Fuels Management in the Southern Appalachian Mountains, Hot Continental Division

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The Southern Appalachian Mountains, Hot Continental Mountains Division, M220 (McNab and others 2007) are a topographically and biologically complex area with over 10 million ha of forested land, where complex environmental gradients have resulted in a great diversity of forest types. Abundant moisture and a long, warm growing season support high levels of productivity across the area. Disturbances such as fire, severe windstorms, ice storms, and outbreaks of pathogens and insect infestations are common and can affect large areas. The interactions among these factors can produce a dynamic forest fuels situation, requiring frequent monitoring and updating of fuel loads. Fire exclusion since the early 20th century has allowed a buildup of fuels, both living and dead, across the Southern Appalachian Mountains. A rapidly expanding wildland-urban interface and the potential for climate change to increase the frequency and severity of wildfires will require that more resources be devoted to fuel management. In this new environment, managers will need more effective methods of fuel management to reduce the potential for hazardous wildfires and maintain landscape diversity.

Fire History

Fire played an integral role in determining historic patterns of forest vegetation across the Southern Appalachian Mountains (Delcourt and Delcourt 1997). Historical accounts suggest that recurrent burning by Native Americans was common in Southern Appalachian forests, starting 10,000 to 12,000 years ago and extending through the arrival of Europeans (DeVivo 1991, Fowler and Konopik 2007, Van Lear and Waldrop 1989). Fowler and Konopik (2007) outline five periods with unique fire regimes, all having different impacts on vegetation as fire regimes and forest structure have been influenced by changing cultures, fluctuations in population sizes, and altered land-use priorities.

During the first period, Native Americans burned entire valleys near settlements to clear land for agriculture and selectively burned upper slopes and ridges to promote wildlife habitat and mast production (Delcourt and Delcourt 1997, DeVivo 1991). Fire frequency during this time was likely negatively correlated to distance from settlements

(Delcourt and Delcourt 1997). Some estimates suggest that the fire frequency was 7 to 12 years on ridges and upper slopes at elevations below 1000 m, but less frequent at upper elevations (Frost 1995). Others suggest that frequency was annual or biannual in some areas (Barden 1997). This presettlement landscape was likely a “shifting mosaic of open grasslands, woodlands, and closed forests with widely scattered Indian villages” (Buckner 1989).

The second period of fire use began with the arrival of Europeans in the 16th century. The new arrivals introduced pandemic diseases, and native populations plummeted. The initial result for the landscape was a reduction of fire frequency, which altered forest structure. By the 17th century, the European population increased, and much of the landscape was occupied by settlers who began adopting many Native American burning practices.

A third period characterized as the Industrial Revolution began in the latter 19th century as railroads made previously isolated parts of the mountains easily reachable and enabled large-scale transportation of commodities. Subsequent large-scale timber exploitation resulted in heavy fuel loads from slash and led to fires that were much more intense, albeit not much more frequent than in previous periods (Harmon 1982).

The high intensity and often stand replacing fires ushered in the fourth period of fire exclusion beginning in the early 20th century. Complete fire exclusion was the policy of Federal and State land management agencies throughout the century and continues to the present.

The fifth period of fire management began in the late 20th century. Currently, prescribed fires are the dominant form of fire use in the Southern Appalachian Mountains. Suppression is still practiced on most wildfires. Natural ignition by lighting is infrequent (Barden and Woods 1973, Harmon 1982)

Fire exclusion caused important changes in the structure and function of Southern Appalachian forests (Vose 2000). Stem density has increased in the shrub layer and species composition has changed with a greater dominance of shrubs such as mountain laurel (*Kalmia latifolia*). High vegetation density has inhibited regeneration of overstory species and decreased diversity of herbaceous communities in the understory (Chastain and Townsend 2008). Fuel loads have also increased for reasons not directly related to fire exclusion, such as overstory mortality resulting from native and nonnative pathogens and insects. Accelerated mortality has increased the quantity of coarse woody debris and other organic matter, which have increased carbon and nutrient pools in the forest floor. This effect varies across the Southern Appalachian Mountains and among ecosystems, presenting a difficult situation for forest management and restoration where a “one size fits all” approach may not be suitable. If managers understand how the interactions of past land use and disturbances have given rise to current stand conditions, they can take appropriate actions to mitigate fuel risks.

The recognition of the role of fire in maintaining biodiversity and its usefulness as a forest management tool resulted in the active use of prescribed fire in Southern Appalachian Mountains beginning in the 1980s. Fires today are less frequent and generally much smaller than those of the past (Barden and Woods 1973, Lafon and others 2005). Despite the usefulness of prescribed fire, its application is often limited by air quality issues and operational complexities within a rapidly growing wildland-urban interface.

Of the approximately 15.2 million forested hectares encompassed by the Southern Appalachian Mountains, 84 percent (about 13 million ha) is privately owned (Southern Appalachian Man and the Biosphere Program 1996a) and 16 percent is in public management. Approximately 2.2 million ha are under Federal management, primarily in national forests or national parks. Federal lands represent the vast majority of Southern Appalachian area where fuels are being managed. They include 10 national forests, the Great Smoky Mountains National Park, and the Shenandoah National Park. An additional 230 000 ha are managed by State agencies, a little over 40 000 ha by the U.S. Department of Energy and U.S. Department of Defense, and about 20 000 ha by the Eastern Band of the Cherokee Indians.

Southern Appalachian Forests

Climate, Ecosystem Processes, and Disturbance Regimes

The Southern Appalachian Mountains stretch from northeastern West Virginia through western Virginia and North Carolina, to northwestern South Carolina and northern Georgia, to northeastern Alabama (Southern Appalachian Man and the Biosphere Program 1996a). Elevations generally range from 600 m in major river valleys to over 3000 m on upper ridges and peaks. Local climate varies dramatically along latitudinal and elevational gradients. Increases in elevation are associated with decreasing temperature and increasing precipitation, relative humidity, and cloud cover. Summers are usually hot, with daytime temperatures frequently exceeding 32 °C, and below freezing temperatures are common throughout the winter. Mean temperature is 19 °C in the south, decreasing to 8.3 °C in the north. Annual precipitation is abundant, ranging from a maximum in the south of approximately 200 cm to just over 75 cm in the north. Most precipitation is in the form of springtime rain, but winter snows and summer thunderstorms are frequent. Widespread drought occurs about once every decade. See chapter 3 for additional detail on the physical setting for this area.

The Southern Appalachian Mountains are well known for biological diversity and are home to a variety of forest ecosystems that are generally distributed along strong elevational and topographic gradients (Whittaker 1956). Other factors—such as precipitation and temperature—also vary along these gradients, affecting forest composition and ecosystem processes such as decomposition (Abbott and Crossley 1982), turnover of soil carbon (Garten and Hanson 2006), and aboveground forest productivity (Bolstad and others 2001). Evidence suggests that these ecosystem processes control fuel loading (Iverson and others 2003, Kolaks and others 2004, Waldrop and others 2004), but variation in rates of input across the landscape may be balanced by corresponding rates of decomposition (Kolaks and others 2003, Waldrop 1996). Evidence from over 1,000 study plots at low to middle elevation across the farthest southern extent of the Appalachian Mountains found little difference in surface fuels across topographic positions (Waldrop and others 2007). Instead, disturbance history and type were found to play a greater role in determining fuel loads.

In addition to fire, other disturbances occur at variable frequencies and severities, with impacts ranging from single-tree mortality to large areas of mortality resulting from high wind, hurricanes, floods, pathogen outbreaks, insect infestations, drought, and ice storms. A great deal of evidence suggests that these disturbances may also vary in intensity along environmental gradients (Elliott and Swank 1994, Harmon and others 1984, McNab and others 2004, Reilly and others 2006, Stueve and others 2007). The interactions between environmental gradients and disturbance hold implications for fuels management because they alter dead and down surface fuels and patterns of regenerating live fuels in recently disturbed areas. Waldrop and others (2007) found less litter on sites that had been burned in the last 10 years and higher 1-hour fuel loads on sites recently infested by southern pine beetles (*Dendroctonus frontalis*). In areas that had been subjected to beetle attack, fire, or wind—or all three—larger woody fuels were more abundant than on undisturbed sites.

Major Forest Ecosystems

The diverse vegetation in the Southern Appalachian Mountains has the potential to create a wide array of fuel management scenarios. We present an ecosystem-based approach using major vegetation and "macro" habitat groups (Southern Appalachian Man and the Biosphere 1996b). These forest ecosystems correspond well with those described by others (McLeod 1988, Newell and others 1999, Whittaker 1956) and provide managers with a useful classification scheme. Additionally, geographic information system data on the distribution and occurrence of these ecosystem types are readily and freely available from the Southern Appalachian Assessment Online Database (http://samab.org/data/SAA_data.html).

Bottomland hardwood forests

Bottomland hardwood forests are found at the lowest elevations in the major river valleys and cover approximately 183 00 ha in the Southern Appalachian Mountains. These forests are dominated by several species including sweetgum (*Liquidambar styraciflua*), yellow-poplar (*Liriodendron tulipifera*), red maple (*Acer rubrum*), river birch (*Betula nigra*), American sycamore (*Platanus occidentalis*), green ash (*Fraxinus pennsylvanica*), American elm (*Ulmus americana*), silver maple (*Acer saccharinum*), and eastern cottonwood (*Populus deltoides*). Bottomland hardwood forests are very productive with rapid decomposition rates resulting from seasonal flooding and high soil moisture. Floods play a role in the disturbance regime of bottomland hardwood forests and may redistribute coarse woody debris and remove litter, especially after large events.

Invasion of nonnative species has potentially altered fuel structure in bottomland hardwood forests. Dense thickets of Chinese privet (*Ligustrum sinense*) and multifloral rose (*Rosa multiflora*) may form large patches of continuous fuels capable of spreading fire under dry conditions. Large patches of kudzu (*Pueraria lobata*) reaching into canopies along forest edges may also occur. The presence of these species may warrant the use of fuels management to reduce localized fire hazards and control further spread of invasive species.

Oak forests

Oak forests (*Quercus* spp.) occur across a wide range of middle elevations and vary in topographic moisture. These are the most extensive ecosystems in the Southern Appalachian Mountains and cover approximately 5.4 million ha. Xeric oak forests are dominated by chestnut oak (*Q. prinus*) and scarlet oak (*Q. coccinea*) with an abundant ericaceous shrub layer. Post oak (*Q. stellata*), black oak (*Q. velutina*), southern red oak (*Q. falcata*), blackjack oak (*Q. marilandica*) and bear oak (*Q. ilicifolia*) may be found at lower elevations. Mesic oak forests are dominated by white oak (*Q. alba*) and northern red oak (*Q. rubra*). Pignut hickory (*Carya glabra*) may also be present. A thick layer of potentially flammable ericaceous shrubs composed mostly of mountain laurel with several species of blueberry (*Vaccinium* spp.) and huckleberry (*Gaylussacia* spp.) is often present throughout. Rhododendron (*Rhododendron maximum*) may be present in mesic oak forests. This shrub layer represents a major source of hazardous fuels, particularly when composed of mountain laurel, and can frequently pose a serious problem for fuel management.

Fire plays a major role in the disturbance regime of oak forests. It is hypothesized that many of these forests developed under a regime of frequent low intensity fires (Abrams 1992). Fires are thought to have encouraged oak regeneration and inhibited encroachment of more fire sensitive mesic species like red maple and blackgum (*Nyssa sylvatica*). Absence of fire in the last century has likely increased the abundance of mountain laurel and other ericaceous shrubs and created hazardous fuel conditions. Wind and logging are also part of the disturbance regime in oak forests. Both of these disturbances have the potential to increase larger woody fuels (Waldrop and others 2007).

Southern yellow pine forests

Southern yellow pine forests (*Pinus* spp.) are present on the xeric upper slopes and ridges of low and middle elevations and make up approximately 1.5 million ha in the Southern Appalachian Mountains. The major constituents are Virginia pine (*Pinus virginiana*), pitch pine (*Pinus rigida*), and Table Mountain pine (*Pinus pungens*), with their respective importance increasing with decreasing topographic moisture and increasing elevation. A dense shrub layer consisting primarily of ericaceous species including blueberry, huckleberry, and mountain laurel is frequently present. Also frequently present in the shrub layer are hardwood species such as oaks, blackgum, and red maple. Piedmont species such as shortleaf pine (*Pinus echinata*) and loblolly pine (*Pinus taeda*) may also occur but are limited to the lowest elevations. Longleaf pine (*Pinus palustris*) has a very limited montane distribution on dry ridges up to 600 m at the farthest southwestern part of the Appalachian Mountains.

Many yellow pine stands were established early in the 20th century before the period of fire exclusion (Brose and Waldrop 2006) and are now in a decadent state (Williams and others 1990). Active programs of prescribed burning are in place to promote regeneration of fire-adapted species, such as Table Mountain pine and pitch pine, by reducing the presence of encroaching shrubs and hardwood species and allowing sunlight to reach the forest floor. Past work has assumed that regeneration of these species required intense stand replacing fires, but more recent work suggests that 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 issues for fuel managers. Potentially flammable evergreen canopies and abundant vertical fuels like mountain laurel can result in high severity crown fires. In addition, disturbance such as wind, ice storms, and southern pine beetle infestations can increase the abundance of both small-diameter and large woody fuels (Waldrop and others 2007). Periodic surface fires would not only facilitate regeneration but they would also reduce dangerous fuel loads.

Mixed pine-hardwood forests

Mixed pine-hardwood forests are found on lower and middle elevation slopes and ridges across the Southern Appalachian Mountains, covering approximately 1.3 million ha. Dominant species include the major constituents of both oak and southern yellow pine forests at varying densities. Oak species may include chestnut, scarlet, white, and northern red oak. At lower elevations, pine species may include loblolly and shortleaf. Middle to upper elevation mixed pine-hardwood forests may include Virginia, pitch, and Table Mountain pines. Fire susceptible species, such as red maple, blackgum, eastern white pine (*Pinus strobus*), and eastern hemlock (*Tsuga canadensis*) may be present in areas where fire has been excluded. A shrub layer consisting of species of blueberry, huckleberry and mountain laurel is also often present.

Disturbance regimes and productivity in mixed pine-hardwood forests are similar to those of oak and southern yellow pine forests. The mixture of species in these forests could be explained by their mid-successional status. In the absence of fire to promote pine regeneration, the most likely eventual fate of southern yellow pine forests is to succeed to oak forests. This process may be accelerated by other disturbances, particularly southern pine beetle attacks, in stands with older pines. These areas may be characterized by large amounts of both small-diameter and large woody fuels on the ground (Waldrop and others 2007). A frequent, low intensity fire regime may promote the coexistence of pine and oaks in these forests.

Mixed mesophytic hardwood forests

Mixed mesophytic hardwoods forests are among the most diverse forest communities in the Southern Appalachian Mountains, covering approximately 1.3 million ha. Dominant trees may often include yellow-poplar, white oak, northern red oak, basswood (*Tilia* spp.), yellow buckeye (*Aesculus octandra*), white ash (*Fraxinus americana*), eastern hemlock, American beech (*Fagus grandifolia*), and sugar maple (*Acer saccharum*). These forests are typically found on moist eastern and northern facing slopes and sheltered coves above 1200 m.

Fires in these forests were historically infrequent and remain that way today. Their topography and upper elevational range likely result in higher fuel moistures relative to other ecosystem types. However, periods of prolonged drought can result in overstory mortality, which may increase both surface fuels and midstory density in resulting canopy gaps (Olano and Palmer 2004).

White pine-hemlock-hardwood forests

White pine–hemlock-hardwood forests are typical of cool, moist ravines over a range of elevations. These forests cover approximately 606 000 ha of the Southern Appalachian Mountains. Species composition is dominated by white pine and eastern hemlock with occasional hardwoods such as yellow-poplar, blackgum, sweet birch (*B. lenta*), Fraser magnolia (*Magnolia fraseri*) and red maple. Rhododendron is common in the shrub layer. Forest structure is often composed of large diameter trees at low density with a thick layer of rhododendron in the midstory.

The historical disturbance regime of white pine-hemlock-hardwood forests was likely dominated by wind. Although generally long-lived, large white pine and eastern hemlock may be susceptible to windthrow, which promotes gap phase regeneration of the less shade-tolerant deciduous species. These forests were likely sheltered from most fires because they are located in high moisture ravines. However, when fire does occur in these forests, mortality can be high (Reilly and others 2006). The recent invasion of the hemlock woolly adelgid (*Adelges tsugae*) has resulted in large-scale mortality of eastern hemlock. High rates of tree mortality will likely cause a pulse in both small and large surface fuels as branches and snags fall.

Northern hardwood forests

Northern hardwood forests are distributed in coves and upper slopes at elevations ranging from 1200 to 1700 m, and cover approximately 249 000 ha of the Southern Appalachian Mountains. Dominant species include sugar maple, American beech, and yellow birch (*Betula alleghaniensis*). Other species such as pin cherry (*Prunus pensylvanica*) and species found in mixed mesophytic hardwood forests may also be present. Species frequently present in the shrub layer are striped maple (*Acer pensylvanicum*) and American mountain ash (*Sorbus americana*).

Disturbance in northern hardwood forests is primarily by wind; fire was likely infrequent historically. Because of the elevational distribution of these forests, fuel moisture is likely higher relative to other ecosystems in the Southern Appalachian Mountains. The response of northern hardwood forests to droughts is likely similar to that of mixed mesophytic forests, where canopy mortality may increase surface fuels and abundant recruitment results in increased sapling densities. These effects may potentially be more dramatic with increased exposure on upper slopes.

Spruce-fir forests

Spruce-fir forests (*Picea* spp.–*Abies* spp.) occur at the highest elevations, generally above 1500 m. These forests cover approximately 36 500 ha of the Southern Appalachian Mountains. Growing seasons are short; weather is characterized by abundant moisture, high relative humidity, and high cloud cover. Dominant species include red spruce (*Picea rubens*) and Frasier fir (*Abies fraseri*). Species common to northern hardwood forests such as yellow birch, sugar maple, and pin cherry may also be present. Woody species found in the shrub layer may include rhododendron, Catawba rosebay (*Rhododendron catawbiense*), mountain maple (*Acer spicatum*), and American mountain ash.

The disturbance regime of spruce-fir forests includes wind and ice storms. Although fires are infrequent, these forests are structurally similar to boreal forests and large high severity fires have occurred during prolonged drought. In October of 1925, one North Carolina fire in Haywood County burned approximately 10 000 ha in 3 days in what is now the Shining Rock Wilderness Area (Barden 1978). Local accounts describe a stand replacing fire near Mt. Mitchell during the early 1900s. More recently, acid precipitation and attacks of the balsam woolly adelgid (*Adelges piceae*) have resulted in large-scale mortality of canopy trees. Areas recently disturbed by ice or the balsam wooly adelgid (Smith and Nicholas 2000) may have abundant coniferous regeneration capable of spreading intense fire.

Fuel Management in the Southern Appalachian Mountains

Current fuels management in the Southern Appalachian Mountains is performed primarily by public land managers on oak, southern yellow pine, and mixed pine-hardwood forests. The most common technique employed by land managers is prescribed fire.

Goals of fuel management in the Southern Appalachian Mountains vary; but in addition to reducing the risk of wildfire, they also include promoting biodiversity, restoring native ecosystems, and improving wildlife habitat. Decreasing wildfire risk involves reducing surface fuels, and increasing the gap between surface fuels and living crowns (Agee and Skinner 2005). Promotion of biodiversity and restoration of native ecosystems often focuses on regenerating fire-adapted species like Table Mountain and pitch pines. Fuel treatments for restoring native ecosystems also include reducing the density of mountain laurel, rhododendron, and fire-susceptible tree species like red maple (Nowacki and Abrams 2008)—species that may substantially reduce regeneration of oak and other desirable species. Fuel treatments such as prescribed fire and thinning which increase surface light levels may also be used to improve wildlife habitat by increasing the growth of new vegetation and by promoting flowering (Whitehead 2003), which increases visitation of pollinators (Campbell and others 2007) and fruit production (Blake and Hoppes 1986, Greenberg and others 2007). Most studies on fuel treatments have primarily concentrated on prescribed fire and its effects on forest structure and live fuels, with little emphasis on the forest floor and dead and downed fuels. However, results from the National Fire and Fire Surrogate Study explicitly address effects of fuel treatments on the forest floor as well as dead and downed woody fuels (Waldrop and others 2008).

Prescribed Fire

Prescribed fire is by far the most frequently used fuel management technique in the Southern Appalachian Mountains. Prescribed fire has a relatively short history in the area because of fear that hardwoods and soils may be damaged and the potential difficulty in controlling fire on slopes (Van Lear and Waldrop 1989). In the early 1980s, managers first used prescribed fire for site preparation after clearcutting hardwood stands (Phillips and Abercrombie 1987). The use of prescribed fire for restoration of native communities began in the 1990s (Waldrop and Brose 1999).

The effects of prescribed fire as a fuel management technique have the potential to vary a great deal, depending largely on burning conditions and the ultimate goals of managers—both of which will inevitably vary largely across ecosystems and will alter fire intensity and severity. We summarize the effects of prescribed fire as a fuel management tool from published reports in oak, southern yellow pine, mixed pine-hardwood, mixed-mesophytic, and white pine-hemlock-hardwood forest ecosystems. Caution is advised when considering the results summarized below because they are derived from a limited number of observations and likely do not capture the full range of effects under a wide variety of site and burning conditions.

In oak ecosystems

Prescribed fires in oak ecosystems are generally low to moderate severity surface fires (Elliott and others 1999, Vose and others 1999, Waldrop and others 2008), which can be attributed to the characteristics of surface fuels in broadleaf forests and the resilience of most oak species to fire damage. However, areas of higher intensity fire can occur where there is a thick layer of ericaceous shrubs. Phillips and others 2006 reported that fire intensity in an oak forest ranged from 9.9 to 53.6 kW/m with flame lengths ranging from 0.3 to 0.5 m, and the rate of spread ranged from 0.3 to 1.4 m/minute; at 1 to 2 m above the ground, temperatures ranged from less than 52 to 160 °C. Vose and others 1999 reported that mean soil temperature reached 59 °C from 16.8 mm downward to 52 mm.

Table 1 shows that prescribed fires in oak ecosystems generally have only minor effects on forest structure (Elliott and others 1999, Waldrop and others 2007). Although the effect on stand basal area is small, density of saplings initially decreases after treatment. However, vigorous sprouting of hardwoods and ericaceous shrubs can result in sapling density that reaches or exceeds pretreatment levels 2 to 3 years after application of prescribed fire (Waldrop and others 2007). The effects on surface fuels are mostly limited to consumption of about half the mass of small wood and litter, and the effects on the humus layer and coarse woody debris are minor (Vose and others 1999). The high productivity of most sites means that surface fuels rapidly attain pretreatment loadings.

In southern yellow pine ecosystems

Prescribed fires in southern yellow pine ecosystems, particularly those dominated by Table Mountain and pitch pine, have the potential for high severity and are therefore likely to confront managers with some of their greatest challenges. Mountain laurel can act as a vertical fuel where it is abundant, allowing flames to reach into pine canopies. Flame temperatures have reached >800 °C, with a 59 °C heat pulse penetrating 24 mm into the forest floor (Vose and others 1999). Flame lengths can vary a considerably, ranging from as low as 1 to 3 m to as high as 12 to 46 m (Welch and others 2000). Ignition of crowns on upper slopes and ridges is also possible (Elliott and others 1999).

Prescribed fires in southern yellow pine ecosystems can have major effects on forest structure (table 2). Studies have reported reductions of 20 to 35 percent in basal area,

Table 1. Effects of prescribed fires on live and surface fuels in two Southern Appalachian oak ecosystems, (1) in March 2003 and 2006 on the Green River Game Lands in North Carolina and (2) in April 1995 on the Nantahala National Forest in North Carolina.

Site (elevation)	Measurements taken	Basal area: All sizes	Basal area: ≥5 cm	Density: <10 cm	Density: ≥5 cm	Density: 1 to 4.9 cm	Litter	Humus	Woody debris: <7.5 cm	Woody debris: ≥7.5 cm
		m²/ha		stems/ha			kg/ha			
Green River Game Land	Pretreatment	26.5	–	1,500[a]	–	–	–	–	–	–
	1 year after treatment	26.3	–	700[a]	–	–	–	–	–	–
	3 years after treatment	26.1	–	1,500[a]	–	–	–	–	–	–
	5 years after treatment (1 year after second treatment)	25.9	–	800[a]	–	–	–	–	–	–
Nantahala National Forest (1500 to 1700 m)	Pretreatment	–	28.7	–	1,448	8,518	3775	14 780	4234	8096
	1 year after treatment	–	28.4	–	1,365	1,556	2825	13 849	2465	7308

– = No data available.
[a] Exact values were not reported but were estimated for this summary based on figures.
Sources: Waldrop and others (2008), Elliott and others (1999), and Vose and others (1999).

Table 2. Effects of prescribed fires on live and surface fuels in three Southern Appalachian yellow pine ecosystems, (1) in October 1995 and (2) in May 1996 at the George Washington and Jefferson National Forest in Virginia, (3) in May 1996 at the Pisgah National Forest in North Carolina, and (4) in April 1995 in the Nantahala National forest in North Carolina.

		Basal area		Density						Woody debris	
Site (elevation)	Measurements taken	≥2.5 cm	>5 cm	<2.5 cm	≥2.5 cm	1 to 4.9 cm	≥5 cm	Litter	Humus	≥7.5 cm	<7.5 cm
		----------*m²/ha*--------		--------------------*stems/ha*-------------------				--------------------*kg/ha*----------------------			
George Washington and Jefferson	Pretreatment	23.4	–	1113	1525	–	–	–	–	–	–
	1 year after treatment	16.6	–	2912	625	–	–	–	–	–	–
	Pretreatment	28.4	–	1788	1594	–	–	–	–	–	–
	4 months after treatment	15.9	–	3250	295	–	–	–	–	–	–
Pisgah	Pretreatment	32.3	–	1712	1850	–	–	–	–	–	–
	4 months after treatment	25.9	–	2295	888	–	–	–	–	–	–
Nantahala National Forest (1500 to 1700 m)	Pretreatment	–	26.8	–	–	12 178	1545	5362	30 609	8776	6933
	1 year after treatment	–	18.7	–	–	409	913	1873	28 449	7726	1369
	2 years after treatment	–	–	–	–	5 692	–	–	–	–	–

– = No data available.
Sources: Welch and others (2000), Elliott and others (1999), and Vose and others (1999).

and 40 to 75 percent in overstory-tree stem density (Elliott and others 1999, Vose and others 1999, Welch and others 2000). Despite a large initial reduction of density in the sapling layer, shrubs and hardwoods sprout even after these higher severity fires, potentially increasing densities in the years following fire (Welch and others 2000). Consumption of surface fuels is only 60 to 70 percent of the mass of small wood and litter, and the effects on the humus layer and coarse woody debris are minor (Vose and others 1999).

In mixed pine-hardwood ecosystems

Studies on prescribed fire in mixed pine-hardwood ecosystems have shown the potential for large variations in fire intensity and severity from site to site (Waldrop and Brose 1999), likely driven by mountain laurel density and with the relative proportion of more flammable pine crowns and less flammable deciduous crowns. Hubbard and others (2004) reported flame lengths from 0.3 to 1.52 m; estimates from temperature-sensitive paints on ceramic tiles showed a maximum of 135 °C at 30 cm above the ground and 59 °C at 1.0 cm below the forest floor. Waldrop and Brose (1999) reported high fire intensity with crowning occurring on upper ridges.

Prescribed fires in mixed pine-hardwood forest ecosystems can also have highly variable effects on forest structure and soils (table 3). Waldrop and Brose (1999) documented the effects of this variation on stand structure, regeneration, and composition of the forest floor. Sites burning at low intensity had an average reduction in basal area of approximately 20 percent among trees >5 cm d.b.h. (diameter at breast height) compared to 96 percent for sites burning at high intensity. Decreases in the density of trees 2.5 to 4.9 cm d.b.h. ranged from 40 percent in low-intensity plots to 99 percent in high-intensity plots. Although all stems smaller than 2.5 cm d.b.h. were killed, hardwood regeneration was abundant in all sites regardless of intensity. Pine regeneration varied among fires and was highest at medium-low intensity and lowest at medium-high

Table 3. Effects of prescribed fires on live and surface fuels in two Southern Appalachian mixed pine-hardwood ecosystems, (1) in March 2001 on the Chattahoochie National Forest in Georgia and Cherokee National Forest in Tennessee, and (2) in April 1997 on the Chattahoochie National Forest in Georgia.

			Basal area		Density						Woody debris	
Site (elevation)	Severity	Measurements taken	≥5 cm	≥3 m tall	>0.5 m tall	≥0.5 m tall, but <5 cm	2.5 to 4.9 cm	≥5 cm	Litter	Humus	<5 cm	≥5 cm
			------m²/ha------		-----------------stems/ha-----------------				-------------------kg/ha-------------------			
Chattahoochie/ Cherokee[a] (260 to 415 m)	Low	Pretreatment	31.1	–	68,480	9,100	–	1,485	6028	11 435	6906	7611
		After 1 year	28.8	–	138,120	5,900	–	1,362	1833	10 837	4425	6696
		After 2 years	23.9	–	113,740	9,525	–	1,150	–	–	–	–
Chattahoochie (885 to 1100 m)	Low	Pretreatment	–	28.3	–	–	95[b]	716[b]	–	–	–	–
		After 3 months	–	22.7	–	–	0[b]	430[b]	–	–	–	–
	Medium Low	Pretreatment	–	34.5	–	–	200[b]	847[b]	–	–	–	–
		After 3 months	–	11.1	–	–	0[b]	177[b]	–	–	–	–
	Medium High	Pretreatment	–	17.4	–	–	105[b]	775[b]	–	–	–	–
		After 3 months	–	1.6	–	–	0[b]	45[b]	–	–	–	–
	High	Pretreatment	–	27.0	–	–	110[b]	776[b]	–	–	–	–
		After 3 months	–	1.0	–	–	0[b]	6[b]	–	–	–	–

– = No data available.
[a] Between the first and second year after burning, several sites were impacted by southern pine beetles, so changes are not wholly attributable to fire.
[b] Exact values were not reported but were estimated for this summary based on tables.
Sources: Elliott and Vose (2005), Hubbard and others (2004), and Waldrop and Brose (1999).

and high intensity. Regardless of intensity, consumption on the forest floor was limited to litter, with little consumption of humus or exposure of mineral soil. Other studies in mixed pine-hardwood ecosystems have found similar results of low-intensity prescribed fire on forest structure and the forest floor (Elliott and Vose 2005, Hubbard and others 2004).

In mixed mesophytic hardwood ecosystems

Mixed mesophytic hardwood ecosystems commonly occupy sheltered sites with high moisture, and thus tend to burn at lower intensity during prescribed fires. Although mountain laurel may be present, rhododendron and mesic-hardwood saplings are generally the most abundant live fuels. Because the fire risk is lower compared to other Southern Appalachian ecosystems, fuel treatments in mixed mesophytic hardwood ecosystems may be of low priority to forest managers. As a result, studies and observations on prescribed fire in this type are limited. Available observations report that intensity is substantially lower than in other ecosystems—temperatures at 1 to 2 m above the ground were consistently below 52 °C; and on average, temperatures of 49 °C penetrated an average of only 0.5 mm into the ground (Vose and others 1999).

Low intensity prescribed fires in mixed mesophytic hardwood ecosystems have little effect on live fuels in the overstory and midstory (table 4). Elliott and others (1999) found no overstory mortality; although stems in the midstory were killed, their presence was maintained after the fire by vigorous sprouting, and the effect on surface fuels was small (Vose and others 1999). The effect on the mass of coarse woody debris, small wood, and litter was small, but the mass of the humus layer increased.

Table 4. Effects of prescribed fires on live and surface fuels in a mixed mesophytic hardwood forest ecosystem in April 1995 in the Nantahala National Forest in North Carolina.

Site (elevation)	Measurements taken	Basal area	Density				Woody debris	
		≥5 cm	1 to 4.9 cm	≥5 cm	Litter	Humus	<7.5 cm	≥7.5 cm
		----m^2/ha----	-----------stems/ha-----------		------------------------------kg/ha------------------------------			
Nantahala (1500 to 1700 m)	Pretreatment	27.7	2,153	1,167	4 151	11 038	3 560	15 720
	1 year after treatment	27.8	2,652	1,117	4 028	13 410	3 231	15 596

Sources: Elliott and others (1999) and Vose and others (1999).

In white pine-hemlock-hardwood ecosystems

Although white pine-hemlock-hardwood forest ecosystems generally occur on moist sites, research suggests that fires of moderate intensity can occur, particularly in areas with thick layers of ericaceous shrubs. Clinton and others (1998) found that flame lengths range from 0.3 to 1.5 m for backing fires and from 1.2 to 4.5 m for head fires. The rate of spread varied from 1.8 to 3.0 m/minute for head fires to 0.3 m/minute for backing fires. Maximum flame temperatures ranged from 260 to 704 °C. On average, about half the mass of small wood (<8 cm d.b.h.) and litter was lost, and about 20 percent of the humus layer was lost.

Burning can be overly damaging to white pine because the species has thin bark a crowns low to the ground, particularly when young.

Limitations of prescribed fire

Future use of prescribed fire may be reduced by smoke management requirements, lack of fiscal resources, operational complexities within the wildland-urban interface, and concern for litigation arising from smoke impacts or prescribed fire escapes. In the absence of prescribed fire, fuels continue to accumulate, making the application of alternative treatments necessary. These methods primarily include mechanical or chemical treatment alone or in combination of mechanical with prescribed fire.

Other Fuel Management Techniques

Mechanical treatment

Although the use of mechanical fuel reduction treatments is currently limited, they may be useful alternatives in areas where the risks associated with prescribed fires are unacceptable. Mechanical treatments may lack many of the ecological benefits of fire and are typically more expensive to apply. In the Western United States, mechanical fuel treatments usually include some degree of thinning followed by various methods of yarding and treatment of residual slash, possibly with prescribed fire (Youngblood and others 2007). Because mechanical treatment of Appalachian forest fuels has been limited, not much historical information is available on its effects.

Recent results from one site of the National Fire and Fire Surrogate explicitly addressed effectiveness of mechanical fuel treatments (Waldrop and others 2007) as well as providing a detailed look at the effects of fuel-reduction treatments on forest structure in western North Carolina. The mechanical treatment involved chainsaw felling of stems >1.8 m tall and <10.2 cm d.b.h. and of all shrubs regardless of size. In addition, two prescribed fires—one with and the other without the mechanical treatment—were conducted at a 3-year interval. After 5 years the mechanical treatment alone had no effect on basal area and structure of overstory trees. Density of hardwood saplings decreased initially but slowly returned to levels similar to pretreatment levels as a result of vigorous sprouting. Shrubs—including the dominant shrub species, mountain laurel and rhododendron—experienced a large initial decrease and recovered to less than half their pretreatment abundance by year five.

The combination of mechanical treatment with prescribed fires reduced basal area from 23.8 to 16.6 m^2 /ha after 5 years. Density of hardwood saplings decreased initially but was >100 percent of the pretreatment level after 3 years; a second prescribed fire reduced hardwood sapling density to just slightly higher than the pretreatment levels 2 years later. Cover of all shrubs, including mountain laurel and rhododendron, initially decreased to near zero after the mechanical and burn treatment and remained at very low levels until year five.

Chemical treatment

Herbicides have been studied in the Southern Appalachian Mountains for competition control to favor pines and oaks (Kass and Boyette 1998, Loftis, 1985, Lorimer and others 1994, Neary and others 1984) and for habitat of some wildlife species, including small mammals (McComb and Rumsey 1982) and herpetofauna (Harpole and Haas 1999). However, no study has examined herbicide use for fuel reduction in the area. This treatment may be viable where fire or mechanical treatments are impractical—such as along the wildland-urban interface or on steep inaccessible slopes—but its impacts are unknown. Studies in the pine flatwoods of Florida (Brose and Wade 2002) and in Gulf Coast longleaf pine (Haywood 2009) show short-term increases in fuel loading, which led to increases in fire intensity and damage. Similar results could occur in the Southern Appalachian Mountains, although differences in species composition make impacts difficult to predict. Waldrop and others (2010) showed increased fire intensity for 5 years after chainsaw felling shrubs and small trees in the Southern Appalachians. Although untested, a similar pattern would likely occur if herbicides had been used instead.

Combining Fuel Treatments

Despite the absence of a large body of information from different ecosystems on the effects of fuel- treatment alternatives to fire, the existing literature offers some evidence of differences in the effectiveness of treatment options. Results from the National Fire and Fire Surrogate Study in oak ecosystems strongly suggest that combining removal of shrubs and small trees with prescribed fire is the most effective way to control mountain laurel and other ericaceous shrubs, a fuel of concern in the Southern Appalachian Mountains (Youngblood and others 2007). Studies from a variety of ecosystems consistently demonstrate the ability of mountain laurel and other ericaceous shrubs to increase rapidly after treatment. Whether these fuels respond to mechanical and burn treatments in other ecosystem types is unknown. The ubiquity of mountain laurel and other ericaceous shrubs across the landscape suggests that the response of different ecosystems could be similar. However, interactions with other variables, such as moisture patterns and disturbance regimes, could produce different responses. Results from future studies in other ecosystems could shed light on whether combining mechanical and burn treatments is only effective in oak forests or could be useful across the landscape.

The feasibility of widespread application of a mechanical plus burning treatment is questionable because treating large areas is expensive and time consuming. In addition, mechanical treatments alone are not effective, so the risks associated with prescribed burning are still a factor. Although there is no one solution, the use of mechanical treatment may be most useful in areas with immediate needs for hazardous fuels treatment, such as the wildland-urban interface. Also, mechanical treatments can be very effective in preparing long unburned sites for prescribed burning. Clearly a manager must be flexible and open to cautiously experimenting with different combinations of techniques, drawing on experience and observation until more experimental data become available.

Prioritizing areas for treatment is critical to allocate resources most effectively. Fuel treatment prioritization hinges on managers making decisions that will protect vital assets without decreasing the amount of acreage that is in an acceptable condition. For example, a best management practice would be to focus initial efforts on maintaining areas that currently have low fuel loads and are simple to burn, and only afterward allocating resources to problem areas so that the total amount of untreated acreage does not increase. A burn prioritization model can streamline treatment programs and be useful for mapping current conditions and designating treatments within a spatial context (Hiers and others 2003).

The diversity and productivity of ecosystems in the Southern Appalachian Mountains coupled with a complex disturbance regime poses a challenge for fuels management. Understanding this relationship will better enable managers to understand the dynamic interactions among disturbances, which can alter fuel loads over short periods of time. Although rates of decomposition across the area are rapid, increases in dead and downed fuels following disturbance may create pulses in the abundance of hazardous fuels. In this situation, understanding the variation in the fuel distribution over time may be as important as understanding spatial variations. This is especially pertinent in the context of climate change scenarios that predict more frequent droughts and warmer temperatures that could exacerbate the effects of disturbances such as native and nonnative insects and pathogens. These effects could be especially important in long-unburned mature stands that contain older decadent individual trees and well developed shrub layers. Effective mitigation of these threats depends on effective fuels monitoring at large scales and adaptive management to meet future challenges.

Research Needs

Prescribed burning is a relatively new tool in the Southern Appalachian Mountains. As a result, less is known about fuel reduction treatment impacts in this area than is known in other areas of the United States. Critical research needs include the studying the impacts of mechanical and chemical treatments, comparing season and frequency of prescribed burning, and identifying the cumulative effects of repeated fuel reduction treatments over many years. More information is needed to understand the impacts of these treatments on most components of the ecosystem—biotic and abiotic—and the probability of introducing new and possibly unwanted components, such as nonnative invasive plants.

Research on smoke prediction in the Southern Appalachian Mountains is just beginning and is extremely difficult because of the complex topography and weather patterns that must be considered.

Literature Cited

Abbott, D.T.; Crossley, D.A., Jr. 1982. Woody litter decomposition following clearcutting. Ecology. 63: 35–42.

Abrams, M.D. 1992. Fire and the development of oak forests. BioScience. 42: 346–353.

Agee, J.K.; Skinner, C.N. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management. 211: 83–96.

Barden, L.S. 1978. Regrowth of shrubs in grassy balds of the Southern Appalachians after prescribed fire. Castanea. 43(4): 238–246.

Barden, L.S. 1997. Historic prairies in the Piedmont of North and South Carolina, USA. Natural Areas Journal. 17: 149–152.

Barden, L.S.; Woods, F.W. 1973. Characteristics of lightning fires in Southern Appalachian forests. Proceedings, 13th annual Tall Timbers fire ecology conference. Tallahassee, FL: Tall Timbers Research Station: 345–361.

Blake, J.G.; Hoppes, W.G. 1986. Influence of resource abundance on use of tree-fall gaps by birds in an isolated woodlot. Auk. 103: 328–340.

Bolstad, P.V.; Vose, J.M.; McNulty, S.G. 2001. Forest productivity, leaf area, and terrain in Southern Appalachian deciduous forests. Forest Science. 47: 419–427.

Brose, P.H.; Wade, D. 2002. Potential fire behavior in pine flatwood forests following three different fuel reduction techniques. Forest Ecology and Management. 163: 71–84.

Brose, P.H.; Waldrop, T.A. 2006. Fire and the origin of Table Mountain pine–pitch pine communities in the Southern Appalachian Mountains, USA. Canadian Journal of Forest Research. 36: 710–718.

Buckner, E.R. 1989. Evolution of forest types in the Southeast. In: Waldrop, T.A., ed. Proceedings of pine-hardwood mixtures: a symposium on management and ecology of the type. Gen. Tech. Rep. SE-58. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 27–33.

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

Chastain, R.A.; Townsend, P.A. 2008. Role of evergreen understory shrub layer in the forests of the central Appalachian highlands. Journal of the Torrey Botanical Society. 135(2): 208–223.

Clinton, B.D.; Vose, J.M.; Swank, W.T. [and others]. 1998. Fuel consumption and fire characteristics during understory burning in a mixed white pine-hardwood stand in the Southern Appalachians. Res. Pap. SRS-12. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 8 p.

Delcourt, H.R.; Delcourt, P.A. 1997. Pre-Columbian Native American use of fire on Southern Appalachian landscapes. Conservation Biology. 11: 1010–1014.

DeVivo, M.S. 1991. Indian use of fire and land clearance in the Southern Appalachians. In: Nodvin, S.C.; Waldrop, T.A., eds. Fire and the environment: ecological and cultural perspectives. Gen. Tech. Rep. SE-69. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 306–312.

Elliott, K.J.; Hendrick, R.H.; Major, A.E. [and others]. 1999. Vegetation dynamics after a prescribed fire in the Southern Appalachians. Forest Ecology and Management. 114: 199–213.

Elliott, K.J.; Swank, W.T. 1994. Impacts of drought on tree mortality and growth in a mixed hardwood forest. Journal of Vegetation Science. 5: 229–236.

Elliott, K.J.; Vose, J.M. 2005. Effects of understory prescribed burning on shortleaf pine (*Pinus echinata* Mill.)/mixed-hardwood forests. Journal of the Torrey Botanical Society. 132: 236–251.

Fowler, C.; Konopik, E. 2007. The history of fire in the Southern United States. Human Ecology Review. 14: 165–176.

Frost, C.C. 1995. Presettlement fire regimes in southeastern marshes, peatlands, and swamps. In: Cerulean, S.I.; Engstrom, R.T., eds. Fire in wetlands: a management perspective. Proceedings, 19th Tall Timbers fire ecology conference. Tallahassee, FL: Tall Timbers Research Station: 39–60.

Garten, C.T., Jr.; Hanson, P.J. 2006. Measured forest soil C stocks and estimated turnover times along an elevation gradient. Geoderma. 136: 342–352.

Greenberg, C.H.; Levey, D.J.; Loftis, D.L. 2007. Fruit production in mature and recently regenerated upland and cove hardwood forests of the Southern Appalachians. Journal of Wildlife Management. 71: 321–335.

Harmon, M.E. 1982. Fire history of the westernmost portion of the Great Smoky Mountains National Park. Bulletin of the Torrey Botanical Society. 109: 74–79.

Harmon, M.E.; Bratton, S.P.; White, P.S. 1984. Disturbance and vegetation response along environmental gradients in the Great Smoky Mountains. Plant Ecology. 55: 129–139.

Harpole, D.N.; Haas, C.A. 1999. Effects of seven silvicultural treatments on terrestrial salamanders. Forest Ecology and Management. 114: 349–356.

Haywood, J.D. 2009. Eight years of seasonal burning and herbicidal brush control influence sapling longleaf pine growth, understory vegetation, and the outcome of an ensuing wildfire. Forest Ecology and Management. 258: 295–305.

Hiers, J.K.; Laine, S.C.; Bachant, J.J. [and others]. 2003. Simple spatial modeling tool for prioritizing prescribed burning activities at the landscape scale. Conservation Biology. 17: 1571–1578.

Hubbard, R.M.; Vose, J.M.; Clinton, B.D. [and others]. 2004. Stand restoration burning in oak-pine forests in the Southern Appalachians: effects on aboveground biomass and carbon and nitrogen cycling. Forest Ecology and Management. 190: 311–321.

Iverson, L.R.; Yaussy, D.A.; Rebbeck, J. [and others]. 2003. Spatial and temporal distribution of fire temperatures from prescribed fires in the mixed oak forests of southern Ohio. In: Van Sambeek, J.W.; Dawson, J.W.; Ponder, J.O. [and others], eds. Proceedings of the 13th central hardwood forest conference. Gen. Tech. Rep. NC-234. St. Paul, MN: U.S. Department of Agriculture Forest Service, North Central Research Station: 293–294.

Kass, D.J.; Boyette, W.G. 1998. Preharvest herbicide method to develop competitive oak reproduction in upland oaks stands in the mountains and Piedmont of North Carolina: 7-year results. In: Waldrop, T.A., ed. Proceedings of the ninth biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-20. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 253–257.

Kolaks, J.; Cutter, B.E.; Lowenstein, E.F. [and others]. 2003. Fuel loading in the central hardwoods. In: Proceedings of the 2nd international wildland fire ecology and fire management congress [CD–ROM]. Pap. 1A-3. Boston: American Meteorology Society. http://www.ametsoc.org/exhibitr/firecongressexhprospectus.pdf. [Date accessed: July 23, 2011].

Kolaks, J.; Cutter, B.E.; Lowenstein, E.F. [and others]. 2004. The effect of thinning and prescribed fire on fuel loading in the central hardwood region of Missouri. In: Yaussy, D.A.; Hix, D.M.; Long, R.P.; Goebel, P.C., eds. Proceedings of the 14th central hardwood forest conference. Gen. Tech. Rep. NE-316. Newtown Square, PA: U.S. Department of Agriculture Forest Service, Northeastern Research Station: 168–178.

Lafon, C.W.; Hoss, J.A.; Grissino-Mayer, H.D. 2005. The contemporary fire regime of the central Appalachian Mountains and its relation to climate. Physical Geography. 26: 126–146.

Loftis, D.L. 1985. Preharvest herbicide treatment improves regeneration in Southern Appalachian hardwoods. Southern Journal of Applied Forestry. 9: 177–180.

Lorimer, C.G.; Chapman, J.W.; Lambert, W.D. 1994. Tall understory vegetation development as a factor in the poor development of oak seedlings beneath mature stands. Journal of Ecology. 82: 227–237.

McComb, W.C.; Rumsey, R.L. 1982. Response of small mammals to forest clearings created by herbicides in the central Appalachians. Brimleyana. 8: 121–134.

McLeod, D.E. 1988. Vegetation patterns, floristics, and environmental relationships in the Black and Craggy Mountains. Chapel Hill, NC: University of North Carolina. [Number of pages unknown]. Ph.D. dissertation.

McNab, W.H.; Cleland, D.T.; Freeouf, J.A.; [and others]. 2007. Description of ecological subregions: sections of the conterminous United States [CD-ROM]. Gen. Tech. Rep. WO-76B. Washington, DC: U.S. Department of Agriculture Forest Service, Southern Research Station: 80 p.

McNab, W.H.; Greenberg, C.H.; Berg, E.C. 2004. Landscape distribution and characteristics of large hurricane-related canopy gaps in a Southern Appalachian watershed. Forest Ecology and Management. 196: 435–447.

Neary, D.G.; Douglass, J.E.; Ruehle, J.L.; Fox, W. 1984. Converting rhododendron-laurel thickets to white pine with picloram and mycorrhizae-innoculated seedlings. Southern Journal of Applied Forestry. 8(3): 163–168.

Newell, C.L.; Peet, R.K.; Ulrey, C.J. [and others]. 1999. Geographic variation in forest distribution across five landscapes in the Southern Appalachian Mountains of North and South Carolina. In: Eckerlin, R.P., ed. Proceedings of the Appalachian biogeography symposium. Spec. Publ. 7. Martinsville, VA: Virginia Museum of Natural History: 19–34.

Nowacki, G.J.; Abrams, M.D. 2008. The demise of fire and "mesophication" of forests in the Eastern United States. BioScience. 58(2): 123–138.

Olano, J.M.; Palmer, M.W. 2003. Stand dynamics of an Appalachian old-growth forest during a severe drought episode. Forest Ecology and Management. 174: 139–148.

Phillips, D.R.; Abercrombie, J.A. 1987. Pine-hardwood mixtures—a new concept in regeneration. Southern Journal of Applied Forestry. 11: 192–197.

Phillips, R.J.; Waldrop, T.A.; Simon, D.M. 2006. Assessment of the FARSITE model for predicting fire behavior in the Southern Appalachian Mountains. In: Connor, K.F., ed. Proceedings of the 13th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-92. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 521–525.

Reilly, M.J.; Wimberly, M.C.; Newell, C. 2006. Effects of a wildfire on beta diversity and species turnover in a forested landscape. Journal of Vegetation Science. 17: 447–454.

Smith, G.F.; Nicholas, N.S. 2000. Size- and age-class distributions of Frasier fir following balsam woolly adelgid infestation. Canadian Journal of Forest Research. 30: 948–957.

Stueve, K.M.; Lafon, C.W.; Isaacs, R.E. 2007. Spatial patterns of ice storm disturbance on a forested landscape in the Appalachian Mountains, Virginia. Area. 39: 20–30.

Southern Appalachian Man and the Biosphere Program. 1996a. The Southern Appalachian Assessment summary report. Southern Appalachian Man and Biosphere Program. Atlanta, GA: U.S. Department of Agriculture Forest Service, Southern Region. 164 p.

Southern Appalachian Man and the Biosphere Program. 1996b. The Southern Appalachian Assessment, terrestrial technical report. Chapter 2: Wildlife and plant species. Southern Appalachian Man and Biosphere Program. Atlanta, GA: U.S. Department of Agriculture Forest Service, Southern Region: 11–19.

Van Lear, D.H.; Waldrop, T.A. 1989. History, use and effect of fire in the Southern Appalachians. Gen. Tech. Rep. SE-54. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station. 20 p.

Vose, J.M. 2000. Perspectives on using prescribed fire to achieve desired ecosystem conditions. In: Proceedings, 21st Tall Timbers fire ecology conference. Tallahassee, FL: Tall Timbers Research Station: 12–17.

Vose, J.M.; Swank, W.T.; Clinton, B.D. [and others]. 1999. Using stand replacement fires to restore Southern Appalachian pine-hardwood ecosystems: effects on mass, carbon, and nutrient pools. Forest Ecology and Management. 114: 215–226.

Waldrop, T.; Phillips, R.J.; Simon, D.A. 2010. Fuels and predicted fire behavior in the Southern Appalachian Mountains following fire and fire surrogate treatments. Forest Science. 56(1): 32–45.

Waldrop, T.A. 1996. Dynamics of coarse woody debris – a simulation study for two southeastern forest ecosystems. In: Crossley, D.A., Jr.; McMinn, J.A., eds. Biodiversity and coarse woody debris in southern forests. Gen. Tech. Rep. SE-94. Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 18–24.

Waldrop, T.A.; Brose, P.H. 1999. A comparison of fire intensity levels for stand replacement of Table Mountain pine (*Pinus pungens* Lamb.). Forest Ecology and Management. 113: 155–166.

Waldrop, T.A.; Brudnak, L.; Rideout-Hanzak, S. 2007. Fuels on disturbed and undisturbed sites in the Southern Appalachian Mountains. Forest Ecology and Management. 37: 1134–1141.

Waldrop, T.A.; Glass, D.W.; Rideout, S. [and others]. 2004. An evaluation of fuel reduction treatments across a landscape gradient in Piedmont forests: preliminary results of the National Fire and Fire Surrogate Study. In: Connor, K.F., ed. Proceedings of the 13th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-92. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 54–57.

Waldrop, T.A.; Yaussy, D.A.; Phillips, R.J. [and others]. 2008. Fuel reduction treatments affect stand structure of hardwood forests in western North Carolina and southern Ohio, USA. Forest Ecology and Management. 255: 3117–3129.

Welch, N.T.; Waldrop, T.A.; Buckner, E.R. 2000. Response of Southern Appalachian Table Mountain pine (*Pinus pungens* Lamb.) and pitch pine (*P. rigida*) stands to prescribed burning. Forest Ecology and Management. 136: 185–197.

Whitehead, M.A. 2003. Seasonal variation in food resource availability and avian communities in 4 habitat types in the Southern Appalachian Mountains. Clemson, SC: Clemson University. [Number of pages unknown]. Ph. D. dissertation.

Whittaker, R.H. 1956. Vegetation of the Great Smoky Mountains. Ecological Monographs. 26: 1–80.

Williams, C.E.; Johnson, W.; Carter, W. 1990. Age structure and the maintenance of *Pinus pungens* in pine-oak forests of southwestern Virginia. The American Midland Naturalist. 124: 130–141.

Youngblood, Andrew; Metlen, Kerry; Knapp, Eric E. [and others]. 2007. Implementation of the fire and fire surrogate study—a national research effort to evaluate the consequences of fuel reduction treatments. In: Peterson, Charles E.; Maguire, Douglas A., eds. Balancing ecosystem values: innovative experiments for sustainable forestry. Proceedings of a conference. Gen. Tech. Rep. PNW-GTR-635. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station: 315–321.