probability Project*** seeks to further understand this complex relationship by linking a fire's seasonality and total heat output to the sprouting response of the advance regeneration of 10 common hardwood species. This linkage is done on 6-ft circular plots by 1) determining litter and woody fuels loadings before and after a prescribed fire; 2) measuring heat output of the passing flame front with a datalogger/thermocoupler; and 3) tagging advanced regeneration of different hardwood species and excavating dead and sprouting hardwood advanced regeneration several weeks after the burn. These data will be developed into predictive models that land managers can use to better plan and implement prescribed fires to aid in regenerating Pennsylvania's oak forests.

Closely related to the above project is the ***Shelterwood—Burn Applicability Study*** done in cooperation with the Allegheny National Forest (ANF). This project seeks to ascertain whether a forestry practice that originated in the Piedmont Region can work in northern Pennsylvania, given the differences in forest conditions. From 2002 through 2004, extensive preburn data were collected on four fenced oak shelterwood stands on the ANF. In May 2005, half of each shelterwood stand was burned with a moderately intense prescribed fire to release the oak regeneration from competing hardwood reproduction. Initial post-fire data collection in 2006 indicated that the fires had done their job; densities of competing hardwood regeneration, primarily black birch (*Betula lenta*) and red maple (*Acer rubrum* L.), were reduced and nearly all the oak reproduction had sprouted. Another inventory is scheduled for summer 2008 and results will be published in a forestry journal.

One of the arguments against burning shelterwood stands is the distinct possibility of killing a high value tree or damaging the butt log. Post-fire mortality models have existed for decades for predicting whether a tree is likely to die within 3 years of being injured by a fire. Unfortunately, these models are limited to oaks and do not take into account the other high-value species, such as black cherry (*Prunus serotina* Ehrh.), sugar maple (*Acer saccharum* Marsh.), and yellow-poplar (*Liriodendron tulipfera* L.), that are often important associates in a mixed oak forest.

The ***Residual Tree Damage Project*** is being undertaken to fill that knowledge gap. Before a partially cut stand is burned, it is scouted for residual trees with excessive fuel loadings near their bases. These fuels are inventoried and the tree base and fuels photographed. The fuels are reinventoried after the fire and the tree is monitored for survival for 3 years. Preliminary results indicate only woody fuels exceeding 10 tons/acre that lie within 5 ft of a tree's base pose a threat of killing that tree.

A key part of any prescribed fire program is having and using fuel models to aid in predicting fire behavior. However, hardwood fuel models are few in number, are broad and general in terms of application, and have not been rigorously tested. Since 2002, the ***Eastern Fuel Models Project*** has been underway to evaluate the existing hardwood fuel models, develop new models for unrepresented fuel types, and disseminate this knowledge to forest managers in a photo series. Early results indicate that Fuel Model 9 (loose leaf litter) is reasonably accurate for oak-dominated forests in fall and spring; Fuel Model 8 (compacted litter) is appropriate for Allegheny hardwood and northern hardwood forests in spring; and no model seems to consistently work well for heath shrubs (e.g., blueberry, huckleberry, and mountain laurel) and hardwood slash.

RED MAPLE (*ACER RUBRUM*) RESPONSE TO PRESCRIBED BURNING ON THE WILLIAM B. BANKHEAD NATIONAL FOREST, ALABAMA

Stacy L. Clark and Callie J. Schweitzer[1]

[1]Research Foresters, U.S. Forest Service, Southern Research Station, Huntsville, AL
SLC is corresponding author; to contact, call (256) 585-0652 or email at stacyclark@fs.fed.us

Prescribed burning is part of Land and Resource Management Plans on National Forest System lands throughout the southeastern United States, and is sometimes implemented to achieve a desired future condition of oak-dominated forest or woodland habitat. However, effects of burning on oak (*Quercus* spp.) competitors, such as red maple (*Acer rubrum* L.) are not well understood. In this study, we documented red maple seedling and sapling response to prescribed burns at both large (stand-level) and small (0.01-acre plot level) scales, and we examined relationships between fire behavior and red maple response. The study was implemented in 2005 within a larger replicated silvicultural study on the William B. Bankhead National Forest in Lawrence County, AL. At the stand-level scale, prescribed burning had no effect on red maple sapling mortality, sprouting, or diameter growth in the largest size class (1.5- to 5.4-inch diameter at breast height [d.b.h.]). Burned stands showed a significant increase in red maple seedlings (< 4.6 ft height) and a significant decrease in small saplings (> 4.5 ft height, < 1.5 inch d.b.h.) compared to unburned stands. At the plot-level scale, neither maximum recorded temperature or heat index (duration of temperature > 90 °F) of the fire could significantly explain the variation in sapling density changes, sapling sprouting occurrence, sapling diameter growth, or changes in seedling density. Results indicate that one dormant-season prescribed burn was not effective at decreasing oak competition of red maple in the midstory but did increase red maple competitors in the understory.

SURFACE FIRE EFFECTS ON CONIFER AND HARDWOOD CROWNS—APPLICATIONS OF AN INTEGRAL PLUME MODEL

Matthew Dickinson[1], Anthony Bova[1], Kathleen Kavanagh[2], Antoine Randolph[3], and Lawrence Band[3]

[1] Research Ecologist (MD), Physical Scientist (AB), U.S. Forest Service, Northern Research Station, Delaware, OH
[2]Associate Professor, Department of Forest Resources, University of Idaho, Moscow, ID
[3]Graduate Student (AR), Professor (LB), Department of Geography, University of North Carolina, Chapel Hill, NC
MB is corresponding author; to contact, call (740) 368-0096 or email at mbdickinson@fs.fed.us

An integral plume model was applied to the problems of tree death from canopy injury in dormant-season hardwoods and branch embolism in Douglas fir (*Pseudotsuga menziesii*) crowns. Our purpose was to generate testable hypotheses. We used the integral plume models to relate crown injury to bole injury and to explore the effects of variation in fire behavior.

For dormant-season hardwoods, effects of the plume on the crown were modeled by a branch necrosis routine involving heat transfer and thermal tolerance. If a stem was predicted to survive stem heating from flames, aggregate branch death from plume heating determined proportional crown loss, which, in turn, determined tree allocation to spring refoliation. Hardwood mortality occurred as a function of subsequent growth. Differences in bark thickness among hardwood species led to large differences in susceptibility to modeled girdling from flames. For trees predicted to survive flame effects, branch diameter contributed significantly to differences in height of branch kill. A parametric equation based on multiple model simulations is provided to describe branch kill height as a function of fire behavior and crown characteristics.

Branch embolism in Douglas fir is hypothesized to occur in response to foliage exposures to high vapor pressure deficits within the plume occurring on a time scale too short for stomatal response. We speculate that the resulting branch embolism may be a pervasive effect of surface fires on Douglas fir during periods when stomata are open. Cuticular water loss was not sufficient to cause embolism.

FIRE HISTORY AND THE ESTABLISHMENT OF OAKS AND MAPLES IN SECOND-GROWTH FORESTS

Todd Hutchinson[1], Robert Long[1], Robert Ford[1], and Elaine Kennedy Sutherland[2]

[1]Research Ecologist (TH), Research Plant Pathologist (RL), Forestry Technician (retired) (RF), U.S. Forest Service, Northern Research Station, Delaware, OH

[2]Supervisory Research Biologist, U.S. Forest Service, Rocky Mountain Research Station

TH is corresponding author; to contact, call (740) 368-0064 or email at thutchinson@fs.fed.us

We used dendrochronology to examine the influence of past fires on oak and maple establishment. Six study units were located in southern Ohio, where organized fire control began in 1923. After stand thinning in 2000, we collected basal cross sections from cut stumps of oak (*n* = 137) and maple (*n* = 204). The fire history of each unit was developed from the oaks, and both oak and maple establishment were examined in relation to fire history. Twenty-six fires were documented from 1870 to1933; thereafter, only two fires were identified. Weibull median fire-return intervals ranged from 9.1 to 11.3 years for the period ending in 1935; mean fire occurrence probabilities (years/fires) for the same period ranged from 11.6 to 30.7 years. Among units, stand initiation began ca. 1845 to 1900 and virtually no oak recruitment was recorded after 1925. Most maples established after the cessation of fires. In several units, the last significant fire was followed immediately by a large pulse of maple establishment and the cessation of oak recruitment, indicating a direct relationship between fire cessation and a shift from oak to maple establishment.

FIRE HISTORY AND AGE STRUCTURE ANALYSIS IN THE SHERBURNE NATIONAL WILDLIFE REFUGE, MINNESOTA: ESTABLISHING REFERENCE CONDITIONS IN A REMNANT OAK SAVANNA WOODLAND

Kurt F. Kipfmueller[1] and Tim Hepola[2]

[1] Assistant Professor, Department of Geography, University of Minnesota
[2] Regional Fire Ecologist, U.S. Fish and Wildlife Service, Ft. Snelling, MN
TH is corresponding author; to contact, call (612) 713-5479 or email at tim_hepola@fws.gov

Oak savanna woodlands were once a dominant ecotone in southwestern Minnesota and throughout the upper Midwest. These ecosystems represented a transitional zone between prairie communities to the west that eventually graded into Big Woods forest. Most of the oak savanna landscape of southern Minnesota (and indeed most of the Midwest) were extensively homesteaded and farmed during the middle 1800s and few intact savanna landscapes remain today. The structure, origin, and factors that have maintained presettlement oak savanna have not been well documented or established. Fires, perhaps ignited by Native Americans, are thought to have maintained these landscapes, but the disturbance regime of these landscapes prior to widespread Euro-American impact remains elusive. Given the current interest in preserving, maintaining, and restoring these systems, it is imperative that the natural factors that have shaped these areas be investigated. Prescribed fires have become an important tool in this regard, but fire management activities can be better implemented if the natural role of fire is uncovered. Reconstructing the historic fire regime in oak savanna woodlands is challenging because few people are of sufficient age to extend to periods before appreciable impacts by human activities. However, this information is critical for establishing reference conditions that can be utilized to guide restoration activities and initiate management activities. This research will investigate the potential of developing reference conditions in a relatively intact oak savanna in the Sherburne National Wildlife Refuge (Minnesota). Vegetation plots will be established throughout the remnant to determine the variability in savanna structure and composition. Fire scars will be collected throughout the remnant to determine the variability in fire frequency and associated changes in savanna age-structure and composition related to fire activity. This research will provide a context for current management activities centered on maintaining and restoring oak savanna ecosystems. The principal questions to be answered include: What is the fire frequency of the oak savanna remnant and how much variation exists over space? How has fire frequency changed over time? Are there particular time periods when fire was more/less prominent? Are forest compositional and structural characteristics related to fire frequency? Has the seasonality of fire changed over time? Identifying the historic fire regime of oak savanna landscapes, the associated forest structures, and spatial variations is critical to developing sound management guidelines for the restoration of these landscapes. Since the proposed study area is one of the few old-growth remnants of this ecosystem in the refuge, information developed in this study will be useful in managing other parts of the refuge. Few detailed investigations of oak savanna fire history and age structure have been completed. This research will add substantially to our understanding of the vegetation dynamics and disturbance regime that characterize this landscape.

OAK REGENERATION ACROSS A HETEROGENEOUS LANDSCAPE IN OHIO: SOME LIMITED SUCCESS AFTER THINNING, TWO FIRES, AND SEVEN YEARS

Louis Iverson[1], Todd Hutchinson[1], Anantha Prasad[1], and Matthew Peters[1]

[1]Research Landscape Ecologist (LI), Research Ecologist (TH), Ecologist (AP), GIS Specialist (MP), U.S. Forest Service, Northern Research Station, Delaware, OH
LI is corresponding author; to contact, call (740) 368-0097 or email at liverson@fs.fed.us

We document an increase in oak and hickory advance regeneration, depending on landscape position, in the sixth year (2006) after mechanical thinning (2000) and repeated prescribed fires (2001 and 2005) across two sites (Raccoon Ecological Management Area and Zaleski State Forest) in southern Ohio. Each of four 20+ ha units (two controls and two thin plus burn twice) were modeled and mapped into a long-term moisture regime (integrated moisture index [IMI]) and plots were established at each of 242 points on a 50-m grid. Plots were monitored for light and vegetation in 2000, 2001, 2004, and 2006. From these data, we developed two simple models: (1) a model of oak "competitiveness," based on advance regeneration of oaks and their competitors; and (2) a model estimating the probability of a plot's becoming "competitive for oak" based on canopy openness, IMI class, and number of oak and hickory seedlings present. For dry or intermediate (not mesic) sites with at least 5,000 oak and hickory seedlings/ha, opening the canopy to 8.5-19 percent, followed by at least two fires, should allow oak and hickory to be "competitive" over about 50 percent of the area. Overall, these results suggest promise for partial harvesting and repeated fires as a management strategy to reverse the accelerating loss of oak dominance in the central hardwoods region.

PRESCRIBED FIRE AND OAK SEEDLING DEVELOPMENT IN AN APPALACHIAN FOREST

Jacob Royse[1], Mary Arthur[1], and David Loftis[2]

[1]Graduate student (JR) and Professor (MA), Department of Forestry, University of Kentucky, Lexington, KY
[2]Research Forester, U.S. Forest Service, Southern Research Station, Asheville, NC
JR is corresponding author; to contact, call (859) 257-8289 or email at Jacob.Royse@uky.edu

In recent decades considerable research has focused on the use of prescribed fire in oak-dominated forests with the management objective of promoting oak regeneration for future overstory dominance. These studies typically focus on the response of oak seedlings and saplings already in place when experimental prescribed fires are set because it is difficult to time fires to coincide with the production of an acorn mast crop. We examined the development of white oak (*Quercus alba*) and chestnut oak (*Q. prinus*) seedlings growing from seed after a mast crop in 2005. At the time the mast crop occurred, an experiment was already in place with three treatments: Fire-excluded (control), 1x-burn (burned in 2003), and 3x-burn (burned in 2003 and 2004, with an additional fire in 2006 when acorns were on the ground). We tracked seedling survival and growth over two growing seasons, in 2006 and 2007. Litter depth and percentage open sky were significant predictors of oak seedling mortality after the first and second growing seasons. As litter depth increased, so did oak seedling mortality. In contrast, as percent open sky increased, mortality decreased. White oak seedlings growing in low-light conditions (5 to 10 percent canopy openness) had higher mortality rates than did chestnut oak. Higher litter depth associated with fire-excluded and 1x-burn treatment led to greater stem lengths during the first growing season, likely due to excess stem elongation to breach the litter layer for available light. However, relative growth rate of stem length after 2 years revealed that oak seedlings on three-burn treatments far exceeded that of seedlings on fire-excluded treatments. Percent open sky and oak seedling diameter were positively correlated, and basal diameter of oak seedlings on three-burn and one-burn treatments had significantly greater diameters than fire-excluded seedlings after the second growing season. This study reveals the importance of light and litter depth on the early survival and growth of oak seedlings. Newly developing oak seedlings originating from acorns on recently burned sites displayed greater survival and growth rates than seedlings growing under the nearly closed canopy found on fire-excluded sites.

PREDICTING FIRE SCARS IN OZARK TIMBER SPECIES FOLLOWING PRESCRIBED BURNING

Aaron P. Stevenson[1], Richard P. Guyette[2], Rose-Marie Muzika[2]

[1]Forest Community Ecologist, Missouri Department of Conservation, West Plains, MO
[2]Research Professor (RG), Associate Professor (RMM), Department of Forestry, University of Missouri, Columbia, MO
APS is corresponding author; to contact, call (417) 255-9561 ext 272 or email at Aaron.Stevenson@mcd.mo.gov

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A potential consequence of using prescribed fire is heat-related injury to timber trees. Scars formed following fire injuries are often associated with extensive decay in hardwoods. The ability to predict scarring caused by prescribed fire is important when multiple management goals are incorporated on a single forest site. We evaluated the size and frequency of prescribed fire scars for six tree species in the Missouri Ozark Highlands. Tree species chosen for evaluation were hickories (*Carya* spp.), shortleaf pine (*Pinus echinata*), white oak (*Quercus alba*), scarlet oak (*Q. coccinea*), post oak (*Q. stellata*), and black oak (*Q. velutina*). We established plots on upland forest sites managed with prescribed fire and examined 3,945 trees for fire scars on these sites. Scarred trees were tallied and the largest scars were measured for each plot. Post-fire scarring models indicate bark char, a proxy for fire intensity, is the most important predictor of scar frequency and scar size. Other important predictors of scarring include tree size, aspect, and fetch. Species-specific scarring models developed from this study may be used to predict the extent of fire scars caused by prescribed burning.

AIRBORNE FIRE MONITORING—EXTRACTION OF ACTIVE FIRE FRONTS FROM TIME-SEQUENCE IMAGING OF THE ARCH ROCK FIRE IN SOUTHEAST OHIO

Loredana Suciu[1], Anthony S. Bova[2], Matthew B. Dickinson[2], Jim Dyer[1], Robert Kremens[3], and Valerie L. Young[4]

[1]M.S. candidate (LS) and Associate Professor (JD), Ohio University, Athens OH
[2]Physical Scientist (ASB) and Research Ecologist (MBD), U.S. Forest Service, Northern Research Station, Delaware, OH
[3]Senior Research Scientist, Rochester Institute of Technology, Rochester, NY
[4]Associate Professor, Ohio University, Athens, OH
MBD is corresponding author; to contact, call (740) 368-0096 or email at mbdickinson@fs.fed.us

This project investigates a prescribed fire in southeastern Ohio through remote sensing measurements of emitted radiance. Time-sequence airborne imaging consisted of multiple over-flights with a repeat interval of 3-6 minutes performed by an aircraft equipped with an infrared camera. Images were processed and analyzed to extract the active fire fronts and estimate the direction of fire propagation, rate of spread, fuel consumption, and fire intensity. Using airborne high-resolution infrared imagery provides an unprecedented opportunity to accurately estimate fuel consumption and fire intensity over large burn units. The resulting fire behavior information can then be used to determine the fire's ecological effects.

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Contains 10 full-length papers and 12 abstracts of posters that were presented at the 3rd Fire in Eastern Oak Forests conference, held in Carbondale, IL, May 20-22, 2008. The conference was attended by over 200 people from a variety of groups, including federal and state agencies, nongovernmental organizations, universities, and private citizens.

KEY WORDS: fire, eastern forests, forest management, oak

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