FIRE AND THE HERBACEOUS LAYER OF EASTERN OAK FORESTS

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

INTRODUCTION

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

THE HERBACEOUS LAYER

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

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

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

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

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

DIRECT FIRE EFFECTS ON PLANTS AND SEEDS

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

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

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

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

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

INDIRECT FIRE EFFECTS ON PLANTS AND SEEDS

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

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

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

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

FIRE EFFECTS ON HERB-LAYER COMMUNITIES: THREE CASE STUDIES

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

Missouri Ozarks Dry Oak Forest (Hartmann and Heumann 2003)

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

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

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

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

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

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

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

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

Southern Illinois Oak Barrens (Taft 2003)

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

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

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

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

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

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

FIRE EFFECTS ON HERB COMMUNITIES: GENERALIZATIONS

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

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

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

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

FIRE EFFECTS ON HERB COMMUNITIES: CAUSES OF VARIATION

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

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

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

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

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

EFFECTS OF SILVICULTURE AND FIRE ON THE HERBACEOUS LAYER

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

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

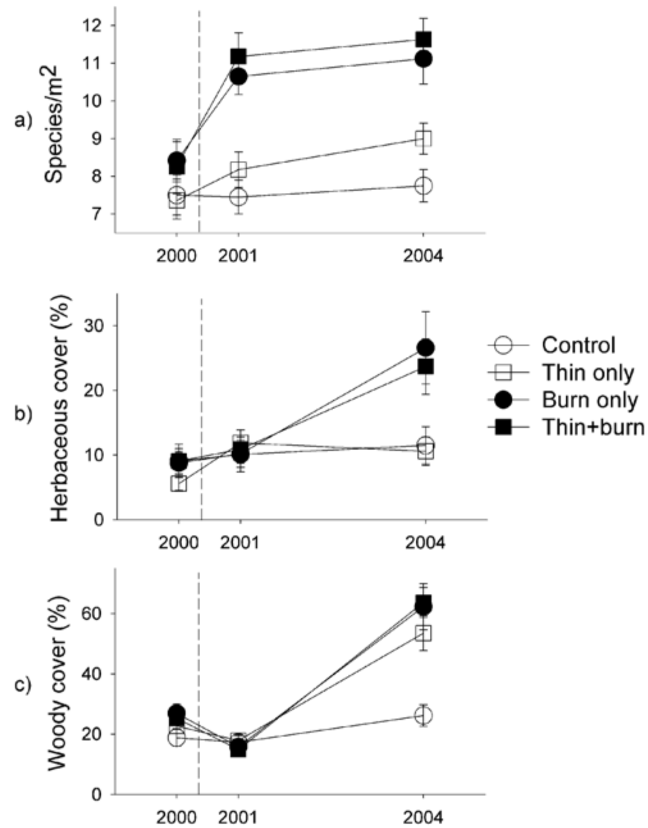

Figure 2.—Response of herbaceous layer vegetation to thinning, burning, and thinning+burning treatments at two replicate sites (Raccoon Ecological Management Area and Zaleski State Forest) of the Ohio Hills Fire and Fire Surrogate study site. Mean annual values (± 1 S.E.) for a) species richness per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

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

RARE PLANTS AND FIRE

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

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

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

MANAGEMENT IMPLICATIONS

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

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

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

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APPENDIX

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

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

APPENDIX—continued

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