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# Restoration in the Southern Appalachians:

## A Dialogue among Scientists, Planners, and Land Managers

W.T. Rankin and Nancy Herbert, Editors


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**Cover:**
Before-and-after photographs of a restoration project on the Buck Creek Serpentine Barrens in western North Carolina. Left: Spring 2005 (photo by Paul Davison, University of North Alabama). Right: Fall 2010 (photo by W. T. Rankin, USDA Forest Sevice)

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## PROLOGUE

On April 28, 2009, leaders from the Southern Region (R8) and the Southern Research Station (SRS) of the Forest Service, U.S. Department of Agriculture (USDA) met in Asheville, North Carolina, to assess the status of restoration efforts in the Southern Appalachian Mountain region. Forest Supervisors from the national forests in Virginia, North Carolina, South Carolina, Georgia, Tennessee, and Kentucky described restoration efforts underway on their national forests, with a focus on the portion of their national forests in the Southern Appalachians. Interspersed in their summaries were comments about the need for more research to support a particular restoration technique or restoration goal.

Although many of the topics they discussed have been the subject of considerable research, the information was not always in a useful format for Forest Supervisors and their staffs. Former SRS Director Jim Reaves offered to summarize the pertinent information into a format that would serve their needs. Responding to this request, the group met for a brainstorming session to identify the most pressing questions confronting ecosystem restoration in the Southern Appalachians. This session produced three areas of focus:

- The role of fire in the Southern Appalachians
- Early successional habitat in the Southern Appalachians
- Oak regeneration in the Southern Appalachians

SRS then worked with R8 Forest Supervisors to develop a list of questions in each of these focus areas. A survey of the planning staffs on the six national forests led to a list of additional questions. These questions were presented to SRS experts in the three areas of focus. Experts providing responses included employees from the Southern Research Station, Northern Research Station, District, Forest Health Protection, and Air Resources staffs in the Southern Region, and the University of Tennessee.

This publication uses a question-and-answer format. The questions were posed by Forest Supervisors and their planning staffs, and address issues in the restoration of unhealthy or degraded forest ecosystems. The answers were provided by conservation scientists from multiple organizations, all of whom shared a deep concern about the issues confronting the region. Common and scientific names for all species mentioned are listed in the taxonomic index beginning on p. 44.

# Restoration in the Southern Appalachians: A Dialogue among Scientists, Planners, and Land Managers

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## ABSTRACT

We address three key questions for restoration ecology in the Southern Appalachian Mountains. First, what is the role of fire, especially when used as a management tool for oak-dominated ecosystems? Second, what is the relationship between early successional habitat and biodiversity? And third, how do we regenerate oak ecosystems? To answer these questions, first, we examine the historic role of fire in the mountains, discuss its effects on forest resources, and summarize a strategy for restoring fire to ecosystems with a long history of fire exclusion. Second, we examine the relationship between early successional habitats and wildlife resources in the mountains, discuss the pattern and rate of natural disturbance, and provide suggestions for creating and maintaining early successional habitat. And third, we review current management for oak regeneration and discuss the implications for oak ecosystems in the absence of management. In addition to addressing current questions in restoration ecology, we provide an extensive bibliography of the scientific literature, especially for fire management. Our goal is to provide a concise and practical summary of the current restoration literature for use by forest planners and managers throughout the Southern Appalachian Mountains.


**Keywords**: Early successional habitat, natural disturbance patterns, natural fire, oak regeneration, prescribed fire, restoration ecology, Southern Appalachian Mountains, wildlife.

## 1. THE ROLE OF FIRE IN THE SOUTHERN APPALACHIANS

### 1.1 Did fire occur in the Southern Appalachians historically?

Fire has apparently played an integral role in determining historical patterns of forest vegetation across the Southern Appalachian Mountain region. Fossil pollen and charcoal-particle analyses suggest recurrent fire was common in forests of the region for at least 3,000 years before the arrival of Europeans (Delcourt and Delcourt 1997, Fesenmyer and Christensen 2010). Historical accounts suggest anthropogenic fire, often used to affect forest structure and composition, was common both before and after European colonization (DeVivo 1991, Fowler and Konopik 2007, Stewart 2002, Van Lear and Waldrop 1989). In addition, many of the traits characteristic of plant species in the Southern Appalachians can be interpreted as evolutionary responses to fire (Christensen 1977, Landers 1991, Lorimer 1985).

**The Five Periods of Anthropogenic Fire Regimes**

Fowler and Konopik (2007) outlined five periods of anthropogenic fire regimes in the Southern Appalachians, based on changing cultures, population sizes, and land use priorities:

**Circa 12,000 BP to 1500 AD**—During the first period, approximately 12,000 BP to 1500 AD, Native Americans most likely burned valleys near settlements to clear land for agriculture, while upper slopes and ridges were selectively burned to promote wildlife habitat. Based on estimates of population size and the amount of cleared land necessary to support these populations, the spatial effects of Native American burning may have reached one-quarter to one-half the amount of the current farmland in the Eastern States (Stanturf and others 2002). Fire return intervals varied between 1 and 12 years, depending on elevation, slope, aspect, and proximity to native villages (Barden 1997, Delcourt and Delcourt 1997, Frost 1995).

**Circa 1500 to mid-late 1800s AD—**The second period of fire use began with the arrival of European colonists in the 16th century. As the number of colonists increased, much of the landscape was occupied by settlers who adopted Native American practices. Recent dendrochronologies addressing this period have documented fire return intervals in xeric, central Appalachian oak and pine forests between 5 and 20 years (e.g., Aldrich and others 2010).

**Mid-late 1800s to early 1900s AD—**The third period of fire coincided with industrialization, beginning in the latter half of the 19th century, as railroads improved both the access to the mountains and the movement of large amounts of commodities. Large-scale timber harvests between 1880 and 1920 resulted in heavy fuel loads from slash, and created drier, more open stands. Fires were used to reduce slash and enhance grazing. The high fuel levels from the slash produced much higher intensity fires than in previous eras, although the frequency of the fires remained similar to earlier periods (Harmon 1982).

**Early 1900s to late 1900s AD—**The fourth period of fire began in the early 20th century. Following the high-intensity fires of the third period, forest managers actively suppressed wildland fire and discontinued the use of anthropogenic fire. Fire exclusion, however, caused important changes in the structure and function of Southern Appalachian forests, especially increases in fire-intolerant species, and concomitant decreases in fire-tolerant species (Vose 2000, 2003).

**Late 1900s to present—**The fifth period of fire began in the late 20th century. A half century of fire suppression created forests with heavy fuel loads, creating the potential for devastating wildland fires. Beginning in the 1970s, forest managers in the Southern Appalachians began using prescribed fire, especially in xeric forests dominated by pines and oaks, to reduce fuel loads and improve forest health. Prescribed fires are now the most common form of anthropogenic fire in the Southern Appalachian Mountains.

In summary, fire is a long-standing feature of the Southern Appalachian landscape. Although cultural perceptions and management practices have changed, especially during the past 50 years, all available evidence, from pollen cores to dendrochronologies to written and verbal histories, suggest fire, either natural or anthropogenic, has played an important role in the Southern Appalachians for many centuries (Spetich and others 2011).

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## 1.2 What are the different kinds of fire?

Fire can be classified in multiple ways. One of the more common classifications is to distinguish between prescribed fires and wildfires. Fires can also be classified by intensity and season.

### Classifying Fires

Prescribed fire, also known as controlled burning, is fire applied to ecosystems, at specific locations, and under specific weather conditions, to accomplish predetermined management objectives. Fire prescriptions typically control effects on ecosystems by controlling fire intensity, either by choosing the proper environmental conditions—wind, humidity, fuel moisture—or through site preparation. Fire prescriptions also address fire behavior and spread, by specifying the movement of the fire relative to the wind: head fires spread in the direction of the wind, backing fires spread against the wind, and flanking fires spread at right angles to the wind. Because wind patterns and fuel conditions are more variable in the mountains than in other regions of the South, considerable experience and training are required to conduct a successful prescribed fire in the Southern Appalachians (e.g., Achtemeier 2008).

Wildland fires, on the other hand, are unplanned. Although prescribed fires and wildland fires can share many characteristics, wildland fires are more likely to burn under severe fuel and weather conditions, creating hot fires that are difficult, and dangerous, to control. Because they are more likely to burn hot, wildland fires are also more likely to adversely affect Southern Appalachian forests, killing desirable trees and consuming the organic portion of the soil.

Some wildland fires are used to achieve management objectives. These fires, although not started by humans, are allowed to burn under specific fuel and weather conditions. In the Southern Appalachians, these fires are generally limited to wilderness areas or lands managed by the National Park Service.

Low-intensity fires rarely have the same effects as high-intensity fires. For example, less intense fires are less likely to produce early successional habitat than hotter, more intense fires. The effects of fire intensity, however, also depend upon the season. In general, dormant-season fires in the Southern Appalachians are more intense than growing-season fires, because growing-season moisture, combined with high humidity, often suppresses fire intensity. The effects of low-intensity fires during the growing season, however, can be similar to, or even more severe than, the effects of high-intensity fire during the dormant season, because the stem of most woody plants is severely damaged when the cambium layer reaches 145 °F (Wright and Bailey 1982), and this temperature is more easily reached during the heat of the growing season. In addition, most of the carbohydrates in shrubs and trees are located aboveground (Knapp and others 2009), so growing-season fire typically kills woody species more effectively than dormant-season fires. When these plants are topkilled, the plant contains fewer reserves for resprouting (Drewa and others 2002).

Early results from ongoing research suggest that multiple growing-season burns reduce woody cover while increasing herbaceous cover.[1] In general, however, the effect of growing-season fire on plant and animal communities in the Southern Appalachians is poorly documented, and not well understood (see Knapp and others 2009). This is especially true for long-term effects over a range of fire intensities (Fontaine and Kennedy 2012).

[1] Unpublished data. On file with: Craig Harper, Professor of Wildlife Management and the Extension Wildlife Specialist at the University of Tennessee, 280 Ellington Plant Sciences Building, Knoxville, TN 37996.

**Fire Effects**

Prescribed fires are used for a wide variety of objectives throughout the South. They can reduce hazardous fuels, dispose of logging debris, prepare sites for seeding or planting, influence vegetation composition and structure for many wildlife species, manage competing vegetation, control insects and disease, and improve forage for grazing. Scientists at Coweeta Hydrologic Laboratory have studied three forms of prescribed fire treatments: fell and burn, stand replacement, and understory. Each of these approaches has different ecological effects, and choosing among them depends on the desired future condition for the ecosystem.

**Fell-and-burn**—In the fell-and-burn approach, the ecosystem is thinned, and the slash cured in place for several months. This process increases the amount of coarse fuel in the ecosystem, which, in turn, increases the fire intensity. The fell-and-burn technique has been applied to pine-hardwood ecosystems, where it increases the productivity and insect resistance of commercial tree species (Vose 2000).

**Stand-replacement**—Stand-replacement fires are ignited during periods of low fuel moisture and high winds. These conditions increase the intensity of the fire to a point where the overstory trees are killed, allowing the establishment of a new, or replacement, stand. Stand-replacement fires produce a mosaic of fire effects across the landscape, due to spatial heterogeneity in fire intensity. Compared to the fell-and-burn technique, stand-replacement fires are less likely to impact the biogeochemical cycle—for example, nitrogen in the soil, carbon on the forest floor, and the chemistry of the streams—with most of the losses occurring on ridges (Vose and others 1999).

**Understory**—Understory fires are ignited during periods of high fuel moisture and low winds, limiting the effects to the fine fuels, such as small twigs and fallen leaves, on the forest floor. Understory fires are used to reduce fuels and influence the composition and structure of understory vegetation, which indirectly affects food and cover resources for many wildlife species (Jackson and others 2007, McCord and Harper 2011). Several researchers have shown understory fires in xeric, intermediate, and mesic sites increase the diversity of understory plants, with no measurable negative impacts on water quality or site productivity (Elliott and Vose 2005b, Elliott and others 2004, Hubbard and others 2004, Vose 2003).

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## 1.3 What are the effects of fire on nongame species in the Southern Appalachians?

Fire effects on wildlife are most closely associated with changes to habitats and microhabitats in the forest, such as changes to the trees, shrubs, and leaf litter. Low-intensity burns generally do not kill trees. Because the trees are not killed, the general structure of the forest remains unchanged, and microhabitats within the stand either are little affected or recover quickly. In contrast, hot fires that kill trees can change the forest structure, significantly altering light levels and microclimates throughout the stand, and therefore may have a substantial effect on wildlife communities. In the Southern Appalachians, most prescribed fire is low intensity and does not significantly alter the general structure of the stand.

**Small mammals**—The few published studies examining the effect of burns on small mammals are inconsistent, likely due, in part, to high variability among the sites, and low levels of replication within the studies. Some studies report similar white-footed or deer mouse (*Peromyscus* spp.) abundance in both burned and unburned hardwood forest (Ford and others 1999, Keyser and others 2001, Raybuck and others 2012), while others (Kirkland and others 1996) report a lower abundance of white-footed mice in burned stands. In contrast, Krefting and Ahlgren (1974) reported higher densities of deer mice in burned, mixed conifer-hardwood forest sites. In the Southern Appalachians, we found more white-footed mice in burned stands, possibly because the fire reduced the depth of the leaf litter (Greenberg and others 2006). Increased populations of *Peromyscus* on burned sites have been attributed to better visibility and higher concentrations of seed, a food source for the mice, after reductions in litter cover and depth (Ahlgren 1966, Tester 1965).

**Bats**—Prescribed fire may have direct, short-term effects on bats as well as indirect, long-term effects. Short-term effects include disturbance, and potential mortality, of litter-hibernating bats such as eastern red bats during winter burns (Moorman and others 1999, Saugey and others 1989), and disturbance of crevice-roosting bats such as northern long-eared and Indiana bats during late spring and summer burns (Dickinson and others 2009). Fire may also destroy roosting sites, temporarily reducing the suitability of burn sites for roosting (Moorman and others 1999, O'Keefe and Loeb 2010).

Most studies in the Eastern United States, however, have found the long-term effects of prescribed fire are either neutral or beneficial to bats. Because fire often reduces physical obstructions in the environment that may interfere with echolocation and maneuver, bat foraging activity often increases in burned areas (Lacki and others 2009, Loeb and Waldrop 2008, Smith and Gehrt 2010). Insect abundance may also increase following fire, producing more suitable foraging sites (Lacki and others 2009, Malison and Baxter 2010).

Prescribed fire may also improve summer roosting habitat. Northern long-eared bats (Lacki and others 2009) and evening bats (Boyles and Aubrey 2006) select roost trees in burned sites, perhaps due to increased solar radiation. In addition, male Indiana bats appear to select roosts in both pine trees (MacGregor and others 1999) and hardwood trees (Johnson and others 2010) in burned sites. The effects of fire on Indiana bat maternity colonies in the Southern Appalachians are currently being investigated (O'Keefe and Loeb 2010).

**Herpetofauna**—Among the few studies of the herpetofauna in the Southern Appalachians, most suggest prescribed fire does not substantially change amphibian abundance. For example, Ford and others (1999) reported high-intensity prescribed fire had no effect on woodland salamanders. Matthews and others (2010) found low-intensity prescribed burns that did not kill overstory trees had no effect on salamanders, and we found similar results in a study at the Cold Mountain Game Lands (North Carolina), where a prescribed burn had no detectable results on terrestrial salamanders (Raybuck and others 2012). In contrast, we have also found that although a single prescribed burn that killed overstory trees did not reduce the relative abundance of terrestrial salamanders (Greenberg and Waldrop 2008), salamander abundance declined after a second burn in the same study sites (Matthews and others 2010).

Several studies indicate prescribed fire does not affect the abundance of frogs, and may increase toad abundance (Greenberg and Waldrop 2008, Matthews and others 2009). Hot fires that produce overstory mortality may benefit reptiles, particularly lizards, by creating more open conditions and warmer temperatures (Greenberg and Waldrop 2008, Matthews and others 2010, Moorman and others 2011, Moseley and others 2003, Renken 2006, Russell and others 1999).

**Birds**—In the Southern Appalachians, we found low-intensity, dormant-season burns had few detectable effects on breeding birds (Greenberg and others 2007b). In contrast, total bird species richness and density increased in communities treated with hot prescribed fire that killed trees and created snags. Responses to prescribed fire differed considerably among bird species, according to their associations with specific habitat features. Because the fire removes the leaf litter and shrub layers, bird species associated with these structural features may show short-term decreases in abundance. For example, ground-nesting worm-eating warblers, or shrub-dwelling hooded warblers, may decline until the leaf litter is replenished the following autumn, or until shrub cover increases within a year or two. In general, however, high-intensity burning with heavy tree-kill can be used to increase bird species associated with open habitats, such as indigo buntings and eastern bluebirds, or snags (woodpeckers and secondary-cavity nesters), while retaining many forest and generalist species (Greenberg and others 2007b).

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## 1.4 What are the effects of fire on soils in the Southern Appalachians?

Due to slow rates of decomposition, brought on by a combination of dry conditions and low nutrient concentrations, fire-dependent forests tend to accumulate woody debris and plant materials on the forest floor (e.g., Woodall and Likens 2008). As a result, fire can play an important role in these stands, quickly returning nutrients to the mineral soil, and creating a more open environment that allows seed germination and seedling establishment.

Much of the research on fire effects to forest soils has focused on soil organic matter. Soil organic matter is critical for maintaining soil quality because it retains water, carbon, and essential nutrients, such as nitrogen. A meta-analysis of more than 40 published papers showed that, on average, prescribed burning had no effect on total soil carbon and nitrogen concentrations (Johnson and Curtis 2001). The effect of fire on soil carbon and nutrients, however, varied greatly with site and burning conditions. In general, higher intensity burns reduce soil organic matter by directly consuming organic carbon, as well as soil nutrients, by volatilizing organic nitrogen (Knoepp and others 2005). High-intensity burns also consume large amounts of the organic materials on the forest floor, removing future soil organic matter, or exposing mineral soil, potentially increasing both soil erosion and nutrient loss (Robichaud and Waldrop 1994). On the other hand, burning woody debris and plant material on the forest floor can add cations such as calcium and magnesium to the surface soil, increase soil pH, and increase nitrogen availability, especially if some of the volatilized nitrogen is retained (Kovacic and others 1986, Raison and others 1990). In addition, low-intensity burns appear to have little effect on soil erosion, even in the steep terrain of the Southern Appalachians, due to rapid regrowth of the burned community (Van Lear and Danielovich 1988).

Most research on fire effects in the Southern Appalachians has examined the impacts of single fires. In comparison, the effects of multiple fires over a period of years are poorly studied. Although most soil variables show little change immediately after burning (Knoepp and others 2004), recurring fire may produce cumulative effects not evident after a single fire. For example, Vance and Henderson (1984), working in Missouri on an oak flatwoods burned repeatedly over 30 years, found nitrogen mineralization was reduced by long-term burning, and Neary and others (2003), working in the Western United States, found that shorter fire return intervals may reduce carbon and nitrogen in surface soils. Long-term burning may also result in an accumulation of recalcitrant forms of carbon, including black carbon (Ponomarenko and Anderson 2001).

Only one study, the National Fire and Fire Surrogate Study (e.g., Youngblood and others 2007), has shown impacts to soils after repeated burning in the Appalachian region. The study encompassed two mid-elevation, oak-hickory forest communities in the Appalachian plateau of southeastern Ohio, with two dormant-season fires on each site. Principal effects included the following:

- Direct soil heating, which may alter soil properties and kill soil organisms (Boerner and others 2005, Giai and Boerner 2007), but did not increase soil compaction (Boerner and others 2007);
- The volatilization and convection, in ash, of nitrogen, phosphorus, and cations (Coates and others 2008, Huang and others 2007);
- Minor levels of mineral soil exposure, which may lead to sheet erosion if the slope is sufficient (Coates 2006);
- Subtle and transient changes in pH, nitrogen availability, organic carbon, C/N ratio, and soil microorganisms (Boerner and others 2006, 2007; Coates and others 2008);
- Complex effects due to the interactions of site quality, slope position, and fire behavior (Boerner 2006).

Boerner and others (2007) concluded that both prescribed fire and restoration thinning can be applied to oak-hickory forests without significant negative effects on forest soils.

In summary, high-intensity fires, such as the fires used to prepare sites for planting, may reduce soil organic matter, reducing the amount of soil nutrients, and increase the amount of exposed mineral soil, increasing the risk of erosion. When properly applied, however, low-intensity, prescribed fire minimizes these effects, producing little change in soil nutrients (Boerner and others 2007, Knoepp and others 2004) and soil erosion (Van Lear and Danielovich 1988).

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## 1.5 What are the effects of fire on air quality in the Southern Appalachians?

The primary products of burning forest fuels are heat, water, and carbon dioxide. Wildland fires, however, rarely burn fuels completely. As the temperature of the fire drops, fuels begin to smolder, producing smoke. Depending on the amount and condition of the fuels, all woodland fires produce smoke to some degree. Smoke is a complex mixture of toxic gases, including carbon monoxide, sulfur dioxide, nitrogen dioxide, and ozone, as well as particulate matter, including soot and tar (Chi and others 1979). The effects of wildland fire, therefore, can be related to either the direct effects of the toxic gases on human health, or the indirect effects of the particulates on the human environment, especially in relation to fog and traffic safety. When any component of smoke is present at levels that adversely affect human health and safety, it can be considered a pollutant.

In general, wildland fire, especially prescribed fire, rarely produces gases at levels affecting human health. Less than 3 percent of the total national emissions of carbon monoxide and hydrocarbons can be attributed to prescribed fire (SAMAB 1996). Wildland fire is not likely to release nitrogen oxides in significant quantities because the threshold temperature for the release of these compounds, 2,700 °F, is hotter than the temperatures that normally occur during prescribed fires (McMahon and Ryan 1976). Compared to natural and anthropogenic sources, sulfur dioxide emissions from forest fires are negligible (Hall 1972). Most atmospheric sulfur dioxide comes from natural sources, such as volcanoes, oceans, and plant decay; industrial sources account for about 10 percent of total emissions (Hardy and others 2001, Komarek 1970).

Small amounts of nitrogen oxides and volatile organic compounds, however, are precursors to ground-level ozone and may become important in locations where ozone levels are already problematic. These locations include low-elevation (<3,500 feet), intermountain basins in the Southern Appalachians, especially during the summer months, when high-pressure weather systems limit the amount of upper-level mixing. In addition, sunlight working directly on nitrous oxide compounds can form other compounds, primarily ozone, that are potentially harmful to vegetation, particularly during the growing season. For example, in February 2007, two prescribed fires in central Georgia increased ozone concentrations in the Atlanta metropolitan area approximately 30 percent. These elevated levels were primarily due to increases in nitrogen oxides and volatile organic compounds released from the fuels, but also due to increases in volatile organic compounds released by the heating of the forest canopy (Hu and others 2008, Liu and others 2009). Most wildland fires in the Southern Appalachians, however, occur during the spring and fall, when windy weather produces dry fuels, favoring more efficient combustion. Windy conditions also improve atmospheric mixing, which dilutes emissions over a wider area. As a result, ozone tends to rapidly disperse to acceptable levels. Because dispersal is effective at managing ozone levels, many of the potentially negative effects of wildland fires on air quality can be mitigated by carefully planning and executing controlled burns.

The major pollutant produced by prescribed burning is particulate matter (Dieterich 1971, Sandberg and others 1979). Particulate matter is a complex mixture of soot, tars, and volatile organic compounds (McMahon 1977). Although fires produce many sizes of particles, the most problematic are the smallest particles, typically defined as $<2.5$ microns in diameter, because they can remain airborne for a considerable amount of time, and they can penetrate deeply into human lungs where they may contribute directly to respiratory problems (Hardy and others 2001). Particles $<2.5$ microns represent approximately 70 percent of the particles in smoke from wildland fires (Hardy and others 2001).

Concentrations of particulate matter are tracked by the Forest Service and compared to the National Ambient Air Quality Standard (U.S. Forest Service 2011). Wildland fires in western North Carolina have occasionally exceeded the standard for particles $\leq 2.5$ microns, primarily due to heavy fuel loads produced by outbreaks of southern pine beetle. Compared to gaseous and particulate atmospheric pollutants from industrial, utility, or mobile sources, however, wildland fires are relatively minor (SAMAB 1996). In general, the concentration of particles $<2.5$ microns produced by prescribed burning does not exceed the National Ambient Air Quality Standard (Hardy and others 2001).

On the other hand, public concern is likely to occur before particulate levels violate National Ambient Air Quality Standards. For example, particulates also affect visibility. Most of the haze in the Southern Appalachians can be attributed to particles $\leq 2.5$ microns, particularly in the morning, when humidity tends to be high (Abdel-Aty and others 2011). When wind speed is low and humidity high, moisture in the air condenses around particulates, forming dense smoke or a combination of smoke and fog. This potentially leads to traffic safety issues, and the risk of smoke moving into sensitive areas such as airports, highways, and communities is probably the major concern related to air quality and prescribed burning. Smoke that disperses during the night, when relative humidity is near 100 percent, can also result in superfog, a condition where visibility is reduced to a few feet (Achtemeier 2009). On a busy highway, superfogs can contribute to multiple collisions and traffic pileups. For example, in January 2008, a superfog from a woodland fire in central Florida led to a 70-car pileup that killed 4 motorists on Interstate 4.

Smoke also contributes to regional haze, reducing visibility at scenic views (Tombach and Brewer 2005). Recently, the Forest Service participated in a technical analysis aimed at reducing fire emissions and regional haze in the Southern Appalachians (SAMI 2002). Currently, the emissions from wildland fires are not considered a significant contributor to regional haze (SAMAB 1996).

Problem smoke is a chronic issue in the South for several reasons. First, prescribed fire produces large amounts of smoke. To avoid damaging tree roots, prescribed burning is typically conducted when soil and litter are moist. Moist fuels burn less efficiently and smolder longer than dry fuels, both increasing the amount of smoke produced and reducing the amount of heat available to carry the smoke aloft. This combination, in turn, increases the likelihood that the smoke will stay close to the ground. Second, the climate is characterized by air masses that trap smoke close to the ground. During the winter, shallow valleys can develop atmospheric inversions overnight, trapping smoke near the ground, and drainage winds can carry smoke as much as 10 miles, far enough to reach roadways in many locations. Finally, the region is densely populated, producing a large amount of wildland-urban interface and exacerbating interactions between smoke and the human environment.

Because of the potentially serious effects of prescribed fire on air quality, guidelines for smoke management have been developed by the Forest Service to reduce the atmospheric impacts of prescribed fire (Hardy and others 2001). Although prescribed fire produces the same emissions as wildland fire, forest managers can choose the season and weather conditions for prescribed fire, allowing them to mitigate many of the effects of the fire. For example, dormant-season burns occur at lower fuel moistures and under more consistent weather conditions than growing-season burns, producing lower emissions and improving control over smoke movements.

In addition, the Forest Service uses several computer models to predict the movement of smoke across the southern landscape. For example, VSMOKE models the daytime movement and concentration of particulate matter in smoke, assuming level terrain and unchanging winds (Lavdas 1996). A second application, the PB-Piedmont model, models the movement of smoke trapped near the ground at night across the complex terrain of the Piedmont (Achtemeier 2001). Researchers are planning two sister programs of the PB-Piedmont model, one for the Appalachians, and one for coastal areas influenced by sea/land circulations.

Most Southern States require a permit for prescribed burning. The permitting process provides a method for managing smoke in critical areas, and a mechanism for administering laws and other regulations (Brenner and Wade 1992, Haines and Cleaves 1995, Yoder and others 2003). The permit process is particularly useful for maintaining regional air quality standards because it allows State agencies to coordinate and evaluate the total amount of smoke produced on a regional basis. Obtaining a permit may be difficult in some areas of the Southern Appalachians because favorable days for controlled burning may be few, and many resource managers may want to burn when favorable conditions occur. In general, however, well-planned controlled burns performed under favorable weather and with good estimates of fuel consumption and fire intensity, can be implemented with a high confidence that air quality standards will be maintained (Hardy and others 2001).

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## 1.6 Are there ecosystems in the Southern Appalachians where fire isn't appropriate?

The appropriate use of fire depends on a variety of factors, each dependent on the ecosystem in question:

- What is the history of fire in the ecosystem? For example, dry or xeric communities in the Southern Appalachians historically exhibited frequent, but lower intensity, fires. More mesic forests, on the other hand, historically exhibited less frequent, but potentially higher intensity, fires (Stanturf and others 2002: Table 25.1).

- What is the fuel condition of the ecosystem? For example, dry, shrubby forests have higher fuel loads than more mesic, open forests, and are more likely to have carried fire in the past. Fuel loads can also change in response to fire suppression, overstory disturbance, and the presence of invasive plant species.

- What are the perceived effects on the ecosystems? For example, fire can substantially change the biodiversity and structure of a forest.

The following ecosystems occur in the Southern Appalachians, and are therefore potentially subject to prescribed fire. For each ecosystem, Reilly and others (2012) have assessed the fuel loads and the potential effects of prescribed fire.

**Bottomland hardwood forests**—Bottomland hardwood forests are found at the lowest elevations in the major river valleys. These forests are very productive, with rapid decomposition rates due to seasonal flooding and high soil moisture. Floods play an important role in the disturbance regime, and may redistribute coarse woody debris and remove litter, especially after large events.

Floodplain forests are particularly prone to invasion by exotic species (Brown and Peet 2003), and these species have potentially altered the fuel structure in bottomland hardwood forests. For example, dense thickets of Chinese privet and multiflora rose may form large patches of continuous fuels capable of carrying fire under dry conditions, and kudzu may reach into forest canopies along forest edges, creating ladder fuels. The presence of invasive species may warrant the use of fire to reduce localized fire hazards.

On the other hand, the role of fire in these ecosystems is poorly understood. Wade and others (2000) caution the tree species associated with bottomland forests tend to be sensitive to fire, and the species patterns in these communities tend to reflect the hydrology of the system, not the fire regime. As a result, bottomland forests in the Southern Appalachians do not appear to be fire-adapted ecosystems suitable for prescribed burning.

**Oak forests**—Oak forests are the most extensive ecosystems in the Southern Appalachians, occurring across a wide range of elevations, and varying in both topography and moisture regime. Xeric oak forests are frequently dominated by chestnut and scarlet oaks, while mesic oak forests are dominated by white oak and northern red oak.

A thick layer of potentially flammable shrubs, primarily mountain laurel, blueberry, and huckleberry, is often present in oak forests, especially in more xeric conditions (Waldrop and others 2007). Shrubs can represent a large proportion of the hazardous fuels in the community, particularly when composed of mountain laurel, and frequently pose a serious problem for fuel management (Stanturf and others 2002, Waldrop and Brose 1999).

Most studies show only limited benefits to oak following prescribed fire (Alexander and others 2008, Hutchinson and others 2005b, Signell and others 2005, Wendel and Smith 1986). Although the relationship between oak regeneration and fire is complex, controlling competing vegetation and modifying light in the understory are necessary to maintain oak forests in the face of succession towards a more mesic condition, in which stands currently dominated by oaks would be replaced by stands dominated by species such as red maple (Nowacki and Abrams 2008). In addition, prescribed fire appears to increase herbaceous cover and diversity in the understory of oak forests (Burton and others 2011, Elliott and others 2011, Hutchinson 2006, Hutchinson and others 2005a).

Because of the historical role of fire in creating and maintaining healthy oak forests, prescribed burning can be a valuable management tool in these ecosystems.

**Southern yellow pine forests**—Southern yellow pine forests occur on the xeric upper slopes and ridges of the Southern Appalachians, at low and middle elevations. Dominant species include Virginia pine, pitch pine, shortleaf pine, and Table Mountain pine. A dense shrub layer, including blueberry, huckleberry, and mountain laurel, is frequently present.

Many yellow pine stands were established early in the 20th century before the period of fire exclusion (Brose and Waldrop 2006) and have become decadent (Williams and others 1990). Prescribed fire has been commonly used to promote regeneration in yellow pines, especially Table Mountain and pitch pines, by reducing the number of encroaching shrubs and hardwood species. At one time, regeneration was associated with intense, stand-replacing fires, but more recent research suggests periodic surface fires of moderate intensity may be sufficient (Brose and Waldrop 2006, Waldrop and Brose 1999).

Southern yellow pine ecosystems represent one of the most challenging situations for fuel managers. Flammable evergreen canopies and abundant vertical fuels, such as mountain laurel, can result in severe crown fires. In addition, disturbance such as wind, ice storms, and southern pine beetle infestations can increase the abundance of both small- and large-diameter woody fuels (Waldrop and others 2007). Periodic surface fires would not only facilitate regeneration but would also reduce dangerous fuel loads. As a result, southern yellow pine ecosystems appear ideally suited to a program of prescribed fire.

**Mixed mesophytic/rich cove forest**—Mixed mesophytic forests, also known as rich cove forests, are among the most diverse communities in the Southern Appalachians. These forests are typically found on moist, east- and north-facing slopes and sheltered coves, at low and mid-elevations. The forests are dominated by yellow poplar, sweet birch, sugar maple, and black cherry, and generally support a diverse herbaceous flora.

Because they occur in sheltered coves that collect and retain moisture, rich coves are generally more mesic than other mid-elevation forest communities in the Southern Appalachians, with higher fuel moistures. These conditions generally reduce the frequency of fire, and rich coves are usually associated with low fire frequencies, although few studies have examined this relationship closely. Disturbance in cove forests is more often associated with canopy gaps produced by the fall of one or a few trees (Runkle 1982, 1990). Periods of prolonged drought can exacerbate overstory mortality, which may increase surface fuels and midstory density, especially in canopy gaps, increasing the possibility of catastrophic fire (Olano and Palmer 2003). As a result, prescribed fire may help reduce the likelihood of devastating fire in these ecosystems. Compared to oak and pine ecosystems, however, the role of fire in rich cove forests has been rarely studied, and remains poorly understood (Wade and others 2000). In general, rich cove forests do not appear suitable for a program of prescribed fire.

**White pine-hemlock-hardwood forests**—White pine-hemlock-hardwood forests are typical of cool, moist ravines over a range of elevations. Also known as acidic coves, these forests are often composed of large-diameter trees at low density, with a thick shrub layer of rosebay rhododendron.

The historical disturbance regime of white pine-hemlock-hardwood forests was likely dominated by wind (Lorimer 1995). Although generally long-lived, white pine and eastern hemlock are both characterized by shallow root systems, and are therefore susceptible to windthrow. These forests are also characterized by low fire frequencies because they characteristically grow in ravines with high humidity and soil moisture. When fire does occur, however, mortality can be high (Reilly and others 2006).

The recent invasion of the hemlock woolly adelgid has devastated hemlock communities throughout the Southern Appalachians. High rates of mortality will likely cause a pulse in both small and large surface fuels (e.g., Waldrop and others 2007). Because of these large fuel loads, white pine-hemlock-hardwood communities in the Southern Appalachians may be unusually susceptible to catastrophic fire at the present time.

Given the unusually high fuel loads, the infrequent fire return intervals, and the general resistance of the ecosystem to burning, white pine-hemlock-hardwood forests do not appear suitable for prescribed fire.

**Northern hardwood forests**—In the Southern Appalachians, northern hardwood forests occur in coves, and on upper slopes at elevations above 4,000 feet. These stands are dominated by hardwood species characteristic of northern forests, such as beech, sugar maple, and yellow birch. The understory tends to be moist, and dominated by ferns.

Disturbance in northern hardwood forests is primarily due to wind (Lorimer and Frelich 1994). High rainfall and soil moisture keep fuel moisture relatively high, and fire has probably been infrequent, with return intervals between 300 and 500 years (Lorimer 1977). Because of the infrequent fire intervals and the overall resistance of the ecosystem to burning, fire does not appear to be an important element of these forests, and northern hardwood forests do not appear suitable for prescribed fire.

**Spruce-fir forests**—Spruce-fir forests occur at the highest elevations in the Southern Appalachians, generally above 5,000 feet. These forests are dominated by Fraser fir and red spruce, with thick litter and relatively few understory plants. Growing seasons are short, and the weather is characterized by abundant moisture, high humidity, and frequent cloud cover. The disturbance regime includes wind and ice storms.

These forests are structurally similar to boreal forests, and large, high-severity fires may occur during periods of prolonged drought (White and others 1985). More recently, acid precipitation and balsam woolly adelgid infestations have resulted in large-scale mortality of canopy trees, creating hazardous fuel conditions, and areas disturbed by ice or the adelgid may contain abundant coniferous regeneration capable of carrying intense fire (Smith and Nicholas 2000). On the other hand, fire frequency is very low, with estimated return intervals reaching into the millennia (White and others 1985). Given the low fire frequency and the extremely moist conditions, spruce-fir forests do not appear suitable for prescribed fire.

In general, therefore, fire has played an important role in many Southern Appalachian ecosystems, but these ecosystems tend to be the more xeric ones at mid-elevations, typically dominated by oaks or pines. Because the more mesic ecosystems in the Southern Appalachians, such as cove forests, are still subject to catastrophic fire under drought conditions, prescribed burning may be useful, under carefully controlled conditions, to minimize the likelihood of catastrophic fires in the future. In general, however, as ecosystem moisture increases, and elevations approach the extremes of the Southern Appalachian landscape, fire becomes increasingly less frequent, more likely to occur during drought, more catastrophic when it occurs, and therefore increasingly less suitable as a management tool.

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## 1.7 If we don't use fire as a management tool, what else do we use?

Some desirable changes to Southern Appalachian ecosystems are better achieved by treatments other than fire. For example, bat activity (Loeb and Waldrop 2008) and diversity (Leput 2004) are more likely to be improved by overstory thinning than a single prescribed fire. Thinning also increases coarse woody debris on the forest floor (Waldrop and others 2004), which may improve habitat for small mammals. High-intensity fire can accomplish the same objective by killing overstory trees (Waldrop and others 2010), but with less predictable results.

On the other hand, fire, especially low-intensity, high-frequency fire, has been a component of the Southern Appalachians for many years (Wade and others 2000). Many of our native ecosystems are fire-adapted, especially drier forests dominated by pines or oaks, and these forests have proven to be unstable in the 80 years since fire suppression became widespread in the Southern Appalachians (Nowacki and Abrams 2008). Ultimately, the role of fire in restoring and maintaining native ecosystems is unique:

- Fire kills and consumes a portion of the aboveground vegetation with very little impact to the mineral soil.
- Fire rapidly recycles nutrients back into the ecosystem.
- Fire improves seed germination by removing thick layers of duff and coarse fuels.

- Fire selectively removes fire-intolerant species, restoring ecosystem composition and structure.
- Fire improves wildlife value for many game species in the Southern Appalachians, including white-tailed deer, wild turkey, and ruffed grouse.
- Fire addresses restoration goals by increasing biodiversity, including understory plants (Waldrop and others 2008), reptiles (Kilpatrick and others 2010), and pollinating insects (Campbell and others 2007).

Other stand-level treatments, such as thinning overstory trees, felling understory shrubs, and herbicide applications, can mimic the changes to ecosystem structure obtained from prescribed burning, but all of these treatments add fuels to the forest floor, and greatly increase the risk of severe wildland fire over a period of ≥5 years (Waldrop and others 2010).

In summary, the combination of selective responses between fire-tolerant and intolerant species, the removal of forest floor material to allow seed germination, and the addition of nutrients to surface soil cannot be fully replicated using any other fuel treatment or management action. Returning fire to fire-adapted ecosystems is vital to restoring, and improving, the health of our native ecosystems.

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## 1.8 How do we restore fire to ecosystems where it has been excluded for many years?

Fire has a long history in the Southern Appalachian landscape. Soil charcoal, tree-ring scars, and fire-adapted vegetation all provide evidence for the role of fire as a natural process over the past several thousand years (Aldrich and others 2010, Fesenmyer and Christensen 2010, Flatley and others 2013, Zobel 1969). Beginning in the early 20th century, however, land managers in the Southern Appalachians began to prevent or suppress forest fires, effectively excluding fire from the landscape for nearly 80 years (Aldrich and others 2010, Flatley and others 2013). Long-term exclusion of fire has led to major changes in forest structure, function, and composition, particularly among forest types dominated by yellow pines and oaks. For example, excluding fire has increased the density of fire-sensitive trees and shrubs, which, in turn, have prevented pine and oak regeneration, shaded out grasses and forbs, and reduced the diversity of vegetation across the Southern Appalachians (Harrod and others 1998, 2000; Turrill and others 1995).

Since the mid-1990s, land managers throughout the Appalachians have sought to use natural and prescribed fires to reverse the effects of fire exclusion. Fire exclusion, however, has contributed to a buildup of wildland fuels that make wildland fires more difficult to control, and that pose a threat to forest health: when these forests eventually burn, they often burn with undesirable intensity and/or severity (Reilly and others 2012; Vose 2000, 2003). As a result, land managers restoring fire in the Southern Appalachians face two interrelated questions: first, how to effectively reduce hazardous fuels, and second, how to restore fire-dependent communities, especially pine, pine and oak, and oak forests, while minimizing undesirable effects.

**Hazardous fuels**

Wildland fuels in Appalachian forests fall into two general categories—live and dead. Live fuels consist primarily of evergreen shrubs, particularly mountain laurel, that can pose serious problems for fire control, but do not typically contribute significantly to available fuels during landscape-level burns. Dead fuels, on the other hand, are flammable vegetation at or near the forest surface, such as leaf litter, duff, and woody debris. Organic duff is the most common form of dead fuel (50-70 percent of the total). Other dead fuels include litter (10-20 percent) and logs >3 inches in diameter (also called 1,000-hour fuels, 10-20 percent). These fuel classes are not consumed at the same rate by dormant-season burning (Jenkins and others 2011, Vose and others 1999, Waldrop and others 2010). Dormant-seasons burns, which occur in late winter and early spring, consume relatively high amounts of litter, but most of the heavier, longer-burning fuels are not consumed.

In contrast, Jenkins and others (2011) found late summer and fall burns consumed a much higher percentage of duff and 1,000-hour fuels. These growing-season burns generally coincided with the annual peak of the drought index for the region (as measured by the Keetch-Byram Drought Index; Keetch and Byram 1968). Although higher levels of heavy fuel consumption were associated with successful pine regeneration, they were also strongly correlated with higher levels of mortality in the pine and oak overstory, which led to large increases in fuel loading as dead trees fell to the ground. In addition, growing-season burns and wildland fires frequently increase the rate at which nonnative plants invade the community (see Kuppinger 2008).

**Pine and oak restoration**

Specific objectives for restoring pine and oak communities usually center on reducing the abundance of fire-sensitive trees and shrubs, increasing pine and oak regeneration, and increasing the abundance of grasses and forbs. Several burning techniques have been used to achieve these objectives, with mixed results.

Single, and even multiple, low-intensity burns (backing/flanking fires with flame length <3 feet) during the dormant season have not achieved objectives for pine and oak restoration (Chiang and others 2005, Elliott and Vose 2005a, Jenkins and others 2011). In general, pine and/or oak regeneration did not increase following low-intensity burns, and, although all of the studies documented initial reductions in fire-sensitive trees and shrubs, these and other studies also documented prolific and repeated basal resprouting for many of these species.

High-intensity burns (headfires with flame length >8 feet) have also been used during the dormant/early season to address pine and oak restoration objectives. A common response to high-intensity, early-season burns, which has not been widely reported, is for these fires to kill large numbers of overstory trees, creating large, stand-level gaps that subsequently become dominated by hardwood resprouts. This can happen with fires at any time of the year, although pines can regenerate after late-season fires where a seed source exists (Jenkins and others 2011). High-intensity burns have been shown to be successful in regenerating Table Mountain pine (Waldrop and Brose 1999), and may contribute to oak regeneration in formerly pine-dominated sites (Elliott and others 2009). In general, however, these types of fires are not effective in regenerating oak stands, and are not recommended for restoration projects, due to concerns about fire control, burn effectiveness, and the loss of seed trees (Brose and others 2006, Elliott and others 2009, Jenkins and others 2011, Waldrop and Brose 1999).

In summary, we have found the combination of vegetation change and fuel accumulation attributed to fire exclusion, coupled with the topographic complexity of the landscape and the operational constraints in applying fire, poses a conundrum for land managers in the Southern Appalachians.

**Burning too hot**—Burning too hot (high-intensity or dry-season burning) may reduce high levels of fuels and establish pine regeneration, but will typically produce undesirable levels of overstory mortality, which, in turn, will increase forest fuel loading, decrease the oldest structural components in the stand, exacerbate the loss of seed trees, increase the growth of undesirable hardwood resprouts through overstory release, and facilitate the invasion of nonnative plant species.

**Burning too cool**—Burning too cool (low-intensity or dormant-season burning) may avoid some of the negative effects of high-intensity burning, but could produce less-than-desired fire spread, fuel reduction, and reductions in fire-sensitive trees and shrubs. As a result, restoration objectives such as pine regeneration may not be met in a timely fashion, especially while seed trees continue to diminish across the landscape.

**A fire restoration strategy**

What, then, is the best course of action for reintroducing fire into long-unburned sites to restore oak and pine communities in the Southern Appalachians? Because the consequences of burning too hot are far more difficult to overcome than the consequences of burning too cool, we believe the most cautious approach is to begin with cool, dormant-season burns, and to gradually increase burn severity by varying burn season and fire intensity. Our experience in the Great Smoky Mountains has led us to a multiple-burn strategy with the following features:

- Frequent burning (2- to 7-year intervals) over a 20-year period.
- Gradual reduction of heavier fuels such as duff and coarse woody debris.
- First- and second-entry burns that are primarily low intensity (flame length <3 feet) and occur in the dormant or very early growing season. The goal for first- and second-entry burns is to avoid creating stand-level (>2 acre) canopy gaps that increase fuel loads and release fire-sensitive resprouts. This is most important during first entry.
- Subsequent burns that increasingly use variable intensity and seasonality to reduce fuels and achieve desired community structure and composition.
- Some form of monitoring that can be the basis for adaptive management. Monitoring can include simple visual assessments, burn severity maps, or various types of plots.

Restoring fire to long-unburned sites is a long-term process that will require the insight of fire researchers, as well as the experience and skill of fire managers. The strategy outlined above is, at best, an informal consensus, and may not be applicable to the broad range of regional fire management objectives or site-specific conditions. Additionally, this management approach does not consider the use of mechanical treatments, which may contribute to the rapid creation of desired forest structure, but which are typically limited to small-scale projects. Rather, this strategy constitutes a general set of guidelines that should allow managers to achieve long-term vegetation and fuels objectives across broad landscapes while avoiding negative outcomes. The most important principles embedded in this strategy are the reliance on moderation, patience, and adaptive management in the application of varied combinations of fire intensity and seasonality to meet regional goals for fuels management and pine/oak restoration.

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## 1.9 What are the consequences if we don't use fire?

Fire was widespread and frequent throughout much of the Eastern United States both before and after European settlement (Abrams 1992, Fowler and Konopik 2007, Van Lear and Waldrop 1989). Beginning in the 1920s, however, fire was actively suppressed, changing plant communities across the region (Clark 1990, Wolf 2004). In oak and pine communities, these changes combined to produce dense forests dominated by mesophytic tree species.

**Changes in community structure**—Fire suppression changed the structure of many plant communities, as open, fire-dependent communities such as prairies and savannas were invaded by woody species (Abrams and Nowacki 1992). Compared to presuppression communities, oak-pine communities are now structurally dense, with stem densities as much as 10 times higher (Nowacki and Abrams 2008). Higher tree densities have increased stand basal areas, despite declines in average tree diameters, because the stands contain many more trees in smaller size classes (Fralish and others 1991).

**Changes in community composition**—In the absence of fire, mesophytic tree species, such as yellow poplar, maple, and cherry, tend to be competitively superior to more xeric oak and pine species. Fire suppression allowed fire-sensitive, shade-tolerant mesophytic species to replace more fire-dependent, shade-intolerant oaks and pines (Nowacki and Abrams 2008).

Changes in community composition affect the future role of fire in the community, because increases in mesophytic tree species decrease the likelihood of fire (Abrams 1992, Nowacki and Abrams 2008). For example, the high leaf area of shade-tolerant, mesophytic species casts heavy shade and limits air movement, decreasing wind speeds, increasing relative humidity, and creating a moist, cool forest floor (Nauertz and others 2004). Shady, moist conditions reduce understory flammability both directly, because the community retains moisture more effectively, and indirectly, because the additional moisture promotes the decomposition of forest fuels.

**Changes in fuel loads**—Fire suppression also changed the fuels in oak and pine communities (Washburn and Arthur 2003). Compared to the leaves of mesophytic trees, oak leaves are typically thicker, stiffer, and more resistant to decomposition (Abrams 1990, Carreiro and others 2000). Their rigid and irregular structure allows oak leaves to dry more effectively, and remain dry over a longer period of time, than mesophytic leaves, improving aeration, and therefore flammability, in the litter layer (Scarff and Westoby 2006). Mesophytic leaves, on the other hand, tend to lie flat and adhere to the forest floor, trapping moisture, minimizing air pockets, and enhancing decomposition (Lorimer 1985, Van Lear 2004). Oak leaves also contain high amounts of lignin, which delays decomposition, allowing oak leaves to remain in the litter for a relatively long time (Cromack and Monk 1975). The leaf litter produced by mesophytic tree species tends to contain small amounts of lignin, and the leaves decompose rapidly into a moist organic layer that is more likely to resist burning (Nowacki and Abrams 2008, Washburn and Arthur 2003).

Fuel loads are also determined by the amount of woody debris in the stand. In general, woody debris that is dry and retained in the community for a long time increases the flammability of the stand (Nowacki and Abrams 2008). The decomposition rates of woody debris, however, follow the rates of leaves, with oak and hickory debris decaying at the slowest rates, followed by beech, then maple (MacMillan 1988). Tyrrell and Crow (1994) reported that oak logs (with a half-life of 40 years) and pines (13-16 years) were more resistant to decay than mesophytic species such as maple (6-15 years).

All of these changes—increases in stand density, shifts in community composition, and decreases in fuel loads—reduce the flammability of oak and pine communities in the Southern Appalachians. This process—fire suppression leading to increases in mesophytic species that, in turn, reduce the flammability of the community—has been called mesophication (Nowacki and Abrams 2008). It appears to be a common outcome in oak and pine forests wherever fire has been suppressed (Bond and others 2005). The more mesic and fertile the ecosystem, the more rapidly it will undergo mesophication (Nowacki and Abrams 2008).

Once communities become mesophytic, however, returning fire and fire-adapted communities to the landscape can be challenging, due to the increased difficulty of burning, the loss of fire-adapted species, and the increased costs associated with the restoration (Abrams 2005). As a result, the mesophication of Southern Appalachian forests, especially oak-pine forests, is likely to continue (Nowacki and Abrams 2008).

In summary, fire suppression, especially in oak-pine and pine communities, has produced structural and compositional changes in Southern Appalachian forests that have led to a more mesophytic condition. This so-called mesophication of oak and pine forests becomes a positive feedback loop because mesophytic trees species produce leaf litter and woody debris that is less likely to burn than oaks and pines, further suppressing fire. In the absence of prescribed fire, we expect mesophication of oak and pine forests to continue, increasing the challenges facing land managers as they attempt to restore oak and pine forests in the Southern Appalachians.

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# 2. EARLY SUCCESSIONAL HABITAT IN THE SOUTHERN APPALACHIANS

Succession is an orderly progression of changes in community characteristics following disturbance, from early successional communities characterized by open conditions and ruderal species, to late successional communities characterized by closed conditions and competitive species (Odom and Barrett 2005, Odum 1969). As plant communities undergo succession, so do wildlife communities: some wildlife requires early successional communities for habitat, while other wildlife requires mid- or late successional habitats.

Ecologists often distinguish between primary succession, on sites previously unoccupied by vegetation, and secondary succession, on sites previously occupied by vegetation. Secondary succession typically follows some form of disturbance to the existing plant community, and the nature of the disturbance influences the composition, structure, and trajectory of successional change over time. In the Eastern United States, secondary succession has traditionally been associated with abandoned agricultural fields (e.g., Cadenasso and others 2008). Because these abandoned fields had been cleared of native vegetation, they exhibited dramatic changes in both species composition and community structure as they underwent succession to native forest. In the forested communities of the Southern Appalachians, however, succession is often initiated by active management and forest regeneration. Because managed forests regenerate largely via stump sprouts and localized seedlings, the composition of a regenerating forest is usually quite similar to the original forest. In this context, the term “succession” is frequently used to describe changes in community structure.

As a result, recent studies of both succession and early successional habitat have focused on functional characteristics of successional communities. Lorimer (2001), for example, distinguishes between early successional habitat, which he defines as communities dominated by species characteristic of highly disturbed areas, and young forest habitat, which he defines as young stands of later successional, forest species. More recently, Greenberg and others (2011a) have used the term early successional habitat to describe a wide variety of communities characterized by (1) an absence of a closed, mature tree canopy; (2) a well-developed ground or shrub component; and (3) young trees.

In the context of forest management and restoration in the Southern Appalachians, early successional habitat is primarily a wildlife resource, produced by either timber harvests or active management, such as the construction of wildlife openings. Compared to undisturbed forests, regenerating forests share many characteristics with traditional forms of early successional habitat, such as small, rapidly growing trees and understory species associated with open conditions. As a result, we believe young, regenerating forests can be functionally equivalent to more traditional forms of early successional habitat, especially for many wildlife species, and especially during the first several years following harvest.

For the purposes of this paper, we will use the wildlife definition of early successional habitat, typically considered a combination of regenerating forests and wildlife openings. When considered a wildlife resource, early successional habitat may also be produced by natural disturbances, such as wildland fire, windstorms, or the natural formation of canopy gaps.

*In researching these questions, Southern Research Station scientists identified a need for a symposium on early successional habitat in the Southern Appalachians. A 1-day symposium on was held on April 8, 2010, as part of the annual meeting of the Association of Southeastern Biologists in Asheville, North Carolina. This symposium addressed the ongoing decline of many plants and animals associated with early successional habitat in the Eastern Upland Hardwood Forest region. Presenters also synthesized the current knowledge about early successional forests and the wildlife associated with these habitats, further addressing many of the questions posed in this paper. A contributed volume that includes proceedings of the symposium, edited by Cathryn H. Greenberg, Beverly Collins, and Frank R. Thompson III, was published in summer 2011 and is available online at www.springer.com/life+sciences/ecology/book/978-94-007-1619-3.*

## 2.1 Why are early successional forests important habitats for wildlife species?

Early successional forests provide two critical resources for wildlife: habitat structure and cover, and forest foods.

**Habitat Structure and Cover**

Vegetation structure, measured both as the vertical strata within a forest and as the distribution of forest types across the landscape, has a strong impact on the diversity and composition of wildlife communities (MacArthur and MacArthur 1961, Thatcher and others

2007). Disturbance can enhance biological diversity, especially at the landscape scale, by creating a mosaic of habitats or successional stages (see Franzreb and others 2011, Shifley and Thompson 2011). Many wildlife species in the Southern Appalachians use early successional habitats to meet various biological needs, including foraging, hunting, nesting, rearing young, escape, thermoregulation, and protection from the elements (Dickson 2001). Other species use a variety of successional stages, but require early successional habitats during a particular biological season or time of year. Some wildlife species do not require early successional habitats, but are more abundant in these habitats, and, in general, their populations and individuals are healthier when a variety of successional stages are available (Fuller and DeStephano 2003).

**Birds**—Many bird species characteristic of early successional habitat in the Southern Appalachians are declining as abandoned pastures and farmlands return to forest and existing forests mature (Askins 2001, Franzreb and others 2011, Shifley and Thompson 2011). Several studies report higher bird species richness, diversity, and density in sites that were disturbed by management activities (Annand and Thompson 1997, Baker and Lacki 1997) or natural disturbance (Blake and Hoppes 1986, Greenberg and Lanham 2001) compared to mature, undisturbed forest.

Bird species associated with early successional habitat include eastern bluebird, American goldfinch, chestnut-sided warbler, golden-winged warbler, yellow-breasted chat, eastern meadowlark, field sparrow, northern bobwhite, loggerhead shrike, and indigo bunting. Different species, however, are associated with different types of early successional habitat. For example, meadowlarks are found in and around herbaceous openings relatively clear of woody encroachment (Roseberry and Klimstra 1970), while golden-winged warblers use herbaceous openings with considerable shrub/bramble encroachment (Klaus and Buehler 2001, Litvaitis 2001). When these areas succeed into dense thickets and brush, yellow-breasted chats and brown thrashers will be present (Burhans and Thompson 1999, Stauffer and Best 1980). Other species, such as chestnut-sided warblers, are found primarily in very young forest (<8 years) with high stem densities (King and Byers 2002), while ruffed grouse are found primarily in young forest (6-20 years; Jones and others 2008, Tirpak and others 2010).

Other bird species use early successional habitat periodically during the year. For example, wild turkeys in the Southern Appalachians preferentially use openings dominated by forbs for brooding (McCord and Harper 2011). American woodcocks perform courtship rituals in grassy openings during the winter, but forage in young forest stands inside riparian zones (Dessecker and McAuley 2001). Cerulean warblers usually nest in mature closed-canopy stands, but forage in canopy gaps (Weakland and Wood 2005). Many bird species associated with mature forest commonly bring their fledglings to early successional forest, presumably because these habitats offer high-quality foraging as well as dense, protective cover (Greenberg and Lanham 2001, Whitehead 2003).

**Mammals**—Mammals associated with early successional habitat in the Southern Appalachians include the Appalachian cottontail, hispid cotton rat, and groundhog. White-tailed deer commonly use herbaceous openings with shrubby cover, which may influence fawn survival (Beier and McCullough 1990, Piccolo and others 2010). Young forests are an important source of deer browse during spring and summer for approximately 7 years after regeneration harvest (Johnson and others 1995). Canopy closure, however, eventually reduces browse in these regenerating forests (Johnson and others 1995), and the stands are not important food resources for deer until hard mast becomes available. Black bears will also frequent openings and young forest stands during summer when soft mast is available. The availability of early successional habitat may reduce bear movements and has been linked to smaller home ranges (Litvaitis 2001). Several species of eastern bats increase foraging activity in recently harvested stands (Loeb and O'Keefe 2011, Tichenell and others 2011).

**Herpetofauna**—Salamanders that require cool, moist conditions generally decline in clearcuts because of their increased risk of desiccation, and populations may take years to recover (Petranka and others 1993, 1994; Tilghman and others 2012). On the other hand, Raybuck (2011), working in western North Carolina, did not detect a significant change in salamander abundance after shelterwood harvests or other silvicultural treatments designed to promote oak regeneration, such as prescribed fire. Some reptile populations, such as fence lizards, increase in recently disturbed areas, likely because of improved opportunity for thermoregulation and foraging (Greenberg 2001, Moorman and others 2011, Russell and others 2004).

**Forest Foods**

Young, upland hardwood forests also function as high-quality food patches by providing abundant fruit, nutritious foliage, and flowers that attract pollinating and foliar arthropods, which, in turn, support high populations of small mammals. These small mammals and arthropods become prey for numerous vertebrate predators. In the Southern Appalachians, many species of native birds and other vertebrates forage opportunistically in young stands for fruit and arthropods.

**Fleshy Fruit**—Native fleshy fruit is a key food resource for both game and nongame wildlife (Martin and others 1951). Most birds and mammals consume fruit at least occasionally (Martin and others 1951, Willson 1986), and fruit can be a critical resource for both migratory birds in the fall (Willson 1986) as well as resident and overwintering birds in the winter, when arthropods and other forest food sources are scarce (Borgmann and others 2004, Greenberg and Forrest 2003, Kwit and others 2004, McCarty and others 2002, Whitehead 2003). Fruit consumption has also been linked to mammalian survival and reproductive success (Eiler and others 1989, Rogers 1976).

Compared to closed-canopy conditions, fleshy fruit production is much greater in forest openings caused by either natural disturbance (e.g., Blake and Hoppes 1986, Thompson and Willson 1978) or by silvicultural disturbance such as harvesting (e.g., Greenberg and others 2007a, Mitchell and Powell 2003, Perry and others 1999). Fruit production in the Southern Appalachians closely correlates with stand age and achieves its highest levels during the first 10 years following two-age harvest (Greenberg and others 2011b). Greenberg and others (2007a), for example, found dry pulp biomass of fleshy fruit was similar in young two-age stands and mature forest during the first 2 years following harvest, but by year three fruit production was 5.0 to 19.6 times higher in young stands compared to mature forest. Canopy openings facilitate the establishment of disturbance-mediated species such as pokeweed and blackberry, both of which produce large amounts of fleshy fruits, and increases in available light, moisture, and space provide optimal growing conditions for many fruit-bearing plants, including species typically associated with mature forests that may already be present in the disturbed areas. For example, huckleberry is a prolific fruit producer in mature forests and recovers rapidly after harvest (Greenberg and others 2007a, Powell and Seaman 1990). Stump sprouts of dogwood, American holly, Fraser magnolia, black cherry, sassafras, and blackgum also produce fruit 1-3 years following harvest. As a result, land managers can enhance fruit availability for many game and nongame species by creating or maintaining patches of young forest (see Greenberg and others 2007a, 2011b).

**Arthropods**—Arthropods play important ecological roles as predators and prey (Hammond and Miller 1998), decomposers (Moldenke and Lattin 1990), nutrient cyclers (Asquith and others 1990), herbivores (Wilson 1987), and pollinators (Westman 1990). They also represent a large proportion of biological diversity and support invertebrate and vertebrate diversity by serving as an important food resource. In the Southern Appalachians, leaf litter-dwelling arthropods are often positively correlated with leaf litter depth, moisture, or both (Duguay and others 2000, Greenberg and Forrest 2003, Harper and others 2001, Haskell 2000, Whitehead 2003). In general, litter-dwelling arthropods are more abundant and compose more biomass in mature forests, where shade and thick leaf litter provide a cooler, moister microclimate compared to young forest. Flying and foliar arthropods, however, are more abundant in young stands where tree density and canopy cover are low (Whitehead 2003), most likely due to higher concentrations of young, palatable foliage on shrubs, tree sprouts, and herbaceous plants. Pollinating insects tend to be most abundant in open, disturbed conditions, such as clearcuts (Healy 1985), recently burned forests with dead trees (Campbell and others 2007), roadsides planted with clover or orchard grass (Hollifield and Dimmick 1995), and herbaceous wildlife openings (Harper and others 2001), because these areas contains the highest concentrations of flowering plants (Greenberg and others 2011b).

**Forage**—Forage, which includes herbaceous plants and woody browse, is important for several wildlife species, including white-tailed deer, cottontails, black bears, wild turkeys, and ruffed grouse. Forage is most readily available in constructed wildlife openings, which can provide 1,000-10,000 pounds of forage per acre per month (Harper 2008). The amount of forage in forests, on the other hand, is influenced by the amount of light penetrating the forest canopy (Ford and others 1993; see Greenberg and others 2011b). Closed-canopy forests in the Southern Appalachians typically provide 50-150 pounds of dry weight forage per acre within 4.5 feet of the ground, while recently harvested stands provide approximately 1,000 pounds of dry weight forage per acre (Beck and Harlow 1981, Della-Bianca and Beck 1985). Available forage declines significantly 5-7 years after harvest because the developing canopy closes the opening (Beck and Harlow 1981, Johnson and others 1995).

Management practices that allow at least 30 percent light to penetrate the canopy also increase forage availability. For example, shelterwood harvests will typically double or triple forage availability compared to closed-canopy conditions, and, when followed by low-intensity prescribed fire, may provide a sevenfold increase in forage availability (Jackson and others 2007, Lashley and others 2011, Shaw and others 2010).

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## 2.2 What species are favored by early successional habitat and how strong is the relationship?

Many species of birds, mammals, and reptiles use early successional habitat in the Southern Appalachians. Some species use openings seasonally, or for specific needs such as foraging. Other species use these areas year-round, or for multiple needs, such as nesting, escape, foraging, and courtship.

**Birds**—Historically, almost the entire Appalachian region was forested, with disturbed areas created by natural factors such as canopy gaps, fires, insect infestations, grazing by native species, and localized weather conditions such as hurricanes, tornadoes, and ice storms. Askins (1995), however, argues that native grassland and other successional habitats were an integral part of the pre-European landscape. Native American agricultural practices are believed to have profoundly affected the landscape, primarily from slash-and-burn methods that cleared forest for agricultural use. As crops depleted soil nutrients, Native Americans abandoned old cropland and cleared additional lands, creating a presettlement landscape consisting of forests and fields in varying stages of succession (Askins 1999, Patterson and Sassaman 1988). These abandoned croplands would have provided critical habitat for disturbance-dependent species. In addition, forest understory was burned by Native Americans to improve conditions for hunting small game and deer (Van Lear and Harlow 2000).

Many disturbance-dependent bird species are associated with more than one type of successional community, and may use a combination of communities such as savannas, woodlands, gaps in mature forest, and shrubby cover. In Eastern North America, 128 bird species considered rare, or believed to be declining in much of their range, are associated with these communities (Hunter and others 2001). Rare or declining species that use disturbance-maintained habitats include the Appalachian yellow-bellied sapsucker, eastern and Appalachian Bewick's wren, golden-winged warbler, cerulean warbler, Swainson's warbler, and indigo bunting.

Due to recent large-scale reductions in the amount and distribution of early successional habitat, most disturbance-dependent birds have experienced a decrease in population numbers, which is expected to continue (Franzreb and others 2011, Hunter and others 2001, Shifley and Thompson 2011). According to Breeding Bird Survey data from 1966 to 1994 for species associated with shrub cover, 10 species (58.8 percent) in the Ridge and Valley, 8 species (50.0 percent) in the Cumberland Plateau, and 9 species (61.5 percent) in the Blue Ridge Mountains Physiographic Regions appeared to have undergone significant population declines (Franzreb and Rosenberg 1997). Early successional habitat may also be important to species usually associated with mature forest (Pagen and others 2000) because some species may seek patches of disturbed habitat at least during part of their life cycle (Anders and others 1998, Vega Rivera and others 1998).

**Mammals**—Early successional habitat, particularly gap openings, can be important foraging areas for several bats in the Southern Appalachians (Loeb and O'Keefe 2011). Larger species such as the hoary bat, silver-haired bat, big brown bat, and eastern red bat often forage in areas with reduced clutter, such as wildlife openings, small cut areas, and gaps within intact forest (Krusic and others 1996, Loeb and O'Keefe 2006, Owen and others 2004). Even some smaller bats such as pipistrelles frequently use openings (Loeb and O'Keefe 2006, Schirmacher and others 2007). The northern long-eared bat, however, avoids openings and prefers closed-canopy forests, presumably because its echolocation and flight characteristics allow the species to forage effectively in forests (Loeb and O'Keefe 2006, Patriquin and Barclay 2003).

In contrast, early successional communities are rarely used for roosting. Male eastern red bats occasionally roost in early successional habitat in the Southern Appalachians (O'Keefe and others 2009), but most bats in the Southern Appalachians roost in trees in mature forests during the summer and hibernate in caves or migrate south during the winter. Early successional forests, therefore, are not important habitats for bat roosting ecology. Because early successional forests are an important foraging habitat for some species, however, the presence of these forests may influence roost site selection. For example, male eastern pipistrelles in the Southern Appalachians roost closer than expected to small, nonlinear openings and two-age harvest areas <5 years old, and red bats roost closer than expected to linear openings such as gated roads and trails (O'Keefe and others 2009). As a result, small openings scattered among mature forests may be beneficial to many bat species in the Southern Appalachians.

Mammals other than bats appear to be less obligated to early successional forests. Litvaitis (2001) suggests that a few mammals are habitat specialists (e.g., eastern cottontails). Some carnivores (e.g., bobcats) rely on early successional habitat for prey items, and a few other mammals (e.g., black bears) depend seasonally on the abundant fruit found in many forest openings.

**Reptiles**—Canopy removal results in higher light levels; warmer, drier microclimates; and reduced leaf litter cover on the forest floor. These conditions, especially warmer temperatures, appear to benefit reptiles (Adams and others 1996, Greenberg 2002, Phelps and Lancia 1995; see also

Moorman and others 2011). Most reptile species require warm temperatures for egg incubation and the successful development of hatchlings (Deeming and Ferguson 1991, Goin and Goin 1971). Hotter, drier microclimates can also facilitate movement and thermoregulation for many reptile species. In the Southern Appalachians, lizards in general, and fence lizards in particular, may become more common in sites with reduced canopy cover, including clearcuts (McLeod and Gates 1998) and large canopy gaps (Greenberg 2001). Lizards may also become more common after one (Greenberg and Waldrop 2008) or two (Matthews and others 2009) prescribed burns.

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## 2.3 Are some wildlife species negatively affected by creating early successional habitat?

Several studies in the eastern hardwood forest suggest that heavy canopy removal treatments such as clearcuts or shelterwoods can adversely affect the local abundance of terrestrial salamanders (Ash 1988, 1997; deMaynadier and Hunter 1995; Harpole and Haas 1999; Petranka and others 1993, 1994; Pough and others 1987; Reichenbach and Sattler 2007; Russell and others 1999, 2004). Canopy removal produces higher light levels; warmer, drier microclimates; and reduced leaf litter cover, which can desiccate salamanders (Crawford and Semlitsch 2008a, 2008b; deMaynadier and Hunter 1995; Renken 2006; Russell and others 2004). Following clearcutting, salamanders and frogs can retreat underground for short periods, evacuate the harvest area, or perish (Semlitsch and others 2008), and declines in salamander populations may not be fully realized for 2-3 years following clearcutting (Ash 1988). In the Southern Appalachians, salamanders may virtually disappear from sites following clearcutting, and may not return to preharvest levels for 20 years (Ash 1988, 1997; Petranka and others 1993, 1994; but see also Adams and others 1996; Harper and Guynn 1999). Recovery time for salamanders tends to vary with leaf litter, which, in turn, appears positively correlated with community moisture levels—mesic forests recover faster than dry forests (Moorman and others 2011). Full recovery, however, may take many years. Crawford and Semlitsch (2008b), for example, found stream salamanders and their terrestrial habitat were less common in Southern Appalachian forest stands <40 years of age compared to stands >41 years of age. Tadpoles of some frog species may develop faster or survive better in ponds inside clearcuts (Semlitsch and others 2009), but juvenile and adult stages still require forested habitat that may require many years to recover.

On the other hand, Raybuck (2011), working in western North Carolina, did not detect a significant change in terrestrial salamander abundance after shelterwood harvests. Frogs and toads do not appear to be strongly affected by forest disturbance (Moorman and others 2011), and disturbances that retain heavy canopy cover, such as midstory removal, two-age harvests, selection harvests, firewood cutting, thinning, heavy browsing by deer, and low-intensity burns, do not affect terrestrial salamander abundance (Adams and others 1996, Brooks 1999, Floyd 2003, Ford and others 1999, Greenberg and Waldrop 2008, Harpole and Haas 1999, Homyack and Haas 2009, Knapp and others 2003, Matthews and others 2009, Messere and Ducey 1998, Moseley and others 2003, Pough and others 1987). Small harvest area and connections to high-quality habitat can also mitigate impacts and provide refuge and recolonization sources.

Harvest effects on stream-breeding salamanders may also be mitigated by the use of riparian buffer zones. Peterman and Semlitsch (2009) found that larval salamanders in streams were negatively impacted by 30-foot riparian buffers, likely due to increased stream sedimentation, and suggested 100-foot buffers to reduce the effects of timber harvest on larval salamanders in Southern Appalachian streams. Crawford and Semlitsch (2007) recommended a 300-foot streamside buffer zone to provide core terrestrial habitat for stream-breeding salamander species in the Southern Appalachians, many of which use terrestrial, riparian habitat during much of their adult lives.

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## 2.4 Do tree species differ in their response to early successional habitat?

Although tree species most likely form a gradient in their response to canopy disturbance, canopy trees are traditionally separated by foresters into three groups: species tolerant of overstory shade, species somewhat tolerant of overstory shade, and species intolerant of overstory shade (Baker 1949). Tolerant species, such as beech and eastern hemlock, tend to grow slowly, but persistently, under closed-canopy conditions. Intolerant species, on the other hand, include many desirable timber and wildlife species, such as red oak and yellow-poplar. Compared to tolerant species, intolerant species grow vigorously in open conditions, but tend to grow very poorly beneath closed canopies and are unlikely to persist without some form of canopy disturbance. Species of intermediate tolerance, such as black cherry and white pine, are able to grow slowly under partial shade. Although species in all three groups may respond positively to gap formation, open conditions created by forest management, such as timber harvest, generally favor intolerant tree species at the expense of tolerant tree species.

This basic concept, however, can be influenced by a variety of factors, most notably by the preharvest composition of a stand. Following clearcut or two-age harvest of a mature forest in the Southern Appalachians, the initial composition of the regenerating stand is largely a reflection of the preharvest stand, especially on dry sites. With few exceptions, most Appalachian hardwood species with commercial value are intolerant of shade and require well-developed root systems to begin, and maintain, the height growth necessary to achieve and hold a canopy position. In addition, most hardwood species produce prolific basal sprouts when the main stem is cut, and utilize nutrients stored in their roots to quickly initiate height growth. As a result, tree species that are not present in the stand at the time of the harvest do not have an adequate seed source, are unable to grow quickly following germination, and are unlikely to become established in the regenerating stand.

On moist sites, canopy composition of the postharvest stand at crown closure often differs from the preharvest stand because rapid height growth from some mesophytic species, particularly yellow-poplar and red maple, are capable of producing intense competition (Beck and Hooper 1986). In addition, the ability of seedlings to grow quickly in height following release from competition, especially for seedlings of yellow-poplar and sweet birch, is another important characteristic that partially determines the eventual composition of early successional forests (Loftis 1990a). As a result, both shade tolerance and the growth response of tree species in the preharvest stand must be considered when predicting the outcome of silvicultural prescriptions.

Tree species may also differ in their response to open conditions created through natural disturbances. Generally, gap phase reproduction correlates poorly with the reproduction of tree species (Bray 1956), although at least two studies have reported significant correlations between gap size and tree reproduction. Rankin and Tramer (2002a), working in the Appalachian province of southeastern Ohio, reported significant correlations between gap size and the tree species successfully colonizing former canopy gaps. Dale and others (1995), working in hardwood forests in the Midwest, reported shade-tolerant species were more frequent in smaller openings, and the proportion of intolerant species increased as the size of the opening increased. In addition, oaks and hickories were more common in openings on the poorer sites, while yellow-poplar was more common in openings on the better sites.

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## 2.5 How much early successional habitat is necessary to provide for biodiversity? What is the optimal percentage of early successional habitat to meet desired goals?

The answer to this question depends primarily on the scale at which biodiversity is measured. Biodiversity occurs at several levels: at the community level, along a gradient between communities, and at the landscape or regional level (Noss 1983, see also Whittaker 1972). Early successional habitat would most likely enhance biodiversity along gradients between communities (Sharitz and others 1992).

Presumably, the optimal amount of early successional habitat for a variety of wildlife species varies between the 6-9 percent of the forest that would occur in natural gap-phase disturbance at any given moment (Runkle 1982) and the 60 percent of the forest estimated by Lorimer (2001) at peak disturbance levels during the late 19$^{th}$ century.

Although the natural gap disturbance rate would seem to be the minimum necessary to provide for biodiversity, Hunter and others (2001) argue that natural disturbance regimes, especially gap-phase disturbances now largely absent from a landscape with few old-growth forests, are insufficient to maintain disturbance-dependent bird species. DeGraaf and Yamasaki (2003) suggest that to optimize early successional species diversity, early successional habitat should compose 10-20 percent of the forest landscape, or roughly twice the natural gap rate. This recommendation is very close to the 10- to 20-percent estimate of naturally occurring early successional habitat derived in Question 2.6, below.

Biodiversity, however, does not vary directly with early successional habitat (Shifley and Thompson 2011). Most of the plant diversity in Southern Appalachian forests occurs in the herb layer, which varies across several environmental gradients, including disturbance. In the Southern Appalachians, herbaceous diversity responds positively to overstory disturbance by increasing the percentage of shade-intolerant species in the herbaceous layer (Shifley and Thompson 2011). Shade-tolerant herbs also respond positively to at least some overstory disturbance, but the response is presumably brief and confined to a small window of time immediately following the disturbance, before shade-intolerant herbs colonize and dominate the site (see Rankin and Tramer 2000b). When overstory disturbance is severe, forest herbs may take decades to recover to pre-disturbance levels (Elliott and others 1997, 2011). Once lost, many forest herbs are severely limited by seed dispersal, and may take many years to recolonize a forest stand (Elliott and others 2011).

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## 2.6 What is the natural distribution of early successional habitat across the landscape?

For the purposes of this report, we define natural disturbance in the Southern Appalachians as any process creating early successional habitat that is not explicitly due to human actions. Using this definition, natural disturbances would include natural fires, windstorms, hurricanes, landslides, outbreaks of native insects, and autogenic forest dynamics, such as gap fall. This definition would exclude modern silvicultural actions, such as timber harvests and prescribed burning. Although we recognize the potential role of Native Americans in shaping the Southern Appalachian landscape, especially through the use of fire, we consider these affects anthropogenic in nature and exclude them from our definition of natural disturbances.

Evidence on the amount and distribution of natural disturbance are derived from several sources, including historical records, old-growth forest stands, pollen and charcoal deposits, and direct observation of contemporary disturbances (Lorimer 2001). While each of these lines of evidence can provide important information, each also has its limitations, and quantitative estimates of natural disturbance must be interpreted cautiously. Lorimer (2001) lists some of the challenges for determining natural disturbance patterns:

- Because of the long history of human occupation in the Southern Appalachians, evidence on natural and anthropogenic disturbance is easily entangled.

- Changes in global climate may alter disturbance regimes over geological time.

- Due to soil, topographic, and human settlement patterns, disturbance patterns on the landscape are spatially nonrandom.

- Although potentially long-lasting, severe disturbance is highly episodic and spatially heterogeneous, making estimates of its impacts problematic.

Because disturbance patterns depend upon biotic and abiotic factors that differ at regional scales, disturbance patterns also differ by geographic region (Runkle 1990). In the Southern Appalachians, natural disturbance patterns are often attributed primarily to gap-phase dynamics (e.g., Runkle 1981). Gap-phase disturbance in eastern deciduous forests averages about 1 percent of the landscape per year (Runkle 1982). Because canopy gaps remain in early successional habitat for approximately 6-8 years following formation, approximately 9 percent of the landscape is in some form of gap-phase reproduction at any time (Rankin and Tramer 2002a). Gap size varies by the number of trees involved in the disturbance, but, in the Appalachian region, generally ranges between 1,000 and 5,000 square feet (Rankin and Tramer 2002a).

Gap-phase disturbance, however, tends to occur in small, localized patches that retain forest characteristics (Runkle and Yetter 1987) and is generally not considered equivalent to more-extensive disturbances created by catastrophic events such as windstorms, infestations, and wildland fire (Shure and others 2006). For example, gap-phase disturbances tend to produce small openings that typically close more quickly than more extensive disturbances that create larger openings. Although qualitatively different, more-extensive disturbances are also characteristic of the Southern Appalachians, and may be just as important as gap-phase disturbance.

**Windstorms**—Greenberg and McNab (1998), using data compiled by Neumann and others (1993), report hurricane-related windstorms have recurred in the Southern Appalachians 14 times since 1871, at intervals ranging from 1 to 24 years. Hurricane Opal damaged approximately 0.3 percent of the national forest lands in northeastern Georgia, western North Carolina, and eastern Tennessee in 1995. Although Greenberg and McNab (1998) calculate that approximately 9.3 percent of the landscape would suffer some form of windthrow disturbance over the 200-year lifespan of many eastern forest trees, only a small portion of this time would be spent in the early successional phase. If this phase extends 20 years from the initial disturbance, then the amount of young forest produced by catastrophic windthrow would affect approximately 1 percent of the Southern Appalachian landscape at any given time.

**Infestations**—Infestations of native insects, such as the southern pine beetle, can be widespread and produce relatively large patches of young, regenerating forest. The U.S. Department of Agriculture, Forest Health Protection estimates the total amount of beetle infestation on a yearly basis. The estimates are not specific to the Southern Appalachians and are presumably based on partial activity within forest stands. For 2001, a year characterized by a high level of beetle activity, the total area of beetle infestations, estimated as one-half of the reported acres for Georgia, Tennessee, and North Carolina, is 1.6 million acres, or approximately 2-5 percent of the Southern Appalachian landscape.

Other insects also affect the Southern Appalachians, but many of these species, such as the hemlock woolly adelgid and the emerald ash borer, are invasive exotics, and presumably would not have contributed to the more natural, presettlement disturbance regime. Because these impacts have not been fully analyzed on a regional basis, they will not be considered further in this report. On the other hand, we recognize these species will have an extensive impact on the Southern Appalachians and may need to be considered in future analyses for early successional habitat.

**Fire**—High-intensity, catastrophic fire is rare in the hardwood forests of the Southern Appalachians. Despite extensive research over the past few decades and widespread acceptance that fire has been an integral part of the Southern Appalachian landscape over both historic and prehistoric periods, the amount of early successional habitat created by wildland fire cannot be adequately characterized. Guyette and others (2006) estimate return intervals for presettlement (about 1650-1850) eastern deciduous forest in the Southern Appalachians at around 10 years, although they caution that return intervals are highly variable in both time and space. The fire literature contains no quantitative estimates for historical levels of fire extent and severity in the Southern Appalachians, and no estimates regarding the amount of early successional habitat produced by fire. If we estimate the amount of early successional habitat produced by wildland fire as roughly the equivalent of the habitat produced by windstorms and insect infestations combined, the total amount of habitat present at any given moment would constitute between 3 and 6 percent of the landscape of the Southern Appalachians.

**Ice storms**—Ice storms occur regularly in the Southern Appalachians (Runkle 1985), with return intervals estimated around 20 years (Abell 1934). These storms can damage and kill overstory trees, producing canopy openings (Boerner and others 1988). Working in southwestern Virginia, Warrillow and Mou (1999) estimated a single ice storm damaged roughly 0.7 percent of a 1,200-acre forest, although localized storm damage can be much higher (Boerner and others 1988). Compared to other forms of landscape disturbance in the Southern Appalachians, however, the effects of ice storms are more variable, and therefore more difficult to categorize, especially in relationship to early successional habitat. If half the canopy damage caused by an ice storm produces early successional habitat, and if the early successional habitat remains for 10 years following the storm, then early successional habitat produced by ice storm damage may occupy around 1 percent of the Southern Appalachian landscape at any given moment.

**Other disturbances**—Other disturbances, such as landslides and floods, may produce locally severe disturbances. Compared to landscape-scale disturbances, such as gap-phase disturbance and hurricanes, however, the amount of early successional habitat created by landslides and floods appears negligible and therefore will not be considered further.

Compared to gap disturbances, catastrophic disturbances occur at longer intervals, are highly episodic, and vary markedly in time and space (Lorimer 2001, Greenberg and others 2011a). As a result, the relative influence of gap and catastrophic disturbances at a specific location is not always clear. Gap disturbance appears ubiquitous in mature, old-growth forest, and, if estimated at 8-10 percent of the original forest, would be the dominant form of disturbance in the mesic forests of the Southern Appalachians, especially in protected coves that are not prone to windstorms and severe fires. Catastrophic disturbance also appears ubiquitous in the Southern Appalachians, and, if estimated at 6-12 percent of the original forest, would be the dominant form of disturbance in more-exposed forests, such as ridges and dry slopes. Combining the two estimates, potentially 20 percent of the landscape would be in early successional habitat at any given time, and roughly half of the early successional habitat would be created by catastrophic events.

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## 2.7 Is there a threshold value for early successional habitat patch size?

Threshold values for early successional patch sizes vary by the species associated with the habitat. In addition, threshold values may depend on the context of the habitat—for example, the availability of nearby roosting or foraging habitat in mature forest may affect the species of bats found in early successional openings (Loeb and O'Keefe 2006). As a result, threshold values are closely related to both the species under consideration and the context of the habitat in the landscape. Quantifying these thresholds may be very difficult.

**Plants**—Rankin and Tramer (2002a, 2002b) used current and reconstructed canopy gaps in eastern hemlock forests in the Appalachian region of southeastern Ohio to examine the role gap size plays in the establishment of both overstory trees and understory herbs. They concluded that canopy gaps less than approximately 1,000-1,500 square feet play virtually no role in the establishment of trees, but gaps >1,500 square feet are strongly associated with tree recruitment. This association is connected with gap size: tolerant trees, such as hemlock and beech, are more likely to become established in medium-sized gaps, while intolerant trees, such as yellow-poplar, are confined to the largest gaps. This finding suggests at least some tree species in the Southern Appalachians exhibit different thresholds for gap size before they can successfully reproduce in natural canopy gaps. Understory plants, on the other hand, all respond to canopy gaps to some degree, but some species appear to utilize sub-gap disturbances to persist beneath closed canopies, suggesting that at least some understory herbs do not exhibit a lower threshold value for canopy disturbance.

Rankin and Tramer (2002a, 2002b) primarily addressed lower threshold values for a few species in the Southern Appalachians. Although we know of no study that explicitly addresses the possibility of upper threshold values for Southern Appalachian herbs, these values may be correlated with the minimum thresholds for the reproduction of shade-intolerant trees, because, at that point, the dominant flora presumably changes from species characteristic of forest environments to species characteristic of open environments. In the Northeastern United States, this threshold is roughly 2 acres of open conditions (DeGraaf and Yamasaki 2003).

**Wildlife**—Threshold values of early successional habitat for wildlife vary by species. Home ranges for early successional species vary from 3 acres for yellow-breasted chat (Thompson and Nolan 1973) to as much as 200 acres of young forest for male ruffed grouse (Thompson and Fritzell 1989). Many wildlife species, however, use early successional habitat as a portion of their home ranges, making threshold levels difficult to determine. Greenberg (2001) and Greenberg and Lanham (2001) suggest that gaps as small as 0.25 to 3 acres can provide important habitat for some early successional birds and reptiles. Based on their studies in the Northeast, DeGraaf and Yamasaki (2003) recommended group selection harvests of at least 2 acres, although small clearcut openings (20-30 acres) appear to meet the habitat needs of more bird species than group selection openings (Costello and others 2000). White-tailed deer will use small group selection harvests—typically, openings between 0.25 and 0.5 acres—for foraging and bedding, but larger harvests may be necessary to regenerate trees in the presence of high deer densities. Well-distributed harvests of approximately 20 acres have been recommended to avoid problems with tree regeneration due to deer grazing (Harlow and Downing 1969). Other studies have shown the number of bird species increases with increasing size of clearcuts, up to 50 acres (Rudnicky and Hunter 1993).

Single-tree selection—the harvest procedure that seems most similar to natural canopy gaps—does not create openings large enough to provide early successional habitats for birds (DeGraaf and Yamasaki 2003). For example, golden-winged warblers, one of the early successional species of most concern in the Southern Appalachians, tend to avoid patches smaller than 5 acres; their numbers tend to increase when patch sizes are between 30 and 100 acres (Buehler and others, unpublished data cited by Hunter and others 2001).

**Landscapes**—The most problematic values for early successional patch sizes occur at the landscape scale: How much of a forest, or a watershed, should be in early successional habitat? Again, this value depends upon the resource associated with the landscape. For example, black bears, the wildlife resource with the largest home range in Eastern North America, require a mixture of successional stages, using both mature forest and early successional areas during different times of the year. Even for bears, however, neither the optimal nor the threshold values of early successional habitat have been determined. Working in New England on Northern Appalachian forests, DeGraaf and Yamasaki (2003) suggest that, to optimize early successional species diversity, early successional habitat should compose about 10-20 percent of the forest landscape.

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## 2.8 How do we sustain early successional habitat in the landscape?

By definition, early successional habitat is transient and disappears from the landscape unless constantly produced by natural or anthropogenic means (Shifley and Thompson 2011). Since reaching peak levels of 60 percent in the late 19th century, young forest in North America has declined steadily, to ≤20 percent in many regions (Lorimer (2001). Due to shifts in land management, declining timber harvests, fire suppression, and urban sprawl, traditional methods for generating and maintaining early successional habitat—primarily logging, wildland fires, firewood cutting, and farm abandonment—are no longer available (Trani and others 2001). Early successional habitat is now created mostly through active management, including timber harvest, fire, and the creation of grass and forb wildlife openings (Harper 2007, Litvaitis 2001).

Working in the northern hardwood forests characteristic of the Northeastern United States, DeGraaf and Yamasaki (2003) offer several suggestions for sustaining early successional habitat in managed forests that might apply to Southern Appalachian forests, while maintaining a balance between early and late successional habitat:

- Use regeneration cuts >2 acres to create favorable habitat for the reproduction of shade-intolerant and intermediate-tolerant tree species. Harvest areas > 2 acres produce dense regenerating stands with an abundance of fleshy fruits important for many wildlife species. Clustering smaller cuts can also maintain adequate amounts of early successional habitat while not fragmenting larger patches of older forests, and may encourage wider use by a range of wildlife.

- Shorten the time between periodic regeneration cuts in a management area to every 10 years. This action would produce permanent stands of forest ≤10 years old, ensuring their presence in the landscape.

- Because they are maintained in an early successional condition, powerline rights-of-way can help meet the needs of wildlife associated with early successional habitat (King and Byers 2002). Bulluck and Buehler (2006), however, caution that powerlines are structurally and functionally different from regenerating forest, and, while creating habitat for many bird species such as Kentucky warblers, powerlines would not support bird communities similar to those found in other forms of early successional habitat. Powerlines may increase the number of bird species, but may also increase the number of brood parasites, creating ecological traps or sinks (Lanham and Whitehead 2011). Other than their effects on birds and butterflies, however, the potential values of powerline rights-of-way as early successional habitats have been little studied (Lanham and Whitehead 2011).

Thompson and DeGraaf (2001) remind us, however, that even though some early successional wildlife species seem to be in decline, the issue remains controversial. Managing for early successional habitat may reduce habitat for at least some late successional species, forcing managers to make decisions between competing resources. In addition, benchmarks and threshold values for early successional habitat are not always clear or widely accepted. Finally, public land management agencies provide only a small proportion of the available young forests in the Eastern United States (Trani and others 2001).

— **Nancy Herbert**, Southern Research Station, Asheville, NC

## 2.9 Are natural disturbances such as canopy gaps sufficient to maintain early successional habitat?

Measured by aerial extent, the rate of gap disturbance in mature forests in the Eastern United States is approximately 1 percent per year, producing a landscape in which 6–9 percent of the forest communities are in active canopy gaps (Runkle 1982; see also Rankin and Tramer 2002a). Gap disturbance, however, may be qualitatively different from disturbance produced by timber harvest, and is certainly different from early successional habitat created through fire and wildlife openings. Gap formation rates in contemporary forests may be less than rates in primary forests, because managed forests tend to be younger, and therefore less prone to gap formation, than fully mature forests (Hunter and others 2001). In addition, canopy gaps are often too small to reach the thresholds of early successional habitat needed by many disturbance-dependent wildlife species (DeGraaf and Yamasaki 2003), although both the effective gap size and the likelihood of producing early successional habitat may be increased through multiple disturbances (Frelich and Reich 1999, Rankin and Tramer 2002a). Because of these differences, several authors argue that natural processes such as gap disturbance can no longer provide sufficient early successional habitats (Askins 2000, DeGraaf and Yamasaki 2003, Hunter and others 2001).As discussed in Question 2.6, however, gap disturbance is only one part of the natural disturbance regime. Other disturbances, such as wildland fire and windstorms, also produce forms of early successional habitat. Compared to gap disturbances, however, large-scale disturbances are both less frequent and less evenly spread across the landscape. Clearly, large-scale disturbances can produce substantial amounts of early successional habitat, but these effects tend to be localized, temporary and unpredictable, and do not address habitat needs across the full range of the Southern Appalachian landscape (DeGraaf and Yamasaki 2003).

— **Duke Rankin**, USDA Forest Service, Southern Region, Atlanta, GA

## 2.10 Are openings created by fire sufficient to produce early successional habitat?

Yes, fire-created openings are sufficient to produce early successional habitat if the fire is sufficiently intense to cause stand replacement (Smith 2000). Fire disturbance, however, may be qualitatively different from early successional habitat produced by timber harvest or natural disturbances. Stand-replacing fire, for example, may leave large numbers of snags, increasing populations of wood-boring insects, woodpeckers, insect- and seed-eating birds, and secondary-cavity nesters. Stand-replacing fire is also more likely to produce early successional habitat in large, contiguous blocks.

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## 2.11 What are the consequences if we do not manage for early successional habitat?

The hardwood forests in the Eastern United States are largely a legacy of stand-replacing, anthropogenic disturbance between 1700 and the early 1900s. This widespread disturbance produced historically high levels of early successional habitat (Lorimer 2001). As current forests mature, the amount of early successional habitat on public lands will most likely decline from these historical highs. This decrease, in turn, will probably reduce the population sizes, and frequencies, of species that depend on this habitat. As a result, species most closely associated with early successional habitat, especially golden-winged warbler, yellow-breasted chat, ruffed grouse, and American woodcock, will likely decrease (Dessecker and McAuley 2001, Thompson and DeGraaf 2001).

The effects of decreased early successional habitat on mammals are not as clear, and few significant effects have been documented (e.g., Heske 1995, Menzel and others 1999). Litvaitis (2001) argues that some game animals, such as turkey, white-tailed deer, ruffed grouse, black bear, and bobcat, would be negatively affected by a significant reduction in the amount of early successional habitat.

On the other hand, older forests will become more prone to natural disturbances, especially canopy disturbances associated with the fall of large, mature trees. When gaps involve multiple trees, or occur in conjunction with weather events, the canopy openings may be large enough to support early successional habitat (Rankin and Tramer 2002a). Under this scenario, mature, functional forests will eventually produce a shifting equilibrium of early successional habitat, proportional to the gap formation rate characteristic of the ecosystem.

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# 3. OAK REGENERATION IN THE SOUTHERN APPALACHIANS

Upland, mixed-oak forests occupy >50 percent of the forested land in the Central Hardwood Region of the United States (Johnson and others 2002), and oak trees play a pivotal role in the forest ecology and economy of the region. Oaks have been called a keystone species in Southern Appalachian forests (Fralish 2004, Petich and others 2002), influencing a wide range of species. For example, oaks have been associated with the distribution, abundance, and behavior of many forms of wildlife, ranging from migratory birds to black bear (Clark 2004, McShea and Healy 2002, Rodewald 2003). Acorns influence small mammal populations that form an important prey base for raptors and carnivores (McShea 2000). And both the physiognomy and decomposition rates of oak leaves, as well as the texture of oak bark, can increase arthropod abundance and availability for many bird species (e.g., Rodewald 2003), contributing to higher bird diversity in oak-dominated forests than in maple-dominated forests.

In addition to their ecological importance, oaks are among the most economically valuable hardwood species (Guyette and others 2004). Oaks provide raw materials for a wide range of wood products including veneer, lumber, pallets, pulp, and paper (Patterson 2004). The wood products sector accounts for an estimated 1.9 percent of all jobs and 2.3 percent of gross regional production across the mainly rural South (Abt and others 2002).

Despite the ecological and economic importance of oak-dominated forests, their sustainability is now threatened by a combination of oak decline, which causes progressive oak crown dieback and mortality (Oak and others 2004), and widespread oak regeneration failure. This failure is generally attributed to changes in disturbance regimes, especially fire, which have produced corresponding shifts in species composition, increasing the competitive pressure on oaks (Aldrich and others 2005). Maintaining oak forests in the face of these threats will require active management throughout the Southern Appalachians.

## 3.1 What are the current management practices for regenerating oak forests?

For upland hardwood stands in the Southern Appalachians, forest management is typically focused on restoring forest structure, function, and species composition, with a particular emphasis on regenerating oak species. On the other hand, upland hardwood stands are often compositionally complex, dominated by a mixture of oak and hickory species in association with hardwood species such as maple, yellow-poplar, magnolia, ash, and black cherry. Many of these species tend to be mid-tolerant of shade, and exhibit a reproductive strategy based on a combination of advance regeneration and overstory disturbance. For mid-tolerant hardwoods in general, and for oaks in particular, successful regeneration is a multistep process that includes seed production, seedling establishment, seedling development into stems that can compete following overstory release, and the release itself (Johnson and others 2002). In addition, some moderate level of overstory or midstory disturbance that increases light levels to the lower levels of the stand is required for seedlings to develop into the larger stems necessary to form an effective advance reproduction (Loftis 1990b).

The success of oak regeneration is therefore dependent upon the size and abundance of advanced oak reproduction prior to any silvicultural manipulation (Loftis 1990a, Sander 1971, 1972). In mesic oak stands in the Southern Appalachians, however, adequate advance reproduction of oaks may be uncommon because, although oaks remain in the overstory, the midstory is often dominated by more shade-tolerant species that suppress oak seedlings. As a result, the advance reproduction is composed of small-stemmed plants, and these plants are unable to compete effectively following overstory disturbance. Under these conditions, regeneration treatments that substantially reduce the forest canopy do not regenerate oaks, because shade-intolerant competitors, such as yellow-poplar, quickly overtop the small-stemmed advanced oak regeneration (Loftis 1990b, Shure and others 2006).

When large-stemmed advance reproduction is absent, the oak shelterwood method is an effective treatment to ensure successful oak regeneration (Loftis 1990b). This system can be described as a two-step shelterwood. The first cut, also called an establishment cut is a noncommercial removal of approximately 20-30 percent of the stand basal area from below, using herbicides to reduce re-sprouting. This effectively removes the shade-tolerant midstory, increasing light levels at the forest floor and encouraging the development of large-stemmed advanced oak reproduction without creating a light environment conducive to the establishment and growth of yellow-poplar. Once the oak seedlings have grown into large-stemmed advanced (>4 feet in height) reproduction, the overstory can be removed—a period of approximately 10 years. Although initially developed to improve oak regeneration on moderate to highly productive stands in the Southern Appalachians, the oak shelterwood method may also be used to regenerate any species of upland hardwoods that relies primarily on large-stemmed advance reproduction (Loftis 1983a).

On xeric oak sites, several factors, including lower soil moisture, fewer competing tree species, lower stand density, and a greater proportion of light reaching the forest floor, generally favor oaks (Johnson and others 2002). Because oaks are more competitive under these conditions, xeric oak stands are more likely to have abundant large-stemmed, advanced oak reproduction, and many regeneration methods can be used successfully on xeric oak stands (Johnson and others 2002).

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## 3.2 In the absence of management, can oaks trees regenerate naturally?

Historically, disturbance events such as low-intensity surface fires, timber harvesting, logging-associated wildland fires, grazing, land-clearing for agriculture, and the loss of American chestnut promoted forest conditions conducive to the establishment, development, and recruitment of oaks in upland hardwood forests (Abrams 1992, Lorimer 1993). Because these conditions were widespread during the past two to three centuries, oaks gained widespread dominance in many forest stands throughout the Southern Appalachians.

Although low levels of oak regeneration has been observed throughout the Central Hardwood Region, local conditions can substantially influence oak regeneration. On drier, less productive sites, such as ridge tops, oaks may regenerate naturally because the tree density and leaf area in the stand are low enough to allow sufficient light for oak seedlings to develop into a competitive regeneration source. In addition, fast-growing competitors, such as yellow-poplar, are typically not present in drier stands (Clinton and others 1994, Johnson and others 2002). In mesic, more productive sites, such as rich coves, oak regeneration is more problematic, primarily due to intense competition from fast-growing mesophytic species (Johnson and others 2002, Kellison 1993, Loftis and McGee 1993).

In these mesic sites, successful oak regeneration is a multistep process that includes (1) acorn production; (2) germination and seedling establishment; (3) seedling development into tall (>4 feet), large-stemmed seedlings, creating a pool of advance reproduction that can successfully compete with faster-growing species; and (4) timely and sufficient release of the advance reproduction through some form of overstory removal (Johnson and others 2002). For oaks to reliably regenerate, all four steps must occur in sequence and within a certain timeframe. On mesic, productive sites in the Southern Appalachians, poor oak regeneration generally results from the failure of the third step—the development of seedlings into a competitive advance reproduction. This failure occurs because the low-light conditions characteristic of mature forest stands decrease both the survival and the growth rate of oak seedlings (Loftis 1983a). As a result, mature forest stands may contain ample oak saplings, but the seedlings are short (<1 foot), small-stemmed, and unable to compete with fast-growing species following overstory removal (Loftis 1983a, 1983b).

Because oaks are intermediate in shade tolerance, midstory disturbance that increases light levels at the forest floor allows oak seedlings to develop into large saplings that form an effective advance reproduction (Loftis 1990b). When released through overstory removal, this advance reproduction is able to effectively compete with fast-growing competitors, promoting the recruitment of oaks into the overstory of the new, developing stand.

In the absence of a midstory disturbance, forest stands may develop in one of two ways. If left undisturbed, the low-light levels characteristic of mature forests promote the growth, and eventual recruitment, of shade-tolerant tree species, such as red and sugar maple (e.g., Spetich and Parker 1998). If, on the other hand, the overstory is removed through harvest or natural disturbance, shade-intolerant tree species, such as the fast-growing yellow-poplar, quickly occupy the site, and eventually dominate the stand (Shure and others 2006). As a result, treatments that remove the midstory, which promote the development of oak seedlings into a competitive advance reproduction while preventing the establishment and development of competing species, are needed to ensure timely and successful oak regeneration (Loftis 1990b).

Because the four steps associated with oak regeneration can occur naturally, successful oak regeneration without management can, and does, occur at various positions on the landscape, but remains unreliable and stochastic at any specific location. Although naturally occurring canopy gaps may facilitate oak regeneration (e.g., Clinton and others 1994), the natural disturbances characteristic of Southern Appalachian forests are generally insufficient to regenerate oak stands on a consistent and reliable basis, at least on mesic, productive sites (Della-Bianca and Beck 1985, Loftis 2004).

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— **David Loftis**, Southern Research Station, Asheville, NC

## 3.3 Are there treatments that could improve forest health in oak communities?

Ultimately, the term "forest health" comes from our human perspective. Although we may equate community health with productivity and resilience, especially among desired species such as oak trees, communities change over time through natural processes that are neither good nor bad.

Within this context, the term "forest health" is typically used when considering community changes brought about by nonnatural processes, such as nonnative pathogens, or changes in the natural disturbance patterns. These nonnatural processes tend to diminish the traits we value in our native forest communities—for example, forest products, native biodiversity, and wildlife values. Managing for forest health seeks to reduce the effects of these nonnatural processes and to encourage natural processes that will enhance the traits we find desirable.

In an oak-hickory forest community, managing for forest health typically means managing for oak production and regeneration. Managing for oaks may involve regeneration harvest, which removes aging trees that become increasingly susceptible to oak decline and mortality, and creates the open conditions necessary for vigorous regrowth of the stand. Other procedures that can improve forest health in these stands include improving the timber and wildlife attributes through direct management (timber stand improvement and wildlife stand improvement, guarding against the introduction of exotic pests and disease, returning natural disturbance regimes, and reducing human impacts, especially to soils and water. Fire may play an important role in maintaining healthy oak forests, by reducing and regulating insect pests (Ahlgren 1974, Komarek 1970, Miller 1979).

Early detection of forest health concerns is also an important step in reducing impacts to oak trees. One potential indicator of forest stress is the ratio of standing dead to live trees, sometimes called the forest health quotient. The quotient averages 0.08 in healthy Midwestern second-growth forests (Spetich and others 1999), and 0.089 in Arkansas forests (Spetich and Guldin 1999). Forests with quotients above these values may indicate forest stress. Although this quotient has not been assessed for the Southern Appalachians, comparing quotients derived from U.S. Forest Service, Forest Inventory and Analysis (FIA) data could help identify emerging forest health issues.

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— **Marty Spetich**, Southern Research Station, Hot Springs, AR

## 3.4 What is the role of fire in regenerating oak trees?

Prescribed fire is increasingly being used by land managers to restore oak communities and promote oak regeneration in the Southern Appalachians (Brose and others 2006, Dey and Fan 2009, Van Lear and Watt 1993). Fire is typically prescribed to reduce competition between fire-tolerant oak species and their fire-intolerant competitors, including red maple and yellow-poplar, especially in the regeneration layer (Waldrop and others 2008), and to increase light levels in the understory, producing larger and more competitive oak saplings (Barnes and Van Lear 1998).

Fire behavior and intensity, and therefore fire effects, vary across the landscape (Albrecht and McCarthy 2006). In general, fires burn at lower intensities on lower, more mesic slopes than on higher, more xeric slopes. This spatial variability in fire behavior and intensity leads to differential mortality among tree species, creating different types of forests on different sites (Elliott and others 1999). For example, prescribed fire can increase the density of competing, non-oak species on mesic sites, but it can increase the density of small-diameter oak saplings on xeric sites (Albrecht and McCarthy 2006).

The differential effect of prescribed fire across the landscape, especially across moisture and productivity gradients, emphasizes the need for site-specific information on the relationship between site quality and oak regeneration. For example, in the Cumberland Plateau region of Kentucky, both single and repeated fires have been effective at reducing overstory stem density and basal area, leading to increased light in the stand, but these benefits for oak regeneration are quickly offset by increased sprouting of competing species such as sassafras and red maple (Alexander and others 2008). Although fire can reduce the number of small (<1 foot tall) red maples, sprouts from larger, topkilled red maple stems often grow more quickly than oak sprouts (Alexander and others 2008), which can accelerate the loss of oaks from the community (Wendel and Smith 1986).

One treatment that has proven effective in regenerating oak forests in the Piedmont region of Virginia is the shelterwood/burn technique proposed by Brose and others (1999a, 1999b). In the first step, the stand is heavily cut to release all regeneration sources, regardless of species. This initial cut—technically, the establishment cut of a shelterwood harvest—promotes root growth in oak saplings and stimulates yellow-poplar seeds in the litter and duff layers to germinate (Brose and others 1999b).

After the establishment cut, stump sprouts from harvested trees advance reproduction present in the stand, and new seedlings all grow and develop under the sparse shelterwood canopy. During this period, oaks allocate a substantial proportion of their carbohydrates to root growth instead of shoot growth (Kolb and others 1990).

Three to 5 years after the establishment cut, the developing stand is burned to control species composition. Oak saplings, with large carbohydrate reserves in the root system and dormant buds located below the soil surface (Burns and Honkala 1990), will often re-sprout vigorously after being topkilled by the fire, while competing species, such as yellow-poplar and red maple, are often incapable of sprouting after being topkilled because they have smaller carbohydrate reserves in their roots (Kolb and others 1990), and hold their dormant buds above the soil surface (Beck 1981). The differential response to topkill increases the number of oaks relative to the competing tree species.

Interestingly, traditional shelterwood methods (e.g., Loftis 1983b, Schuler and Miller 1995) or single applications of prescribed fire (Johnson 1974, Wendel and Smith 1986), when used separately, rarely enhance the competitive position of oak regeneration on high-quality sites in the Appalachians. Unless present as large advance reproduction, oaks are unable to outcompete yellow-poplar seedlings that germinate and develop after shelterwood harvest (Loftis 1983b, 1999b), and prescribed fire alone appears ineffective at creating the light conditions necessary for the development of large advance reproduction (Alexander and others 2008). In addition, prescribed fire appears ineffective at releasing and recruiting advance oak reproduction into larger size classes (Alexander and others 2008), unless the fire intensity is sufficient to cause mortality in the forest canopy (Hutchinson and others 2005b, Signell and others 2005).

The combination of shelterwood regeneration followed by prescribed fire shows promise in regenerating oaks. However, even this method has been proven effective only on sites of moderate productivity in the Piedmont region of Virginia (site index between 70 and 80 feet; base age 50 for white oak; Brose and others 1999a). The shelterwood/burn technique has not been tested on high productivity sites, such as the cove forests of the Southern Appalachians, where competition from yellow-poplar is intense and oak regeneration is especially problematic. Researchers with the Southern Research Station are currently testing the shelterwood/burn technique across a range of sites in the Southern Appalachians of North Carolina and the Cumberland Plateau of Tennessee.

The recent and widespread-use of prescribed fire to regenerate and restore upland oak forests remains relatively untested across the various physiographic subregions of the Southern Appalachians. Most studies suggest prescribed fire, on its own, does not substantially improve oak regeneration (Alexander and others 2008, Hutchinson and others 2005b, Signell and others 2005, Wendel and Smith 1986). Because the effects of fire are spatially and temporally heterogeneous, more research is needed to determine the effects of prescribed fire, either alone or in combination with silvicultural treatments, to improve and promote oak regeneration in the upland forests of the Southern Appalachians.

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— **David Loftis**, Southern Research Station, Asheville, NC

## 3.5 What are the effects of cutting oak trees on high elevation sites?

High elevation sites in the Southern Appalachians are generally associated with elevations greater than 4,000 feet. At these elevations, forests tend to be cooler and moister than lower elevation forests, exhibit shorter growing seasons, and develop into forest communities more characteristic of northern regions. High elevation oak forests are separated from other oak forests by increased dominance of mesic oaks, especially northern red oak, and reduced dominance by white oak and hickories. They are also subject to increased levels of air pollution, especially acid rain, and are often considered particularly sensitive to environmental changes.

High elevation red oak forests rarely undergo Forest Service management actions. As a result, little research has been conducted on the effects of actions such as timber harvest and prescribed fire. In one study, Elliott and Knoepp (2005) examined the regeneration of high elevation oak forests. Situated at 4,500-5,500 feet, stands dominated by northern red oak were harvested using a variety of standard techniques, including two-age, shelterwood, and group selection. Six years later, northern red oak remained a common constituent of the community in the two-age and shelterwood units.

Management actions may also affect high elevation soils. Compared to low elevation forests, high elevation forests receive higher rates of atmospheric nitrogen deposition. As a result, high elevation soils contain seven times more total nitrogen than soils in low elevation, mixed-oak forests (Knoepp and others 2000, 2008). Because the soils contain higher levels of nitrogen, nitrogen cycles through the ecosystem faster than at lower elevations, (Knoepp and others 2000, 2008), increasing nitrogen availability in the soil and elevating nitrogen concentrations in streams draining high elevation watersheds (Knoepp and others 2008, Swank and Douglass 1975, Swank and Vose 1997). Disturbance increases both the rate of nitrogen cycling through the ecosystem and the rate at which nitrogen is exported in streams (Swank 1988). These responses are similar for both management actions and natural disturbances, and are proportional to the size of the disturbed area.

In summary, timber harvest in high elevation forests appears to have little effect on overstory composition. On the other hand, all disturbances at high elevation increase the rate at which nitrogen is exported from the ecosystem.

— **Jennifer Knoepp**, Coweeta Hydrologic Laboratory, Otto, NC

## 3.6 What are the consequences if we do not plan for oak regeneration?

Changes in disturbance regimes—both natural and anthropogenic—are promoting the gradual conversion of forests dominated by oaks into forests dominated by shade-tolerant species such as red maple (Nowacki and Abrams 2008, Orwing and Abrams 1994), or by shade-intolerant species such as yellow-poplar (e.g., Beck and Hooper 1986, Rodewald 2003). Over time, a decrease in the abundance of oaks, combined with associated changes in forest structure and composition, could have cascading effects throughout the upland hardwood ecosystems of the Southern Appalachians. For example, acorns are considered a critical forest resource because of their influence on populations of small mammals (McShea 2000). In turn, small mammals disperse spores and seeds (Wolff 1996) and represent an important prey base for raptors and carnivores.

In addition to the critical role of oaks for wildlife habitat and as a food resource, the loss of oak, in combination with concomitant increases in shade-tolerant species such as beech and maple, has been shown to dramatically reduce herbaceous plant diversity by reducing the amount of light at the forest floor and increasing the depth of the leaf litter (Fralish 2004). The loss of oak-dominated forests would also reduce ecosystem heterogeneity at the landscape level. This could, in turn, result in reduced resistance and resilience to future disturbances, including climate change, and the loss of ecosystem functioning (Yachi and Loreau 1999).

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Since beginning work on this report, **David Loftis** has retired from the Upland Hardwood Ecology and Management Research Work Unit, Southern Research Station, at the Bent Creek Experimental Forest in Asheville, NC. His research focused on the ecology and management of Southern Appalachian hardwood forests, especially predicting and controlling the composition of species following forest regeneration. He also worked on decision models for the management of hardwood forests, and the effects of silvicultural practices on biological diversity.

**W. Henry McNab** is a Research Forester, USDA Forest Service Southern Research Station, Bent Creek Experimental Forest, 1577 Brevard Road, Asheville, NC 28806. His research assignments have included silviculture of loblolly pine, smoke management from prescribed burning, forest biomass estimation, and silviculture of upland hardwoods. For the past 20 years, his research has involved ecological classification of plant communities in the Southern Appalachian Mountains, with an emphasis on quantifying the relationships among environmental gradients, species composition, and productivity of forest sites. His current research includes developing alternatives to site index for classification of forest site quality. He can be contacted at 828-667-5261 or hmcnab@fs.fed.us.

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**Thomas A. Waldrop** is Research Forester and Team Leader for Fire Science at the Center for Forest Disturbance Science, USDA Forest Service, Southern Research Station, 239 Lehotsky Hall, Clemson, SC 29634. His work emphasizes fire ecology in the Piedmont and Southern Appalachian Mountains, with research on fire techniques for stand replacement of Table Mountain pine, fire history, fuel loading characteristics of the Southern Appalachians, and impacts of stand replacement fire on various components of mountain ecosystems. In more recent years, he has been a leader in the National Fire and Fire Surrogate Study, which studies ecological impacts of fire and mechanical fuel reduction techniques in numerous ecosystems of the United States. He is currently developing the Consortium of Appalachian Fire Managers and Scientists to enhance science delivery for fire professionals in the Central and Southern Appalachian region. He can be contacted at 864-656-5054 or twaldrop@fs.fed.us.

# INDICES

## Subject Index



## Author Index

## Taxonomic Index

# APPENDIX
## Additional Fire Ecology Resources Available on the Internet

### Fire Web sites and Search Options

FRAMES—Fire Research and Management Exchange System
http://frames.nbii.gov/portal/server.pt

FRAMES—Southern Fire Portal
http://frames.nbii.gov/southernfire

Fire and Environmental Research Applications
www.fs.fed.us/pnw/fera/

International Association of Wildland Fire
www.iawfonline.org/

Joint Fire Science Program
http://jfsp.nifc.gov/

Association for Fire Ecology
www.fireecology.net/

Fire Learning Network
http://tncfire.org/training_usfln.htm
In particular, click on »South Central FLN (Arkansas-based), »Southeast FLN (Coastal Plain-based), »Southern Blue Ridge (Southern Appalachians) and »Appalachians (Central Appalachians) and »Network Publications

TreeSearch
www.treesearchfs.fed.us

Tall Timbers Research Station
http://ttrs.org/info/fedbintro.htm

Forestry Encyclopedia Network
http://forestencyclopedia.net
Includes the following encyclopedias: Southern Fire Science, Southern Appalachian Forest Ecosystems, Environmental Threats, and Southern Bioenergy

Fire Effects Information System
www.fs.fed.us/database/feis

### Fire Monitoring

FFI (Fire Effects Monitoring Analysis Software)
http://frames.nbii.gov/ffi

Fire Effects Monitoring Handbook
http://www.nps.gov/fire/download/fir_eco_FEMHandbook2003.pdf
Basis for the 5140 USFS Region 8 Fire Effects Monitoring program

FIREMON
www.fs.fed.us/rm/pubs/rmrs_gtr164.html

USFWS Fuel and Fire Effects Monitoring Guide
www.fws.gov/

Measuring and Monitoring Plant Populations
http://www.blm.gov/nstc/library/pdf/MeasAndMon.pdf

### Other Resources

Converting metric units to English
www.srs.fs.usda.gov/landowners/convert_metric_units.htm

Fire Management Today (magazine)
http://www.fs.fed.us/fire/fmt/index.html

InterfaceSouth—Southern Center for Wildland Urban-Interface Research and Information
www.interfacesouth.org/

Communicator's Guide to Wildland Fire
www.nifc.gov/preved/comm_guide/wildfire/index2.html

Fire Brochures for Public Distribution
http://www.interfacesouth.org/resources/brochure_all.html

Smoke Management Guide for Prescribed and Wildland Fire
www.nwcg.gov/pms/pubs/SMG/SMG-72.pdf

Forest Encyclopedia Network
www.forestencyclopedia.net/
The Network connects scientific results and conclusions with management needs and issues. Focus areas include Southern Appalachian ecosystems, Environmental Threats, Southern Fire Science, and Southern Bioenergy.

Summary of the Restoration Goals for the National Forests in North Carolina
www.cs.unca.edu/nfsnc/restoration/priorities.htm

### Fire-related Publications for Hardwood, Upland Forests

Rainbow Series

Wildland Fire in Ecosystems: Effects of Fire on Fauna
www.fs.fed.us/rm/pubs/rmrs_gtr042_1.pdf

Wildland Fire in Ecosystems: Effects of Fire on Flora
www.fs.fed.us/rm/pubs/rmrs_gtr042_2.pdf

Wildland Fire in Ecosystems: Effects of Fire on Soils and Water
www.fs.fed.us/rm/pubs/rmrs_gtr042_4.pdf

Wildland Fire in Ecosystems: Effects of Fire on Air
www.fs.fed.us/rm/pubs/rmrs_gtr042_5.pdf

Wildland Fire in Ecosystems: Fire and Nonnative Invasive Plants
www.fs.fed.us/rm/pubs/rmrs_gtr042_6.pdf

Henri D. Grissino-Mayer's Ultimate Tree-Ring Web Pages
web.utk.edu/%7Egrissino/my_page.htm

Fire and Fire Surrogates Study—Treatments for Ecological Restoration
www.fs.fed.us/ffs/index.html

Southern Forest Resource Assessment Chapter 25: Fire in Southern Forest Landscapes
www.srs.fs.usda.gov/sustain/report/fire/fire.htm

Using Fire to Control Invasive Plants: What's New, What Works in the Northeast
extension.unh.edu/Pubs/ForPubs/WPUFCI03.pdf

Fire Management Today "Fire Uses from the Past" Volume 64, No. 3
www.fs.fed.us/fire/fmt/fmt_pdfs/FMT64-3.pdf

Fire in the Southern Appalachians: Fuels, Stand Structure and Oaks
www.fs.fed.us/r8/boone/fire/fsa/index.shtml

A Joint Fire Science Program project in Daniel Boone National Forest

Prescribed Fire and Oak Ecosystem Maintenance: A Primer for Land Managers
www.siu.edu/~forestry/faculty/groninger/Prescribed%20Fire%20015.pdf

Proceedings from the 2000 Workshop on Fire, People, and the Central Hardwoods Landscape
www.treesearch.fs.fed.us/pubs/3762

Fire Learning Network Publications
tncfire.org/training_usfln_networkpubs.htm

Upland Oak Ecology Symposium: History, Current Conditions, and Sustainability
www.srs.fs.usda.gov/pubs/gtr/gtr_srs073/gtr_srs073.pdf

In particular, read the concluding paper, "Where Do We Go From Here?"
http://www.srs.fs.usda.gov/pubs/gtr/gtr_srs073/gtr_srs073-blaney001.pdf

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— **Beth Buchanan,** *Fire Ecologist*
U.S. Department of Agriculture Forest Service, Southern Region

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**Rankin, W.T.; Herbert, N.** 2014. Restoration in the Southern Appalachians: a dialogue among scientists, planners, and land managers. Gen. Tech. Rep. SRS-189. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 48 p.

We address three key questions for restoration ecology in the Southern Appalachian Mountains. First, what is the role of fire, especially when used as a management tool for oak-dominated ecosystems? Second, what is the relationship between early successional habitat and biodiversity? And third, how do we regenerate oak ecosystems? To answer these questions, first, we examine the historic role of fire in the mountains, discuss its effects on forest resources, and summarize a strategy for restoring fire to ecosystems with a long history of fire exclusion. Second, we examine the relationship between early successional habitats and wildlife resources in the mountains, discuss the pattern and rate of natural disturbance, and provide suggestions for creating and maintaining early successional habitat. And third, we review current management for oak regeneration and discuss the implications for oak ecosystems in the absence of management. In addition to addressing current questions in restoration ecology, we provide an extensive bibliography of the scientific literature, especially for fire management. Our goal is to provide a concise and practical summary of the current restoration literature for use by forest planners and managers throughout the Southern Appalachian Mountains.

**Keywords:** Early successional habitat, natural disturbance patterns, natural fire, oak regeneration, prescribed fire, restoration ecology, Southern Appalachian Mountains, wildlife.

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