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Chapter 6:

Fire and Nonnative Invasive Plants in the Southeast Bioregion

Introduction _____

This chapter identifies major concerns about fire and nonnative invasive plants in the Southeast bioregion. The geographic area covered by this chapter includes the entire States of Louisiana, Mississippi, and Florida; all except the northernmost portions of Delaware and Maryland; the foothill and coastal ecosystems of Virginia, North Carolina, South Carolina, Georgia, and Alabama; and the lower elevation plant communities of Arkansas, southeastern Missouri, southeastern Oklahoma, southwestern Tennessee, and eastern Texas. This area coincides with common designations of the Atlantic Coastal Plain and the Piedmont (the plateau region between the Atlantic and Gulf of Mexico Coastal Plain and the Appalachian Mountains). Soils are generally moist year-round, with permanent ponds, lakes, rivers, streams, bogs, and other wetlands. Elevations vary from 2,407 feet (734 m) on Cheaha Mountain, Alabama, to –8 feet (–2.4 m) in New Orleans, Louisiana (USGS 2001). Westerly winds bring winter precipitation to the bioregion, and tropical air from the Gulf of Mexico, Atlantic Ocean, and Caribbean Sea brings summer moisture. Southward through this region the contribution of winter rainfall decreases, as

does the frequency of freezing temperatures. Tropical conditions occur at the southern tip of Florida. The percentage of evergreen species and palms (*Serenoa* spp., *Sabal* spp.) increases along this climate gradient (Daubenmire 1978).

Plant communities within this portion of the temperate mesophytic forest are complex and subject to a long history of natural and anthropogenic disturbance. Various methods have been used to estimate the dominant presettlement forest types. Plummer (1975) reported that pine (*Pinus* spp.) and post oak (*Quercus stellata*) were the dominant trees on historical survey corner tree lists in the Georgia Piedmont, and Nelson (1957) used soil type to estimate that 40 percent of the Piedmont was dominated by hardwood species, 45 percent was in mixed hardwood and pine stands, and 15 percent was predominantly pine. On the southeastern Coastal Plain, pine savannas may have covered between two-thirds and three-fourths of the area (Platt 1999).

Currently, forests include a mosaic of mostly deciduous angiosperms that form a dense canopy of tall trees with a "diffuse" layer of shorter, shade-tolerant trees, interspersed with disturbance- (mostly fire-) derived pine stands (Daubenmire 1978). Vines are common and

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frequently include native grape (*Vitis* spp.) species. Large streams often have extensive floodplains and oxbows, and areas where the water table occurs at or near the surface year-round usually support stands of bald cypress (*Taxodium distichum*). Coastal dunes are often populated by American beachgrass (*Ammophila breviligulata*) from North Carolina northward and by sea-oats (*Uniola paniculata*) throughout the region. Salt water-influenced wetlands occur landward of coastal dunes and are dominated by a variety of species including inland saltgrass (*Distichlis spicata*), needlegrass rush (*Juncus roemerianus*), smooth cordgrass (*Spartina alterniflora*), and saltmeadow cordgrass (*Spartina patens*) (Daubenmire 1978).

While nonnative plants can be found throughout this region, the highest proportion of nonnative plants is found in southern Florida (Ewel 1986; Long 1974). Prior to this century's increase in transport and trade, the southern Florida peninsula had geographical and geological barriers to plant species introductions from the north, and surrounding waters provided barriers to tropical species introductions. More recently, southern Florida has become especially vulnerable to nonnative plant invasions because of a large number of temperate and tropical species introductions for horticulture (Gordon and Thomas 1997), the proximity of the introduction pathways to potentially invasible habitats, and the relatively depauperate native flora (Schmitz and others 1997). Additional human-caused changes in hydrology, fire regime, and salinity have combined to increase the vulnerability of the vast low elevation freshwater wetlands south of Lake Okeechobee (Hofstetter 1991; Myers 1983).

Discussion of fire and nonnative plant interactions is complicated by the limited number of experimental field studies, the lack of a complete understanding of presettlement fire regimes, and the unknown effects of increasing atmospheric carbon dioxide and nitrogen and other aspects of climate change (Archer and others 2001). Complications notwithstanding, a better comprehension of the factors and forces at work is critical to developing fire and other habitat management practices that provide a more effective means of achieving ecological and societal objectives (D'Antonio 2000).

Fire in the Southeast Bioregion

Naturally occurring fires are, and were, common in this region (for example, Chapman 1932; Harper 1927; Komarek 1964; Platt 1999; Stanturf and others 2002). The Southeast includes locations with some of the highest lightning incidence levels on earth. Six of the eight highest lightning-strike rates in the United States are found in the southeastern region (Tampa-Orlando, Florida; Texarkana, Arkansas; Palestine, Texas; Mobile, Alabama; Northern Gulf of Mexico; and Gulf Stream-East Carolinas).

Little is known about "natural" or prehistoric fire regimes in the Southeast. During the interval between the retreat of the ice 18,000 years ago and the initial influence of Native Americans beginning around 14,000 years ago, variations in soil moisture, lightning strikes, fuel accumulation, and disturbance history likely resulted in a wide range of fire-return intervals, from as short as one year to as long as centuries. Similarly, fire severities probably ranged from minor fires in the understory to stand-replacement events (Stanturf and others 2002).

Fire frequency and severity were important factors in the evolution of southeastern plant communities (Komarek 1964, 1974; Platt 1999; Pyne 1982a; Pyne and others 1984; Snyder 1991; Van Lear and Harlow 2001; Williams 1989), and fire contributes to the high diversity of communities such as pine and shrub (pineland) communities of south Florida (Snyder 1991) and pine savannas (Platt 1999). Estimates of presettlement fire regime characteristics are summarized in reviews by Wade and others (2000) and Myers (2000) for major vegetation types in the Southeast bioregion.

Substantial evidence from many disciplines supports the contention that fire was widespread prior to European arrival (Stanturf and others 2002). Fires induced by native peoples created and maintained open woodlands, savannas, and prairies (McCleery 1993; Williams 1989), and kept forests in early successional plant communities. Native peoples often burned up to twice a year and extended the fire season beyond summer lightning-induced fires (Van Lear and Harlow 2001).

After adopting the practices and utilizing the clearings made by native people, European settlers influenced fire patterns and plant communities by expanding areas of agricultural clearing and repeated burning (Brender and Merrick 1950; Stoddard 1962; Williams 1992), maintaining permanent fields (Stanturf and others 2002), introducing large herds of hogs and cattle (McWhiney 1988; Stanturf and others 2002; Williams 1992), and heavily logging coastal pine forests, bald cypress, and bottomland hardwood stands (Stanturf and others 2002; Williams 1989). Frequent anthropogenic burning, in combination with grazing cattle and feral pigs, eliminated regeneration of pine and other woody species in large areas (Brender and Merrick 1950; Frost 1993).

Subsequent land and fire management practices and policies oscillated between periods of controlled burning and fire exclusion, and varied from place to place (Brueckheimer 1979; Johnson and Hale 2000; Paisley 1968; Stoddard 1931). This range of fire practices was not the result of carefully planned and organized management strategies but instead was a reaction to political and social influences at a variety of geographical scales, local to regional. Little scientific information

was available, especially in the early years, to inform the ongoing debate over fire exclusion and controlled burning (Frost 1993). Intentional burning practices rarely attempted to mimic presettlement fire conditions (Doren and others 1993; Drewa and others 2002; Platt 1999; Platt and Peet 1998; Slocum and others 2003) but were conducted mainly for agriculture and land clearing.

Contemporary objectives of controlled burns in the Southeast bioregion include hazard fuels reduction, wildlife habitat improvement, and range management (Wade and others 2000). Increasing numbers of acres are being burned for ecosystem restoration and maintenance (Stanturf and others 2002) and to sustain populations of rare and endangered plants (Hessl and Spackman 1995, review; Kaye and others 2001; Lesica 1996). Contemporary fire management practices often strive to recreate presettlement fire regimes, assuming that this will promote maximum diversity (Good 1981; Roberts and Gilliam 1995). Because presettlement fire regimes are not always well understood, however, it is difficult to design a fire management program to meet this objective (Slocum and others 2003).

The negative consequences of past fire management practices have been interpreted as an "ecological disaster" (Brenner and Wade 2003). Exclusion of fire from southeastern pine savannas, for instance, has been blamed for loss of fire-adapted, species-rich herbaceous ground cover and subsequent increase in less fire-tolerant native and nonnative woody species (DeCoster and others 1999; Heyward 1939; Platt 1999; Slocum and others 2003; Streng and others 1993; Walker and Peet 1983).

Fire and Invasive Plants in the Southeast Bioregion

Fire can contribute to the establishment and spread of nonnative invasive plants under some circumstances (Mack and D'Antonio 1998). Melaleuca (*Melaleuca quinquenervia*), for instance, invades fire-cleared mineral soils in south Florida (Myers 1975).

Fire exclusion has also been blamed for reducing native species in favor of nonnatives in fire-adapted communities. For example, Chinese tallow (*Triadica sebifera*) invades fresh marshes (Grace 1999), and Brazilian pepper (*Schinus terebinthifolius*) invades subtropical pine habitats (Myers 2000) in the absence of fire. Exclusion of fire from longleaf pine (*Pinus palustris*) communities generally results in woody species overtopping herbs, thicker duff layers, and changes in nutrient availability that "all favor extrinsic species at the expense of endemic residents" (Wade and others 2000, page 66).

Nonnative plant invasions can affect fuel and fire characteristics in invaded communities (Brooks and others 2004; Chapter 3) and may subsequently reduce native plant density and diversity. Altered fuel characteristics associated with some invasive species may result in fires that kill native plants but not fire-resistant invasive species (Drake 1990; Pimm 1984). For example, cogongrass (*Imperata cylindrica*) invasions in Florida sandhills increase biomass, horizontal continuity, and vertical distribution of fine fuels, compared to uninvaded pine savanna. Fires in stands invaded by cogongrass have higher maximum temperatures than fire in uninvaded stands (Lippincott 2000) and may therefore cause greater mortality in native species than fires fueled by native species. Melaleuca invasion can alter the vertical distribution of fuels such that communities that typically experienced low-severity surface fires have a greater incidence of crown fire in invaded communities (Myers 2000). Conversely, Brazilian pepper and Chinese tallow develop dense stands that suppress native understory grasses, resulting in lower fine fuel loads than the fire-maintained plant communities being replaced (Doren and others 1991; Grace and others 2001). Lower fuel loads may lead to reduced fire frequency and lower fire severity, which may favor the fire sensitive seedling stages of the invasives (Mack and D'Antonio 1998).

Controlled burning is sometimes used in an effort to manage invasive plants in the Southeast. However, D'Antonio's review (2000) suggests that fire-versus-invasives results are highly variable and depend on fire intensity, time of burning (Hastings and DiTomaso 1996; Parsons and Stohlgren 1989; Willson and Stubbendieck 1997), weather, and the status of the remaining seed bank (Lunt 1990; Parsons and Stohlgren 1989). It is also important to note that, while dormant season fires have been recommended to control invasive shrubs in grasslands, they may result in increases in nonnatives (Richburg and others 2001, review). It has been recommended that, if fire is used to reduce populations of nonnative invasive plants, burning should be timed to reduce flowers and/or seed production, or at the young seedling/sapling stage (chapter 4). Spot-burning very small populations of invasive plants has also been recommended as "cheaper and easier than implementing a prescribed fire" (Tu and others 2001).

The limited number of replicated, long-term, field experiments on fire and invasive plants reflects the very difficult nature of conducting the needed studies. Even where detailed measurements of the effects of fire on native and nonnative plant species have been collected, the studies are typically short-term and do not necessarily reflect longer-term changes (Freckleton 2004; Freckleton and Watkinson 2001). More research is needed over longer periods of time to better understand the relationships between fire and invasive species in the Southeast bioregion.
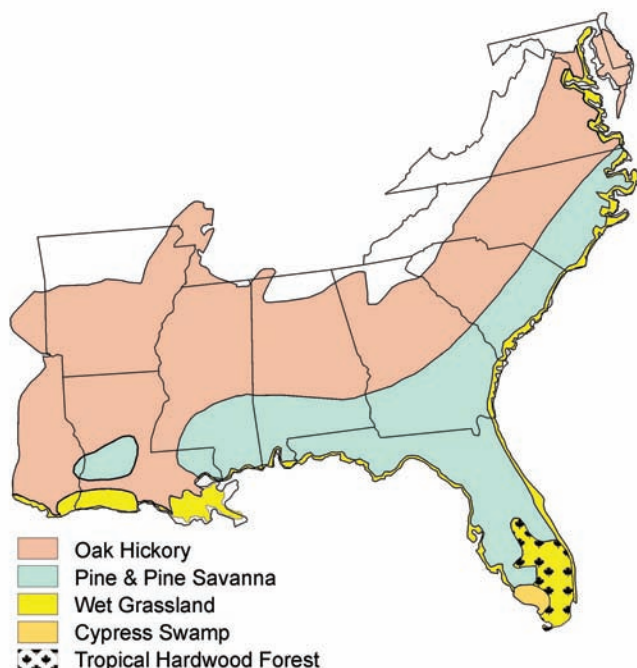
The remainder of this chapter presents information on the known relationships of fire and invasive plant species for five major plant habitats: wet grassland, pine and pine savanna, oak-hickory (*Quercus – Carya*) woodland, tropical hardwood forest, and a brief treatment of cypress (*Taxodium distichum*) swamp (fig. 6-1). For each habitat except cypress swamp, a summary is provided of the role of fire and fire exclusion in promoting invasions by nonnative plant species, fire regimes changed by plant invasions, and use of fire to manage invasive plants, with an emphasis on those species included in table 6-1. The final section presents general conclusions and emerging issues relating to fire and invasive species management in the Southeast bioregion. All parts of the Southeast have been and continue to be affected by management practices including fire exclusion, controlled burning, or both (Brenner and Wade 2003; Freckleton 2004). Therefore, we have made no attempt to make a distinction between "more managed" (for example, pine plantations) and "less managed" (for example, conservation areas) ecosystems in this section.

Wet Grassland Habitat _____

Background

The term "wet grasslands" is used here to include the Everglades region of southern Florida, grassland

Oak Hickory
Pine & Pine Savanna
Wet Grassland
Cypress Swamp
Tropical Hardwood Forest

Figure 6-1—Approximate distribution of major plant habitats in the Southeast bioregion. Upland grasslands and palmetto prairie are not identified here.

savannas with cabbage palm (*Sabal palmetto*) and cypress in Florida (Küchler's (1964) palmetto prairie and cypress savanna, respectively), and the coastal grass-dominated wetlands from Virginia to Texas (Küchler's (1964) northern and southern cordgrass (*Spartina* spp.) prairie). Native species in these wetlands include smooth cordgrass dominating tidally flushed saltmarshes; smooth and gulf cordgrass (*Spartina spartinae*), needlegrass rush, pickleweed (*Salicornia spp.*), inland saltgrass, saltmeadow cordgrass, and saltmeadow rush (*Juncus gerardii*) in less frequently flooded more inland marshes. Aquatic species in fresh marshes include pond-lily (*Nuphar* spp.), waterlily (*Nymphaea* spp.), wild rice (*Zizania aquatica*), cutgrass (*Zizaniopsis miliacea*), pickerelweed (*Pontedaria cordata*), arrowhead (*Sagittaria* spp.), cattail (*Typha* spp.), maidencane (*Panicum hemitomon*), spikerush (*Eleocharis* spp.), and sedges (*Carex* spp.) (Wade and others 2000).

While wet grassland communities border many different habitats, some of the smallest non-graminoid dominated types include 1-to-several acre (0.5-to-several hectare) hardwood forest sites found within marsh, prairie, or savanna in southern Florida. Locally termed "tree islands," fire in these locations is usually driven by processes in the adjacent plant community, and special features related to the spread of fire into the tree islands from adjacent wet grasslands are discussed in the "Tropical Hardwood forest" section of this chapter.

Wet grassland plant communities tend to be flammable and are adapted to an environment of frequent wet season (summer) fires (Leenhouts 1982; Schmalzer and others 1991; Wade 1988; Wade and others 1980). The following information on fire regimes in these plant communities comes from Wade and others (2000) and Myers (2000) (see these reviews for more detail). Presettlement fire regimes in wet grasslands in much of the coastal region in the Southeast are classified as stand-replacement types with 1- to 10-year return intervals (Myers 2000; Wade and others 2000).

Fire behavior differs among wet grassland types due to differences in flammability of dominant species, which vary with groundwater levels and salinity. Cordgrass communities in coastal salt marshes tend to be quite flammable, with green tissues of saltmeadow cordgrass and gulf cordgrass capable of burning several times during a growing season. Fire in these communities will carry over standing water. Flammability in fresh and brackish marshes is more variable due to considerable plant diversity. Grass dominated stands generally experience more intense and continuous fires than forb and sedge dominated stands with important exceptions, including cattail and sawgrass stands (*Cladium jamaicense*) (Myers 2000; Wade and others 2000).

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Table 6-1—Major plant habitats in the Southeast bioregion and perceived threat potential of several important nonnative plants in each habitat (L= low threat, H = high threat, P = potentially high threat, N= not invasive, U = unknown). Designations are approximations based on available literature.

Scientific name	Common name	Grassland			Forest and woodland			
		Wet grassland	Upland grassland	Palmetto prairie	Pine and pine savanna	Oak-hickory woodland	Tropical hardwood forest	Cypress swamp
Ailanthus altissima	Tree-of-heaven	N	N	N	L	H	N	N
Albizia julibrissin	Mimosa	N	N	N	U	P	N	N
Alliaria petiolata	Garlic mustard	N	N	N	P	P	N	N
Dioscorea alata	Water yam	N	N	N	L	P	P	L
Dioscorea bulbifera	Air potato	N	N	N	H	H	H	H
Dioscorea oppositifolia	Chinese yam	U	U	N	U	P	U	U
Elaeagnus pungens	Thorny-olive	N	N	N	U	P	N	N
Elaeagnus umbellata	Autumn-olive	U	H	N	U	U	N	N
Euonymus alatus	Winged euonymus	U	U	N	U	P	N	N
Euonymus fortunei	Winter creeper	U	U	N	U	P	N	N
Hedera helix	English ivy	N	N	N	U	P	U	U
Imperata cylindrica	Cogongrass	N	L	U	H	L	N	N
Lespedeza bicolor	Lespedeza	N	H	N	L	L	N	N
Lespedeza cuneata	Sericea lespedeza	N	L	L	L	L	L	N
Ligustrum spp.	Privet species	U	N	U	L	H	N	N
Lolium arundinaceum	Tall fescue	U	H	N	U	N	N	N
Lonicera japonica	Japanese honeysuckle	N	H	N	H	H	N	N
Lonicera spp.	Bush honeysuckles	N	H	N	U	L	U	N
Lygodium japonicum	Japanese climbing fern	U	U	U	H	P	U	U
Lygodium microphyllum	Old World climbing fern	H	N	H	H	N	H	H
Melaleuca quinquenervia	Melaleuca	H	N	H	H	N	H	H
Melia azedarach	Chinaberry	H	H	H	U	P	U	N
Microstegium vimineum	Japanese stiltgrass	P	H	P	U	H	U	U
Paspalum notatum	Bahia grass	U	L	U	L	L	N	N
Pueraria montana var. lobata	kudzu	U	U	U	H	H	U	U
Schinus terebinthifolius	Brazilian pepper	H	P	H	H	N	P	P
Solanum viarum	tropical soda apple	U	H	U	P	U	N	N
Triadica sebifera	Chinese tallow	H	H	H	P	P	U	U

Postfire succession patterns in wet grasslands are influenced by season of burning and the interplay of hydroperiod and fuels, which together determine whether fires are lethal or nonlethal to the dominant species. Hydroperiod factors that influence the effect of fire on belowground plant parts and substrate include proximity to the water table, tidal conditions, and drought cycles. In most cases, the aboveground vegetation is consumed by fire, and the dominant species that make up the fuel sprout from underground buds, tubers, or rhizomes after fire. Peat fires and postfire flooding are two disturbance events that can kill both above- and belowground organs of existing vegetation. Severe peat fires can occur in organic substrates when severe drought coincides with low water table levels. Vegetation can also be killed by water overtopping recovering vegetation after a fire. Successional species would be expected to be primarily those represented in the seed bank (Myers 2000; Wade and others 2000).

In some areas, native wet grassland communities have been altered by fire exclusion, allowing invasion of woody species. Fire is now being reintroduced in many areas to restore native species compositions (for example, see Leenhouts 1982).

Role of Fire and Fire Exclusion in Promoting Nonnative Plant Invasions in Wet Grasslands

Fire exclusion in wet grasslands during the past century has resulted in less frequent but more severe fires than occurred prior to European settlement. These fires have opened up wet grasslands to invasion by nonnative plant species (Bruce and others 1995).

Melaleuca is well adapted to survive fire and to establish and spread in the postfire environment. There may not be many better fire-adapted tree species in the world than melaleuca. Nicknamed the "Australian fireproof tree" (Meskimen 1962), its native habitats include fire-shaped ecosystems in Australia (Stocker and Mott 1981). It produces serotinous capsules (le Maitre and Midgley 1992) that release as many as 20 million seeds per tree (Woodall 1981) following fire. Complete capsule dehiscence can occur as quickly as a few days following a crown fire (Woodall 1983). Seedlings establish on fire-cleared mineral soil and can survive fire within a few weeks or months after germination (Meskimen 1962; Myers 1975, 1983). Melaleuca sprouts from roots and from epicormic trunk buds to resume growth after fire. Although the outer bark can easily burn (fig. 6-2), the trunk wood is protected from fire damage by spongy inner bark that is saturated with water (Turner and others 1998).

Fire is not necessary for melaleuca establishment (Woodall 1981); in fact, melaleuca spreads readily with changes in hydrology and mechanical damage

to habitats (Cost and Carver 1981). Nevertheless, its ability to capitalize on burned areas is remarkable (Hofstetter 1991; Myers 1983). "If melaleuca were managed as a desired species, prescribed fire would be the single most important tool available to the resource manager" (Wade 1981).

Melaleuca seedlings that establish after fire at the height of the dry season may have 5 to 8 months' advantage over native tree species such as south Florida slash pine (*Pinus elliottii* var. *densa*) and bald cypress, which release seed during the wet season (Wade 1981). Other species that may establish and/or spread following fire in wet grasslands include climbing ferns (*Lygodium* spp.) and the shrub chinaberry (*Melia azedarach*). According to a review by Ferriter (2001), Old World climbing fern (*Lygodium microphyllum*) occurs in sawgrass marsh in southern Florida and may spread following fire. No published information was found related to the response of chinaberry to fire in the Southeast; however, it reproduces vegetatively from both stumps and roots following fire in Argentina (Menvielle and Scopel 1999; Tourn and others 1999).

In areas where fire has been excluded from wet grassland communities, native species are often

Figure 6-2—Melaleuca's papery trunk. (Photo by Forest & Kim Starr, United States Geological Survey, Bugwood.org.)

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replaced by a dense woody overstory composed of nonnative invasive trees and/or shrubs (for example, melaleuca, Chinese tallow, Brazilian pepper and chinaberry). When this occurs, native species numbers and diversity are dramatically reduced (for example, Bruce and others 1995). It has been suggested that fire exclusion can contribute to the invasion of Chinese tallow into coastal prairies (Bruce and others 1995; chapter 7; D'Antonio 2000). Similarly, reduced fire frequency due to human-induced changes to hydrology is blamed for invasion of Brazilian pepper into sand cordgrass (*Spartina bakeri*) and black rush (*Juncus roemerianus*) dominated salt marshes in Florida's Indian River Lagoon (Schmalzer 1995).

The relationship between fire and fire exclusion and the increase in Brazilian pepper in south Florida wetlands is not well understood. Brazilian pepper is not a fire-adapted species (Smith, C. 1985) and is generally kept out by fire in adjacent pinelands (Loope and Dunevitz 1981). The response to fire in wetlands may not be the same as in pinelands, although the field studies have not been conducted on "typical" wetland plant communities. The limited published research includes assessment of the effect of repeated fire (generally every two years) on experimental plots in highly altered "rock-plowed" limestone substrate. Much of this formerly agricultural area was originally sawgrass marsh, although many woody species invaded the rock-plowed portions after the fields were abandoned. Fire exclusion (control plots in this study) resulted in increased Brazilian pepper stem density, but Brazilian pepper stem density also increased in burned plots (Doren and others 1991). The effects of fire were related to the size of the Brazilian pepper plant. Smaller, apparently younger plants were badly damaged or killed by fire, while larger plants either recovered completely or did not burn. The conclusion of the authors was that Brazilian pepper invasion progressed with or without fire, and that fire is not an appropriate management tool for this unique area (Doren and others 1991). Dry season wildfires are thought to contribute to increasing Brazilian pepper populations in "tree islands," which are typically tree-dominated areas within the wet grasslands of southern Florida (Ferriter 1997, FLEPPC review) (see "Tropical Hardwood Forest Habitat" section page 107).

Chinaberry may also invade wet grasslands where fire has been excluded. This shrub occurs primarily in disturbed areas but is also said to invade relatively undisturbed floodplain hammocks, marshes, and upland woods in Florida (Batcher 2000b, TNC review), although no additional information is available. In Texas, riparian woodlands and upland grasslands have also been extensively invaded by chinaberry (Randall and Rice unpublished, cited in Batcher 2000b).

Tall fescue (*Lolium arundinaceum*) is found on disturbed upland grassland sites or where "the natural fire regime has been suppressed (Eidson 1997)" (as cited in Batcher 2004, TNC review).

Effects of Plant Invasions on Fuel and Fire Regime Characteristics in Wet Grasslands

Several nonnative invasive plants are thought to change fire regimes in wet grasslands in the Southeast bioregion by changing the quantity and/or quality of fuels in invaded communities. Changes in fuels may subsequently reduce or increase fire frequency and severity. Examples of both cases are evident in wet grassland plant communities.

Observational and limited experimental evidence suggests that invasive hardwoods such as Chinese privet (*Ligustrum sinense*), Chinese tallow, and Brazilian pepper shade out and/or replace native plant species in southeastern marshes and prairies such that fine fuel loads and horizontal continuity are reduced (for example, Doren and Whiteaker 1990; Doren and others 1991; Grace 1999; Platt and Stanton 2003). When this occurs, fire frequency and intensity may be reduced and fire patchiness increased. However, there is little experimental evidence to support these conjectures, and more research is needed to better understand the implications of these vegetation changes on fire regimes.

Melaleuca invasion can have variable effects on fuels and fire behavior. Large amounts of litter under melaleuca stands (Gordon 1998) promote intense and severe fires (Flowers 1991; Timmer and Teague 1991) that are difficult to control and have large potential for economic damage, loss of human life and property, and negative ecological consequences (Flowers 1991; Schmitz and Hofstetter 1999, FLEPPC review; Wade 1981). These high intensity fires promote melaleuca establishment and spread, and reduce cover of native species. Severe fire also removes the outer, highly-flammable melaleuca bark layers, thus reducing the probability of damage to mature melaleuca from subsequent fires (Wade 1981). Thus, a positive feedback loop of fire-promoting-melaleuca and melaleuca-promoting-fire is created (Hofstetter 1991; Morton 1962). Conversely, intense fires fueled by melaleuca may reduce the chances of subsequent fires when organic soils are consumed and the elevation of the soil surface lowered (Schmitz and Hofstetter 1999). Small changes in water level can then theoretically reduce the likelihood of fire by flooding these formerly unflooded sites.

Old World climbing fern alters plant community fuel structure in wet grasslands and associated tree islands with extensive, dry-standing frond "skirts" that create ladder fuels that facilitate fire spread into tree canopies (Ferriter 2001, FLEPPC review)

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(see section on "Tropical Hardwood Forest Habitat" page 107). Roberts (D. 1996) also reports that fire penetrates into wet grasslands from the margins of forested communities where Old World climbing fern has invaded and provides a novel source of additional fuel (see section on "Pine and Pine Savanna Habitat" page 100). Fire spread may also be promoted by pieces of burning fern frond blowing aloft into grasslands from tree islands (Roberts, D. 1996).

While the potential exists for invasive plant species to influence abiotic factors that affect fire behavior, including water table elevation and surface hydrology, this relationship has not yet been shown to be important in wet grasslands in the Southeast bioregion. It is logical to assume, for example, that invasive species that lower the water table through evapotranspiration could reduce soil moisture and thus affect subsequent fire characteristics. It has been suggested that melaleuca increases the amount of water lost to the environment in sites with standing water by adding its evapotranspiration to the water surface evaporation (for example, Gordon 1998; Schmitz and Hofstetter 1999; Versfeld and van Wilgen 1986; Vitousek 1986). In the southwestern United States, transpiration of dense stands of nonnative tamarisk (*Tamarix* spp.) can result in the loss of large quantities of water on sites where the water table is just below the soil surface (Sala and others 1996). In many wet grasslands in the southeastern United States, however, the water table is at or above the soil surface, and evaporation and evapotranspiration are both driven and limited by solar energy. Simply adding another species to the system does not increase the available energy and therefore does not increase the amount of water lost to the atmosphere, although it may increase the available evaporative surface (Allen and others 1997).

Other invasive species-induced changes in hydrology may have some effect on wet grasslands in the Southeast. The thick (over 1 m) rachis mat formed by decades of Old World climbing fern growth may have diverted shallow stream meandering of the Loxahatchee River in east-central Florida by a distance of about 164 feet (50 m) (R. Stocker, personal observation, fall 1997). At this scale only very small portions of the invaded habitat would be affected. Additional study of the relationships among invasive plants and abiotic factors that affect fire regime is warranted.

Use of Fire to Manage Invasive Plants in Wet Grasslands

Controlled burning has been used extensively to manage invasive plants in the Southeast bioregion, with varied results. Only a small portion of the literature describes research on the use of fire to control invasives in wet grasslands.

Many wet grassland sites are on organic soils, and fires occurring when the organic surface soil is dry can consume the peat and affect the type of vegetation that subsequently develops on the site (Ferriter 2001; Myers 2000; Schmitz and Hofstetter 1999). Therefore, it may be possible to prescribe fires that could substantially damage plant roots in wet grasslands during dry periods (Nyman and Chabreck 1995). Documented success using such burns to control invasive nonnative plant species, however, is lacking (Wade and others 2000), and care must be taken to avoid substantial damage to desirable species.

Frequent fires in wet grasslands during historic and prehistoric times are thought to have maintained grasslands with very little woody vegetation (Schmalzer 1995). It follows that prescribed fires with a frequency and seasonality within the reference range of variation experienced in these habitats might favor native wet grassland species over nonnative woody species. Controlled burning following flooding or plant flowering has been suggested as particularly effective in reducing "unwanted woody vegetation" in salt marshes of the St. Johns National Wildlife Refuge (Leenhouts 1982). Fire is not, however, effective for controlling all invasive species in wet grasslands, some of which are well adapted to frequent fires.

Controlled burning has been promoted as a means to reduce woody vegetation in salt marshes (Leenhouts 1982) but has not been effective in controlling melaleuca (Belles and others 1999, FLEPPC review) and has provided mixed results for Chinese tallow (Grace 1999; Grace and others 2001) and chinaberry (Tourn and others 1999).

Controlled burning alone is not effective at controlling melaleuca (Wade 1981) and will not eliminate mature stands (Belles and others 1999). Fires timed to consume seedlings after most germination has occurred have the best potential to control melaleuca (Woodall 1981). Fire can kill melaleuca seedlings less than 6 months old (Belles and others 1999); however, it is difficult to achieve the needed degree of soil surface dryness and fuel load to carry a fire severe enough to prevent postfire sprouting. Melaleuca seedlings less than 1 year old may sprout from root collars after fire damage (Myers 1984). Susceptibility to fire-induced mortality is reduced as seedlings and saplings grow taller, with more than 50 percent of 1.5-foot-(0.5 m) tall saplings surviving in one study (Myers and others 2001). Because melaleuca seeds are able to survive in flooded organic soils for about 1.5 years and in unflooded sandy soils for 2 to 2.3 years (Van and others 2005), postfire establishment from the soil seed bank is also a concern.

Some resource managers maintain that melaleuca control can be achieved with proper timing of prescribed burns (Belles and others 1999; Maffei 1991; Molnar and others 1991; Pernas and Snyder 1999), but the

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success of this approach depends on postfire rainfall, which often does not follow anticipated patterns. Two seasonal windows of opportunity may exist, depending on rainfall patterns (Belles and others 1999). (1) Burning during the late wet season, when surface soils are likely to be moist but not flooded, would encourage melaleuca seed germination just prior to soil dry-down during the dry season. With average dry-season rainfall, melaleuca seedlings are likely to die before the wet season returns the following May or June. If dry-season rainfall is above average, however, melaleuca recruitment is likely to be high. (2) Burning at the beginning of the wet season also encourages seed germination, and normal rainfall patterns might provide sufficient flooding to kill seedlings. Fluctuating rainfall patterns or less than average quantity during the wet season could result in substantial melaleuca recruitment (Belles and others 1999). Because melaleuca has very small wind- and water-dispersed seeds, reproductive and outlying individuals must be killed if long-term reduction of populations is to be achieved (Woodall 1981).

Repeated fires may have potential for controlling melaleuca; however, fuel loads may be insufficient to carry fire in consecutive years. Some wet grasslands might be capable of providing sufficient fuel for a second fire within 2 or 3 years after the first fire. Nearly all melaleuca seedlings were killed in a second fire 2 years after a wildfire in a wet grassland dominated by muhly grass (*Muhlenbergia capillaris*) (Belles and others 1999).

Recommendations for controlling mature melaleuca stands include using fire only after first killing reproductive individuals with herbicide (Myers and others 2001) (fig. 6-3). Herbicide-treated trees release large quantities of viable seed. Prescribed burning should then be conducted within 2 years (6 to 12 months recommended for Big Cypress Preserve; Myers and others 2001) of the herbicide-induced seed release and subsequent germination (Belles and others 1999). Mature melaleuca stands burned by wildfire should have high priority for management because of the potential for postfire spread following seed release from fire-damaged melaleuca or adjacent stands of unburned melaleuca. Recommendations include additional specifications for herbicide use and careful monitoring for several years after fire (Belles and others 1999).

Repeated burning, especially in combination with other control methods, can effectively control Chinese tallow under some circumstances. Chinese tallow is difficult to manage with fire because fuel loads under tallow infestations are often insufficient to carry fire (Grace 1999). See chapter 7 for more information on the use of fire to control Chinese tallow.

The effect of repeated fire (generally every two years) to control Brazilian pepper has been evaluated

Figure 6-3—Burning herbicide-killed melaleuca at South Florida Water Management District. Melaleuca stand was treated with herbicide in spring of 1996 and burned in winter 2001. The objective was to consume a large portion of the standing dead biomass with the fire, but only the tops of trees were burned. (Photo by Steve Smith.)

in highly altered "rock-plowed" limestone substrate in south Florida. Fine fuel supply was insufficient to carry annual fires, which the authors attributed to the replacement of graminoid species with Brazilian pepper (Doren and others 1991). While density and coverage of Brazilian pepper had increased on both burned and unburned plots at the end of the 6 years of evaluation, increases on burned plots occurred more slowly than on unburned plots (Doren and others 1991). Control of Brazilian pepper using herbicide followed by prescribed burning has also been attempted (fig. 6-4), though results are not reported in the literature.

Figure 6-4—Using a helitorch to ignite herbicide-killed Brazilian pepper in spring 2006. Herbicide was applied one year before the fire. The fire was not effective at consuming dead pepper trees due to standing water and low fuel loads. (Photo by Steve Smith.)

Published reports were not found that document attempts to manage chinaberry with fire in the Southeast bioregion. Chinaberry recovered fully from a single autumn fire in South America, reproducing vegetatively from both stumps and roots (Menvielle and Scopel 1999; Tourn and others 1999). A single surface fire killed all seeds in the seed bank, and fruit production was 90 percent less than in unburned control plots. Chinaberry seedling emergence following the fire was 5 to 20 times greater in unburned control plots than in burned plots; however, the seasonal pattern of seedling emergence and survivorship was not affected by fire (Menvielle and Scopel 1999). The South American studies suggest that a single fire is not effective in controlling chinaberry, with populations quickly returning to prefire levels or even expanding (Tourn and others 1999). Additional research is needed to determine if fire in different seasons, multiple-year fires, or a combination of fire and herbicide application are effective for controlling Chinaberry in the Southeast.

The use of fire alone is not likely to cause enough damage to kill Old World climbing fern plants and prevent postfire sprouting and rapid recovery in wet grasslands. This is due to the high moisture content of most wet grassland fuels resulting in low-severity fire (Stocker and others 1997). Spot burning prior to herbicide application can reduce the amount of herbicide needed to control Old World climbing fern by about 50 percent (Stocker and others, In press). See the "Pine and Pine Savanna Habitat" section for more information on the use of fire to control climbing ferns.

Autumn-olive (*Elaeagnus umbellata),* sericea lespedeza (*Lespedeza cuneata*), shrubby lespedeza (*L. bicolor*), and tall fescue commonly occur on disturbed sites near southeastern grasslands. While no specific studies are available that examine the relationship between these species and fire in Southeast grassland habitats, studies conducted in other areas suggest that prescribed fire has a limited potential for controlling these species under certain circumstances and in combination with other control methods. Autumn-olive may respond to fire damage by sprouting, but empirical information on the relationship of this species to fire and fire management is lacking (Munger 2003b, FEIS review). See FEIS reviews by Munger (2004) and Tesky (1992) and TNC reviews by Stevens (2002) and Morisawa (1999a) for more information on the use of fire for management of lespedeza species. Recent introductions of tall fescue can be controlled by spring burning, and combinations of prescribed burns and herbicide applications have "moderate to high potential for restoration" (Batcher 2004).

Several resource management organizations have suggested that fire can be used successfully to reduce south Florida silkreed (*Neyraudia reynaudiana*) populations prior to spraying regrowth with herbicide. They caution, however, that silkreed is a highly combustible fuel source and, because of that, a special burning permit may be required (Rasha 2005, review). Burning without follow-up herbicide or mechanical control is ineffective in controlling silkreed and may enhance its growth and spread (Guala 1990, TNC review).

Other species that can be found in drier or upland portions of the wet grassland habitat include Japanese honeysuckle (*Lonicera japonica*), bush honeysuckles (Amur honeysuckle (*L. maackii*), Morrow's honeysuckle (*L. morrowii*), and tatarian honeysuckle (*L. tatarica*), Japanese stiltgrass (*Microstegium vimineum*) and tropical soda apple (*Solanum viarum*) (table 6-1). Bush honeysuckles are typically top-killed by fire, and fire may kill seeds and seedlings. Adult plants probably survive by postfire sprouting from roots and/or root crowns. Studies conducted in the Southeast bioregion are not available; however, field work in the Northeast bioregion suggests that repeated prescribed fire may be useful in controlling bush honeysuckles (chapter 5; Munger 2005a, FEIS review). Fire research on Japanese stiltgrass in the Southeast bioregion is also needed; however, studies outside of the Southeast suggest that prescribed fire prior to seed set might aid in controlling this species (Howard 2005c, FEIS review). No published information is available on the relationship of tropical soda apple to fire.

Pine and Pine Savanna Habitat ____

Background

Pine and pine savanna habitats covered here include southern mixed forest, oak-hickory-pine forest, and subtropical pine forest associations as described by Küchler (1964). Pine and oak species are the dominant trees, including longleaf pine, shortleaf pine (*Pinus echinata*), loblolly pine (*P. taeda*), slash pine (*P. elliottii*), pond pine (*P. serotina*), southern red oak (*Quercus falcata*), turkey oak (*Q. laevis*), sand-post oak (*Q. margaretta*), bluejack oak (*Q. incana*), blackjack oak (*Q. marilandica*), post oak, and water oak (*Q. nigra*). Pond cypress and palms are the dominant trees in some wetter and more southern sites. Shrub species are common, including runner oaks (*Q. minima* and *Q. pumila*), sumac (*Rhus* spp.), ericaceous shrubs (for example, *Vaccinium*), palms, wax myrtle (*Myrica cerifera*), and hollies (*Ilex* spp.). Understory species include grasses such as wiregrass (*Aristida stricta* and *A. beyrichiana*), little bluestem (*Schizachyrium scoparium*), and numerous forbs. When Europeans first arrived in the Southeast, pine stands, and especially pine savannas, may well have been the dominant vegetation in most of this area, extending from southeastern

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Virginia to eastern Texas and from northern Georgia and Alabama to the Florida Keys (Platt 1999).

Presettlement fire regimes are poorly understood, but it is inferred that the high number of lightning strikes resulted in a fire-return interval of less than 13 years in pine forests and savannas. Larger and more intense fires probably occurred in May and June, after the start of the lightning/rain season but before large amounts of rain had fallen. Summer fires were probably more frequent but less intense and smaller in area. Ignitions by Native Americans probably increased fire frequency in many locations, shaping the savannas seen by early explorers (Wade and others 2000). Fire intervals may have been 1 to 4 years (1 to 5 years for subtropical pine forest; Myers 2000) before the arrival of European settlers, and then 1 to 3 years until fire exclusion became the norm in the early 1900s (Wade and others 2000).

Fires in these habitats were historically understory fires. Short return-interval (<10 years), understory fires predominated in most of the southern mixed forest and oak-hickory-pine types (*sensu* Küchler 1964). Slash pine and loblolly pine habitats experienced understory and mixed-severity fire regimes, with presettlement fire-return intervals estimated between 1 and 35 years (Myers 2000; Wade and others 2000).

The once-common southern pine forests were dramatically reduced by invasions of native hardwood species when fire exclusion policies were adopted in the 1920s and 1930s. Also affected were the populations of native plant and wildlife species, nutrient cycling, fuel reduction, and range management objectives that were associated with the historical fire regime. It has been suggested that savanna ecosystems that are not too seriously degraded can be restored if the appropriate fire regime (short return-interval, understory, spring and summer fires) is re-introduced, because the native plant species are adapted to this regime (Wade and others 2000).

Among the most threatening nonnative invasive plant species found in these habitats are cogongrass, Japanese honeysuckle, Brazilian pepper, and melaleuca. These species are fire-tolerant, thus reducing the effectiveness of fire in controlling their establishment and dispersal. Additionally, populations of climbing ferns appear to be increasing rapidly in Florida pine habitats and are spreading in Alabama, Florida, Georgia, and Mississippi pine habitats. Japanese climbing fern (*Lygodium japonicum*) has become particularly troublesome where pine straw is collected for sale as mulch. Spores of this fern have been found in straw bales, and the distribution of mulch bales throughout the Southeast has spread Japanese climbing fern into new areas. There is no specific information on fire and management of this species in this vegetation type. Old World climbing fern and melaleuca are also serious problems in wet grasslands and are discussed in the "Wet Grasslands Habitat" section.

Role of Fire and Fire Exclusion in Promoting Nonnative Plant Invasions in Pine and Pine Savanna

Frequent surface fires typical of presettlement fire regimes in most pine habitats promote some invasive species such as cogongrass. When fire is excluded, pine habitats are especially vulnerable to invasions by nonnative plants such as Brazilian pepper and Japanese honeysuckle.

Cogongrass and closely related Brazilian satintail (*Imperata brasiliensis*) are perennial, rhizomatous grasses that are well adapted to frequent fire. Both are early-seral species in wet-tropical and subtropical regions around the world. Discussion about management of these species is complicated by difficulty in distinguishing between the two species, lack of consensus among taxonomists about whether they actually are separate species, and determination of their native ranges (Howard 2005b, FEIS review). Among the more recent treatments, Wunderlin and Hansen (2003) describe them as distinct species, distinguished by anther number, and suggest that, while cogongrass is not native to the United States, Brazilian satintail is native to Florida.

Cogongrass requires some type of disturbance, such as fire, to maintain its dominance in southeastern pine understory. Frequent fire typically favors cogongrass over native species including big bluestem (*Andropogon gerardii*), Beyrick threeawn (*Aristida beyrichiana*), golden colicroot (*Aletris aurea*), and roundleaf thoroughroot (*Eupatorium rotundifolium*). Cogongrass flowering and seed production may be triggered by burning and other disturbances, although flowering has also been observed in undisturbed populations. In the absence of fire, vegetative growth from rhizomes (fig. 6-5) allows expansion of populations. Rhizomes also sprout easily after burning (Howard 2005b). In Mississippi wet pine savannas, cogongrass seedlings had higher levels of survival (for at least two months) in burned than in unburned study plots (King and Grace 2000).

Only limited information is available on two invasive shrub species. Brazilian pepper invades pine rockland (southern Florida habitat on limestone substrate) where fire has been excluded (Loope and Dunevitz 1981). It is suggested that Japanese honeysuckle is intolerant of frequent, low-severity fire and is typically absent from plant communities with this type of fire regime, such as longleaf pine. Therefore exclusion of fire from these communities may promote its establishment and spread (Munger 2002a, FEIS review).

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Figure 6-5—Cogongrass rhizomes. (Photo by Chris Evans, River to River CWMA. Bugwood.org.jpg.)

Old World climbing fern invasions provide a novel source of fuel in pine habitats (Roberts, D. 1996) and alter fire behavior by altering plant community fuel structure with extensive dry-standing frond "skirts" that ladder fire into the canopies of trees (Ferriter 2001) (fig. 6-7). Resulting canopy fires often kill trees that are adapted to low-severity surface fires, as well as native bromeliads (for example, wild pine (*Tillandsia fasciculata*)) resident on tree trunks. Fire spread may also be promoted when pieces of burning fern frond are kited into adjacent areas (Roberts, D. 1996). Increased fuel loads, altered fuel structure, and spotting from Old World climbing fern are blamed for tree mortality and escape of prescribed fires in pine stands at Jonathan Dickinson State Park in Florida. The park's fire management plan has been revised to no longer depend on wetland buffers to act as fire breaks if they contain climbing fern (Ferriter 2001).

Effects of Plant Invasions on Fuel and Fire Regime Characteristics in Pine and Pine Savanna

Nonnative species life-forms that have invaded southeastern pine habitats and altered fuel and fire regime characteristics include trees (melaleuca), shrubs (Brazilian pepper), grasses (cogongrass), and ferns (climbing ferns) (fig. 6-6). In some situations these species replace native species and fill similar forest strata, but in other cases the invasive plants completely alter the horizontal structure and fuel characteristics of the invaded plant community.

Figure 6-6—Old World climbing fern climbing and overtopping vegetation in a pine habitat at the Jonathan Dickinson State Park in central Florida. (Photo by Mandy Tu, The Nature Conservancy.)

Figure 6-7—Old World climbing fern on a slash pine, burning during a routine prescribed burn in pine flatwoods at the Reese Groves Property in Jupiter, Florida. Large slash pine and cypress have been killed by fire when climbing fern "ladders" carry fire into the canopy. (Photo by Amy Ferriter, South Florida Water Management District, Bugwood.org.)

Cogongrass invasion changes fuel properties in southeastern pine communities (Howard 2005b) (fig. 6-8). In a study of fuels and fire behavior in invaded and uninvaded pine stands in Florida, Lippincott (2000) found that invasion by cogongrass may lead to changes in fire behavior and fire effects in these communities. Native plant and cogongrass fuels have similar energy

content; however, fire behavior is driven by factors other than energy content of fuels. Sites invaded by cogongrass had greater fine-fuel loads, more horizontal continuity, and greater vertical distribution of fuels. The resulting fires were more horizontally continuous and had higher maximum temperatures and greater flame lengths than fires in adjacent plots not invaded by cogongrass, and they resulted in higher subsequent mortality to young longleaf pine (Lippencott 2000). Similarly, Platt and Gottschalk (2001) found fine fuel and litter biomass were higher in cogongrass and nonnative silkreed stands than in adjacent pine stands without these grasses. The authors suggest that increases in fine fuels attributed to cogongrass could increase fire intensity at heights of 3 to 7 feet (1 to 2 m) above the ground (Platt and Gottschalk 2001).

Bahia grass (*Paspalum notatum*) is not commonly found within existing pine stands but occupies heavily disturbed pine habitat that has been converted to pasture and rangeland, and interferes with efforts to restore pine communities. Bahia grass forms a continuous "sod fuel layer" in the previously patchy pine community, thus increasing fuel continuity (Violi 2000, TNC review). Because bahia grass is important forage for livestock, rangeland managers use controlled burns in winter to stimulate its growth. Winter burns negatively affect some native understory species, including wiregrass, which responds better to late summer and fall burns (Abrahamson 1984).

Melaleuca invasion in pine flatwoods can alter the fire regime from frequent (1- to 5-year return interval), low-severity surface fires to a mixed regime with less frequent (<35 to 200 year return interval) fires and greater incidence of crown fires. Crown fires are typically nonlethal to melaleuca trees but usually result in pine mortality. This combination of high-intensity fire and crown-fire survival is uncommon in North America (Myers 2000).

Low levels of fuel under mature Brazilian pepper (Doren and others 1991) and the difficulty of burning Brazilian pepper wood and leaves due to their high moisture content (Meyer 2005a, FEIS review) probably reduce fire intensity and fire spread in areas of dense infestation. Similarly, invasion by kudzu (*Pueraria montana* var. *lobata*) may reduce flammability of invaded pine habitats during the growing season due to its luxuriant, moist foliage. Conversely, the large amount of fuel biomass contributed by kudzu (fig. 6-9) and by plants killed by its invasion may increase the potential for dormant-season fires by increasing fuel loads, and its vining nature may increase the chance of fire crowning (Munger 2002b, FEIS review). These conjectures have not, however, been tested empirically.

Figure 6-8—(A) Infestation of cogongrass in a slash pine plantation in Charles M. Deaton Preserve, Mississippi. (Photo by John M. Randall, The Nature Conservancy.) (B) Cogongrass among planted pines in Mitchell County, Georgia, forms large accumulation of fine fuels around the base of trees. (Photo by Chris Evans, River to River CWMA, Bugwood.org.)

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Figure 6-9—Kudzu infestation at Travelers Rest, South Carolina in (A) summer, and (B) winter. Green kudzu foliage in summer may reduce potential for fire spread, while the opposite may be true in winter. (Photos by Randy Cyr, GREENTREE Technologies, Bugwood.org.)

Use of Fire to Manage Invasive Plants in Pine and Pine Savanna

Fire has become an important tool of natural area managers for removal of nonnative invasive species and maintenance of fire-adapted pine communities (Rhoades and others 2002). A study in a pine flatwoods community found that annual winter (non-growing season) burning increased native species richness and provided habitat for rare and listed plant species. Nonnative invasive species were found only in unburned plots. This may, however, be attributed to microsite differences within the treatment areas, as there was greater moisture availability in burned plots than in unburned plots (Beever and Beever 1993).

Old World climbing fern aerial fronds burn easily, and individual fronds can be completely consumed by a fire of sufficient intensity, but there are several reasons why prescribed fire is not expected to provide a major role in management of this species (Ferriter 2001). Pieces of burning climbing fern fronds get caught in fire-induced updrafts, reducing the ability to manage the fire perimeter when they are transported to adjacent areas. Climbing fern spores are very small and probably travel great distances by wind, including fire currents. Old World climbing fern plants in a south Florida slash pine stand were observed to sprout and recover rapidly after low-severity fire was applied using a hand-held propane torch (Stocker and others 1997).

Brazilian pepper is another species that is unlikely to be eliminated from pine stands by fire. Low-severity fire does not kill adult pepper trees, as girdling of the stem results in profuse sprouting from aboveground stems and root crowns (Woodall 1979). Brazilian pepper seeds can be killed by heat (70 °C for 1 hour; Nilsen and Muller 1980), and young seedlings can be killed by fire (Ferriter 1997). However, the intense crown fires necessary to kill adult plants (Doren and others 1991; Smith, C. 1985) do not commonly occur in pine stands with dense Brazilian pepper infestation. Fire does not carry well in mature Brazilian pepper stands, and fire rarely penetrates dense stands (Meyer 2005a). Brazilian pepper litter decomposes rapidly, leaving little litter for fuel, and moisture levels of branches, leaves, and litter are typically high (Doren and others 1991).

Prescribed fire may be more effective for controlling young Brazilian pepper stands. In areas where the water table lies below the soil surface for at least part of the year, grasses should provide sufficient fuels to carry fire of sufficient severity to kill young Brazilian pepper seedlings, and may also kill seeds (Nilsen and Muller 1980). Maintaining fire programs that killed seedlings prior to reaching unspecified "fire-resistant heights" has resulted in pepper-free areas (Ferriter 1997), and it has been noted that fire with a 5-year fire-return interval in Everglades National Park has excluded Brazilian pepper (Loope and Dunevitz 1981). On sites where either higher or lower water tables reduce the development of herbaceous fuels, prescribed fire may not be of sufficient severity to kill young Brazilian pepper plants (Ferriter 1997). In a study in south Florida pinelands, for example, most Brazilian pepper saplings over 3 feet (1 m) tall survived fire by coppicing (Loope and Dunevitz 1981). In any case, Brazilian pepper seed is readily dispersed from nearby stands by animals (Ewel and others 1982). The conclusion of a group of resource managers and scientists is that repeated burning may slow invasions of this species by killing seeds and seedlings, but fire "is not an effective control method for mature Brazilian peppertree stands" (Ferriter 1997).

It has been suggested that Japanese honeysuckle can be controlled by prescribed burning in pine plantations

or in fire-dependent natural communities. Prescribed burns in Virginia are recommended to reduce Japanese honeysuckle cover and to "inhibit spread" for 1 to 2 growing seasons (Williams 1994, Virginia Department of Conservation and Recreation review). Two annual fires in a pine-hardwood forest resulted in an 80 percent reduction in Japanese honeysuckle crown volume and a 35 percent reduction of ground coverage. While these treatments do not eliminate Japanese honeysuckle from the site, the authors suggest that they may reduce the amount of herbicide required in an integrated management program (Barden and Matthews 1980).

Fire by itself does not control cogongrass, and in fact frequent fire promotes cogongrass. Fire can, however, improve the success of an integrated management approach using tillage and herbicides. Fire is also important for maintaining native plant diversity in pine habitat, and restoration of native plant species may be a critical factor in longer term control of cogongrass (Howard 2005b).

Controlled burning has not been effective in killing kudzu, but it can be used to remove vines and leaves to permit inspection of root crowns for population and stand monitoring. Fire also promotes seed germination in kudzu, after which seedlings can be effectively controlled with herbicides. Spring burns are recommended to reduce soil erosion by winter rainfall (Moorhead and Johnson 2002, Bugwood Network review). When removed from a portion of its occupied area, kudzu can re-invade from water- and bird-disseminated seed (Brender 1961). Similarly, controlled burning has not been effective in managing bahia grass because it sprouts readily after fire (Violi 2000).

Oak-Hickory Woodland Habitat ____

Background

The distribution of oak-hickory woodlands in the Southeast bioregion has depended on historical fire management practices. Limited to the most mesic and protected sites during periods of shortened fire-return intervals, oak-hickory woodland habitats have increased in area during the fire-exclusion decades of the early 1900s and continue to occupy many parts of the Southeast region today (Daubenmire 1978).

Plant communities in this type are dominated by a variety of oaks and hickories, with a mixture of other tree species, including maple (*Acer* spp.), magnolia (*Magnolia* spp.), sassafras (*Sassafras* spp.), and ericaceous shrubs. Several vines commonly occur, including grape and greenbrier (*Smilax* spp.). Many pine species are found in areas with edaphic and/or fire disturbances (Daubenmire 1978).

Fire regimes in oak-hickory habitats are classified as understory types with return intervals estimated between 2 and 35 years. Presettlement fire regimes are poorly understood, although estimates based on dendrochronology indicate a fire-return interval of 7 to 14 years in the mid-Atlantic region. After European settlement, fire-return intervals were reduced to 2 to 10 years, with some sites burned annually. At the present time, the fire regime of oak-hickory forests is infrequent, low-severity surface fires occurring principally during spring and fall. They are mainly human-caused and only burn small areas (Wade and others 2000).

Oak-hickory woodland habitats in the Southeast are heavily invaded by aggressive nonnative vines, shrubs, and trees including kudzu, Japanese honeysuckle, privet (*Ligustrum* spp.), bush honeysuckles, and tree-of-heaven (*Ailanthus altissima*). Other invasive species are found in oak-hickory woodland habitats, but much less information is available for them. Mimosa (*Albizia julibrissin*) is a small tree found throughout the Southeast bioregion in many types of disturbed areas, including old fields, stream banks, and roadsides (Miller 2003). While it is a common species, little published information describes its relationship with fire. Giant reed (*Arundo donax*) is commonly found in riparian areas in much of the United States and has been reported as invasive in Georgia, Virginia, and Maryland (Swearingen 2005). Thorny-olive (*Elaeagnus pungens*) is found as an ornamental escape in the Southeast (Miller 2003), but there are no published reports on the relationship of this species to fire. Winged euonymus (*Euonymus alatus*) is reported in a variety of east coast habitats, including forests, coastal scrublands and prairies (USFWS 2004, review), but no information is available on the relationship of this species to fire. No information is available concerning a related species, winter creeper (*Euonymus fortunei*). It is found in many states throughout the East (Swearingen and others 2002), but no specific information identifies habitats where it commonly occurs.

Role of Fire and Fire Exclusion in Promoting Nonnative Plant Invasions in Oak-Hickory Woodland

Only limited information is available on the role of fire and fire exclusion as they affect invasive plant species in Southeast oak-hickory woodland habitat. In some cases, fire exclusion seems to promote establishment and spread of nonnatives, while in other cases the canopy gaps created by fire may increase the likelihood of establishment and spread of nonnatives. In more tropical parts of the Southeast, it may be assumed that increased light penetration into the plant

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community will promote establishment and spread of shade-intolerant nonnative invasive species.

A study of the consequences of hurricane damage in conservation lands of south Florida demonstrates the effects of canopy gaps on nonnative plant invasions. Air potato (*Dioscorea bulbifera*) and other nonnative vine species increased after the tree canopy was damaged by Hurricane Andrew (Maguire 1995). This species may respond in a similar fashion to canopy gaps created by fire. Similarly, fire is one of many types of disturbance that creates canopy gaps that improve the chances for establishment by tree-of-heaven in old-growth woodland. Additionally, tree-of-heaven seed germination is delayed and reduced by leaf litter and may therefore be enhanced by fire when litter is consumed. On the other hand, fire may produce a flush of herbaceous growth that could inhibit tree-of-heaven germination (Howard 2004a, FEIS review). More information is needed on the effects of fire on seed germination in this species.

Soil heating may promote kudzu establishment by scarifying kudzu seedcoats and stimulating germination (Miller 1988). Similarly, mimosa seeds exposed to fire for 1 to 3 seconds had higher germination rates than unheated seeds (Gogue and Emino 1979). It has also been suggested that fire exclusion may promote Japanese honeysuckle spread (Munger 2002a).

Effects of Plant Invasions on Fuel and Fire Regime Characteristics in Oak-Hickory Woodland

The role of invasive plants in altering fire regimes in the Southeast bioregion is complicated by the existing mosaic of fire exclusion and controlled burning. Tree-of-heaven, for instance, is found in many types of woodlands in North America where presettlement fire regimes have been disrupted in many different ways. This makes it difficult to make definitive statements about the potential effect of tree-of-heaven on more natural fire regimes. The large amount of litter produced by tree-of-heaven from large leaves and broken branches, and its tendency to form dense thickets, may contribute to fire spread and crown fires in invaded areas (Howard 2004a).

A FEIS review speculates that the abundant moist foliage of kudzu could inhibit fire, effectively lengthening the time between fires in woodland habitats. On the other hand, the large amount of kudzu biomass may increase the potential for dormant-season fires by increasing fuel loads, and its vining nature may increase the chance of fire crowning. Additionally, increases in standing and surface fuels formed by plants killed following kudzu invasion may increase both fire intensity and frequency. The author points out that

studies needed to test these hypothetical statements have not been conducted (Munger 2002b).

Use of Fire to Manage Invasive Plants in Oak-Hickory Woodland

Fire has not been recommended as a sole management tool to control tree-of-heaven because of this species' potential to burn in crown fire, its ability to sprout from the root crown and/or roots following top-kill from fire, and the potential for fire to promote seed germination. Fire has been used to reduce aboveground biomass of tree-of-heaven (Howard 2004a). A flame-thrower or weed burning device has been suggested to kill lower limbs (Hoshovsky 1988, TNC review), but this is not a population reduction measure.

Fire has been used to reduce cover of Japanese honeysuckle but does not kill plants. Japanese honeysuckle sprouts from subterranean buds, roots, and stems, recovering to various levels after fire (Munger 2002a). Japanese honeysuckle remained a site dominant after two consecutive annual fires in a pine-hardwood forest in North Carolina (Barden and Matthews 1980). Experimental plot (abandoned agricultural field) burns 5 years apart near Nacogdoches, Texas, resulted in Japanese honeysuckle plants with fewer and shorter prostrate shoots than in unburned plots 1 year after the last burn, but plants were not killed (Stransky 1984). Because prostrate shoots are an important part of this species' ability to invade native plant communities (Larson 2000), reduction in numbers of these shoots could theoretically slow the invasion process.

Seasonality of burns can affect postfire response of Japanese honeysuckle. Prescribed burns in October in a Tennessee oak-hickory-pine forest with a maple and dogwood (*Cornus* sp.) understory reduced Japanese honeysuckle coverage by 93 percent; burns in January or March reduced Japanese honeysuckle by 59 percent. Vegetation measurements were taken at the end of the growing season (September) about 1.5 years after burning (Faulkner and others 1989). The Nature Conservancy recommends fall, winter, or early spring prescribed burning to control Japanese honeysuckle in northern states, when Japanese honeysuckle maintains some leaves and most native plants are leafless (Nuzzo 1997, TNC review). This improved ability to target a particular species may have some applicability in southeastern habitats, but more often other native species retain leaves through the winter and may therefore be more subject to damage by fire at those times.

The Nature Conservancy also suggests that integrating fire and herbicide treatments to control Japanese honeysuckle may be more effective than either approach alone, with herbicides applied about a month after

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sprouting occurs following a late fall or winter burn (Nuzzo 1997). Application of herbicide about 1 year after a burn was not effective, possibly because postfire increases in herbaceous vegetation resulted in less herbicide contacting Japanese honeysuckle (Faulkner and others 1989). Fire is also helpful in controlling fire-intolerant Japanese honeysuckle seedlings and young plants. Efforts should be made to avoid soil disturbance as much as possible to reduce subsequent germination of Japanese honeysuckle seeds in the seed bank (Nuzzo 1997).

Prescribed fires have been suggested for controlling bush honeysuckles (Tatarian honeysuckle, Morrow's honeysuckle, Bell's honeysuckle (*Lonicera X bella*), and Amur honeysuckle) in fire-adapted communities (Nyboer 1990, Illinois Nature Preserves Commission review). Spring burns kill bush honeysuckle seedlings and top-kill mature plants; however, plants sprout readily after fire. Effective control may come from annual or biennial fires conducted for 5 years or more (Nyboer 1990).

It has been suggested that Chinese privet is intolerant of fire (Matlack 2002) and can be managed successfully by repeated fire, especially on sites with low stem density and high fine fuel loads (Batcher 2000a, TNC review). A single fire does not result in sufficient kill of mature plants (Faulkner and others 1989) but instead promotes sprouts from root crowns and/or roots (Munger 2003c, FEIS review). Chinese privet burns poorly without additional fuel. However, if sufficient low-moisture fuels are available (Batcher 2000a), annual fires may substantially reduce or kill aboveground portions of Chinese privet, although they will not eliminate it from a site. Three annual prescribed burns did not eradicate Chinese privet from areas where fire had been excluded for more than 45 years (Munger 2003c). Platt and Stanton (2003) suggest that dominance by Chinese privet cannot be reversed, but increases in population size can be prevented with short return interval, lightning-season fires. Japanese privet (*Ligustrum japonicum*) and European privet (*L. vulgare*) also occur in this vegetation type, but no specific information on management and fire for these species in this vegetation type is available.

Prescribed burns have been suggested as part of a strategy to manage kudzu. Information on the limitations of prescribed fire and effects of kudzu removal on native vegetation (presented in the "Pine and Pine Savanna Habitat" section page 100) is relevant to this habitat as well.

Air potato (*Dioscorea bulbifera*) invades woodland habitats throughout Florida (Schmitz and others 1997). While only a limited amount of research has been conducted, prescribed fire may be useful in killing stem growth (Morisawa 1999b, TNC review) and bulbils (Schultz 1993, TNC review) of air potato in woodlands.

A related species, Chinese yam (*Dioscorea oppositifolia*), has been reviewed by The Natural Conservancy (Tu 2002b). Chinese yam is found in mesic bottomland forests, along streambanks and drainageways in many states of the Southeast. Only very limited information is available about the use of fire to manage this species. It was noted that reduced amounts of Chinese yam were present the year following a fall wildfire in Great Smoky Mountains National Park (Tu 2002b), but the specific habitat information is not available.

Tropical Hardwood Forest Habitat

Background

Scattered throughout the grassland and savanna plant communities of south Florida are "islands" of tropical hardwood species, often called hammocks, and typically found on somewhat drier sites. Common species include gumbo limbo (*Bursera simaruba*), black ironwood (*Krugiodendron ferreum*), inkwood (*Exothea paniculata*), lancewood (*Ocotea coriacea*), marlberry (*Ardisia escallonoides*), pigeon plum (*Coccoloba diversifolia*), satinleaf (*Chrysophyllum oliviforme*), poisonwood (*Metopium toxiferum*), and white stopper (*Eugenia axillaris*). While limited in extent compared to the other habitats discussed in this chapter, they are important because of the plant diversity they provide within the grassland landscape.

Fire regime in this habitat varies from low-intensity surface fires to crown fires, with an estimated pre-settlement return interval of 35 to over 200 years. Fire has not been the dominant force in shaping hardwood hammock plant communities because they are usually difficult to burn. Although many of the hardwood species sprout following top-kill from fire, the stands can be destroyed by fire during periods of drought if the organic soil is consumed (Myers 2000).

Many hardwood hammocks are being aggressively invaded by air potato, melaleuca (fig. 6-10), and Old World climbing fern. Water yam (*Dioscorea alata*) has been reported in coastal hammocks in Florida (FLEPPC 1996). This species is related to air potato and may act in a similar manner to air potato in relation to fire, although no specific information is available for either species.

Role of Fire and Fire Exclusion in Promoting Nonnative Plant Invasions in Tropical Hardwood Forest

Limited information is available regarding the role of fire in promoting nonnative plant invasions in tropical hardwood hammocks. Melaleuca is capable

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Figure 6-10—Melaleuca saplings (green trees) ringing a Cypress clump at the Loxahatchee National Wildlife Refuge around 1994. The melaleuca eventually overtook the cypress (Ferriter, personal communication 2007). (Photo by Amy Ferriter, South Florida Water Management District, Bugwood.org.)

of spreading quickly into this habitat following fires that remove most of the vegetation and expose bare mineral soil (Bodle and Van 1999).

Old World climbing fern is rapidly invading hardwood hammocks, but it is not known to what extent invasion is dependent on or retarded by fire. Anecdotal evidence suggests that the fern appears to grow especially well in hammocks where native vegetation was either damaged or killed by fire during a drought. Researchers and natural area managers in southern Florida suspect that Old World climbing fern is particularly robust in areas where native tree island vegetation was damaged or killed by a fire during the drought of 1989–1990. It has also been suggested, although not demonstrated, that convection currents generated by burning fern growth that has formed a trellis up into tall trees could increase the dispersal of spores (Ferriter 2001). Old World climbing fern plants sprout and recover rapidly after low-severity fire (Stocker and others 1997).

The influence of melaleuca and Old-World climbing fern on fuels and fire regimes has been discussed in previous parts of this chapter, as has the use of fire for managing these species. Fire by itself is not an effective method to manage melaleuca or Old World climbing fern, although suggestions have been made for ways in which fire could be incorporated into an integrated management approach with other control techniques. Since native species in this habitat probably did not evolve in a regime of frequent fire, increasing fire frequency with prescribed burning might have unintended effects on native plant species (Ferriter 2001).

Cypress Swamp Habitat _____

Very limited information is available about the relationship between fire and invasive species in cypress swamp habitat. Depressional wetlands in central and south Florida dominated by bald cypress (Küchler's (1964) Southern Floodplain Forest) had a presettlement stand-replacement fire-return interval estimated at 100 to 200 years or greater (Wade and others 2000). This fire regime has been altered by invasions of melaleuca and Old World climbing fern. Both species increase the probability of more frequent stand-replacement fires because of the ease with which they burn and because Old World climbing fern can form a fuel bridge between adjacent, more frequently burned habitats and the much wetter cypress swamp

(Langeland 2006). Information on the role of these species in relation to fire is also presented in the "Wet Grassland Habitat," "Pine and Pine Savanna Habitat" and "Oak-Hickory Woodland Habitat" sections.

Catclaw mimosa (*Mimosa pigra*) occurs on about 1,000 acres (400 ha) in Florida, including cypress swamp habitat. While fire has been used experimentally in Australia to clear mature plants, enhance seed germination for subsequent herbicide application, and kill some seeds (Lonsdale and Miller 1993), no similar studies have been conducted in the United States.

Conclusions and Summary _____

The importance of fire and fire management in influencing species composition and dynamics of plant communities in the Southeast bioregion is often stated, if not completely understood. Incomplete or contradictory information, for instance, describes the plant communities prior to human influence (Stanturf and others 2002). With its high number of lightning strikes, its many fire-adapted communities, and its historical human dependence on fire-maintained habitats, fire has probably been a more important factor in the Southeast than in any other broadly defined region of the country. Plant communities such as longleaf pine once covered millions of acres when human populations supplemented naturally occurring fires with intentional blazes (Chapman 1932). Even wet grasslands in the Southeast tend to be flammable and are adapted to frequent fires (Leenhouts 1982; Schmalzer and others 1991; Wade 1988; Wade and others 1980).

Human influence on fire regime was accompanied by a large number of intentional introductions of plants for agricultural, horticultural, medicinal, and religious purposes, as well as many accidental imports. The large number of intentional introductions and the escape of these introduced plants led to very high levels of invasive plants in the Southeast bioregion, especially Florida. More than 25,000 species and cultivars have been introduced to Florida (D. Hall, personal communication, cited by Gordon 1998), a state with around 2,523 native species (Ward 1990). While many of these introductions have served their intended purposes, a small proportion (about 10 percent (Gordon 1998)) has caused unintended damage to forest and conservation lands. It is not clear whether the large number of nonnative plant invasions in this bioregion is due to the large number of introductions (in other words, propagule pressure), or whether habitats in the Southeast are more susceptible to invasion. The high number of plant species introductions in the Southeast may be related to the diversity of cultural origins of the human populations and a range of climates from tropical to temperate. This latter factor could also influence the susceptibility of the region by providing a wider range of potentially suitable conditions for establishment and spread.

The causes and effects of invasive plant establishment and spread are related to fire and fire management to varying degrees. The role of fire and fire exclusion in promoting nonnative plant invasions is very different among the five habitats discussed in this chapter. Fire exclusion policy, of course, does not mean the absence of fire. Historical and current efforts to exclude fire have often led to less frequent but more severe fires, which then can lead to substantial changes in native and invasive species populations (Wade and others 2000).

Role of Fire and Fire Exclusion in Promoting Nonnative Plant Invasions

The complex and dramatic relationship between invasive species and both fire and fire exclusion in the Southeast bioregion may be best exemplified by melaleuca invasion in pine, wet grassland, and tropical hardwood habitats in south Florida. Melaleuca has spread into thousands of acres apparently without need of fire, but has spread most dramatically following natural fire, controlled burns, and uncontrolled fire following decades of fire exclusion efforts.

Wet grasslands may be the habitat type most affected by fire and fire exclusion. Expansion of melaleuca following fire has been more thoroughly reported for wet grassland habitats (Ferriter 1999). Fire in wet grasslands is also possibly responsible for increases in Old World climbing fern (Langeland 2006) and chinaberry (Menvielle and Scopel 1999; Tourn and others 1999), although the evidence is principally anecdotal. Successful exclusion of fire has been blamed for Brazilian pepper invasions of salt marshes (Schmalzer 1995) and for Chinese tallow (Bruce and others 1995) and tall fescue invasions (Eidson 1997) in wet grassland habitat.

Fire exclusion has clear ecological impacts in pine habitats, which frequently accumulate additional woody shrub species in the absence of fire and ultimately become hardwood dominated habitats with a minor pine component (DeCoster and others 1999; Heyward 1939; Platt 1999; Slocum and others 2003; Streng and others 1993; Walker and Peet 1983). Less has been studied, however, about these longer-term changes and the interplay of fire and invasives. Brazilian pepper (Loope and Dunevitz 1981) and Japanese honeysuckle (Munger 2002a) are among the woody shrubs that are promoted in pine habitats when fire is excluded. Conversely, cogongrass in pine habitats is promoted by frequent surface fires and, in fact, requires some regular disturbance to maintain its dominance (Howard 2005b).

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Effects of Plant Invasions on Fuel and Fire Regime Characteristics

Some of the most obvious visual changes in fire occur when melaleuca invasions fuel crown fires in wet grassland communities. But while these conflagrations are dramatic, the removal of aboveground plant material is not a substantial change to the wet grassland fire regime. It is the secondary effects of these fires, such as consumption of the surface of organic soils followed by flooding, that can lead to major changes in plant communities (Wade and others 2000).

Some evidence suggests that when species such as Chinese privet, Chinese tallow, and Brazilian pepper replace native plant species in wet grasslands, fine fuel loads and horizontal continuity are reduced (for example, Doren and Whiteaker 1990; Doren and others 1991; Grace 1999; Platt and Stanton 2003). Change in fine fuels may lead to reduced fire frequency and intensity, and increased fire patchiness. There is, however, little experimental evidence to support these suggestions.

The most substantial changes in fire regime caused by nonnative plant invasion are probably in pine habitats. Invasions of Old World climbing fern, and possibly its congener Japanese climbing fern, increase incidence of crown fires, carry fire across wetland barriers that would have stopped the fire if they had not contained Old World climbing fern, and possibly "kite" fire to new locations (Langeland 2006). Cogongrass changes fire behavior and effects in pine habitats. Cogongrass invasions lead to increased biomass, horizontal continuity, and vertical distribution of fine fuels when compared with uninvaded pine savanna, and higher maximum fire temperatures have been reported (Lippincott 2000). In this particular example, the detailed studies necessary to show actual replacement of native species have not been conducted. Melaleuca changes the fire regime in pine stands from frequent, low-severity surface fires to a mixed fire regime with less frequent fires and greater incidence of crown fires. These crown fires are often lethal to pines but not to melaleuca (Myers 2000). It is possible that Brazilian pepper invasions have opposite effects and reduce fire intensity and fire spread where it is densely distributed (Doren and Whiteaker 1990; Doren and others 1991).

Use of Fire to Manage Invasive Plants

Among the habitats covered in this section, pine and pine savanna habitats are probably the most conducive to use of fire to manage invasive plants, in large part because of the role of fire in maintaining these pyric communities. Controlled burns in oak-hickory woodland have resulted in reductions of Japanese honeysuckle (Barden and Matthews 1980; Williams 1994) and reduced spread of Chinese privet (Batcher 2000a). Since frequent prescribed fire in oak-hickory woodland will favor pine species at the expense of young hardwoods, it will probably be more difficult to maintain as aggressive a burning practice in oak-hickory woodland than in pine and pine savanna.

In the presence of propagule sources from invasive species such as melaleuca and Old World climbing fern, prescribed fire in wet grasslands must be conducted with extreme care and may result in expansion of these species. The margin between use of fire for successful reduction of melaleuca seedlings following a seed-release event and accidental expansion of melaleuca into fire-cleared seed beds is very small and is affected by weather and water management patterns out of the manager's control (Belles and others 1999).

Additional Research Needs

Among the general information needs related to fire and invasive plants, several specific needs stand out. While we have case studies on short-term effects of various fire related practices for individual species, we don't know which fire management practices in which habitats will provide the most effective means of reducing existing invasive plant populations or preventing future invasions. This is not an easy area of research, in part because we do not know which of the tens of thousands of novel species that could be introduced to southeastern habitats will become management problems, making it nearly impossible to know what practices will provide the most future benefit. There are many nonnative species already present in Southeast bioregion habitats for which no information is available. Included in this category are giant reed, field bindweed (*Convolvulus arvensis*), Chinese silvergrass (*Miscanthus sinensis*), princesstree (*Paulownia tomentosa*), golden bamboo (*Phyllostachys aurea*), multiflora rose (*Rosa multiflora*), sericea lespedeza, five-stamen tamarisk (*Tamarix chinensis*), French tamarisk (*T. gallica*), smallflower tamarisk (*T. parviflora*), saltceder (*T. ramosissima*), bigleaf periwinkle (*Vinca major*), common periwinkle (*V. minor*), Japanese wisteria (*Wisteria floribunda*), and Chinese wisteria (*Wisteria sinensis*).

At a minimum we need longer-term studies that document broad species changes in population size and distribution. For instance, it has been suggested that replacement of wet grassland species by invasive hardwood shrubs (Brazilian pepper) and trees (Chinese tallow) results in reduced fine fuel load and horizontal continuity (Doren and Whiteaker 1990; Doren and others 1991; Grace 1999). These fuel changes could logically lead to changes in fire frequency, severity, and patchiness, but that has yet to be documented. There is also potential for subsequent changes in native plant species coverage and/or diversity—a topic that deserves further study.

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Because so much of the Southeast is relatively low-lying, the effect of natural and artificially-manipulated hydrology on the relationship between fire and invasive plants needs to be examined. For instance, we know that artificially lowered surface-water elevations in south Florida wet grasslands lead to increased fires and exposure of mineral soil that facilitate melaleuca invasions, but management practices that could prevent such invasions have not been determined.

Emerging Issues

The Southeast has more fire and more invasive plants than most other parts of the country. To further complicate the situation, the Southeast is also rapidly increasing its human population. The "sunbelt" is the fastest growing part of the United States, with a 21 percent increase in population between 1970 and 1980, and an 18 percent increase between 1980 and 1997 (NPA 1999). It remains to be seen whether prescribed fire practices can be implemented and maintained with more and more urban incursions into forest and natural areas.

Global warming and related increases in carbon dioxide may well affect the relationships among invasive plants, native plant communities, and fire. Lightning frequency is expected to increase with global warming. The U.S. Global Change Research Program's National Assessment Synthesis Team has predicted that the "seasonal severity of fire hazard" will increase about 10 percent for much of the United States, but a 30 percent increase in fire hazard is predicted for the Southeast (NAST 2000).

Major changes in plant communities have occurred in the Southeast because of the interaction of invasive nonnative plants and fire management policy and practice. However, it is not a given that the Southeast would have avoided its serious invasive plant problems if presettlement fire regimes had been maintained, nor that reinstating presettlement fire regimes in the 1800s would have prevented problems during the next two centuries.

Better understanding of the relationship between invasive plants, fire, and fire management is certainly needed if resource managers are to maximize their ability to prevent further nonnative plant invasions and adjust fire policy to both reduce the existing invasive plant populations and achieve other management objectives.

Notes