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Chapter 5:

Fire and Nonnative Invasive Plants in the Northeast Bioregion

Introduction

The Northeast bioregion extends from Maine to Maryland and northern Virginia, south along the northwest slope of the Appalachians to Tennessee, and west to the ecotone between prairie and woodland from Minnesota to northeastern Oklahoma. It is composed of a wide variety of landforms and vegetation types. Elevation ranges from sea level along the Atlantic coast, to 243 to 600 feet (74 to 183 m) at the Great Lakes, to over 6,000 feet (1,800 m) in the northern Appalachian Mountains.

Much of the native vegetation in the Northeast bioregion was historically closed-canopy coniferous and deciduous forest. Coniferous forests are characterized by spruce (*Picea* spp.) and balsam fir (*Abies balsamea*) in the north and by eastern white, red, pitch, and jack pines (*Pinus strobus*, *P. resinosa*, *P. rigida*, *P. banksiana*) in the northeastern coastal and Great Lakes areas. Deciduous forests in the northern part of the bioregion include those dominated by maple (*Acer* spp.), American beech (*Fagus grandifolia*), and birch (*Betula* spp.), others dominated by aspen (*Populus* spp.) and birch, others dominated by oak (*Quercus* spp.), and lowland and riparian forests dominated by elm (*Ulmus* spp.), ash (*Fraxinus* spp.), and cottonwood (*Populus* spp.). In the central and southern portions of the bioregion, oak and hickory (*Carya* spp.) are codominant species. Scattered stands of mixed oak and pine become more common toward the transition zone between oak-hickory and southern pine forests. Oak savannas, barrens, and tallgrass prairie remnants occur in the transitional area between eastern deciduous forests and central prairie and in other isolated locations. Early successional grasslands and woodlands occur in scattered areas and on abandoned farm land (old fields) (Garrison and others 1977; Smith and others 2001). Stands of eastern white pine often occupy former agricultural fields.

Fire History in the Northeast Bioregion

Information on fire history and fire regimes in the Northeast bioregion is given here and within sections on general plant communities to provide a context in which to discuss relationships between fire and

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nonnative invasive plant species. This information is derived largely from literature reviews, such as those in Brown and Smith (2000), except where otherwise indicated.

In northeastern plant communities, natural stand-replacing disturbances are more often caused by hurricanes, catastrophic wind events (Dey 2002a), and ice storms (for example, see Fahey and Reiners 1981) than by fire. However, fire has played a role in shaping the structure and composition of the vegetation in many areas. Fire has been a recurring disturbance in parts of the Northeast bioregion both before and after European settlement (Cronon 1983; Patterson and Sassaman 1988). At the landscape scale, there was substantial heterogeneity in fire regimes, and the relative evolutionary importance of fire varies among plant communities (Wade and others 2000). While estimates of presettlement fire regimes are difficult to confirm (Clark and Royall 1996), recent attempts have been made for vegetation types in the Northeast bioregion as part of the nationwide LANDFIRE Rapid Assessment (2005c). Evidence indicates that some northeastern communities burned more regularly than others (Parshall and Foster 2002). In general, fire regimes varied from almost no fires in beech-maple forests; to infrequent, high-severity, stand-replacement fires in northern coniferous forests dominated by spruce and fir; to frequent surface fires in oak-hickory forests, savannas, barrens, and prairie remnants (Wade and others 2000).

Lightning-caused fires are rare in the Northeast bioregion (Ruffner and Abrams 1998). People have been and continue to be the primary source of ignition (Leete 1938; Wade and others 2000). Before European contact, Native Americans used fire to manage landscapes for hunting, gathering, agriculture, and travel (Cronon 1983; Day 1953; Delcourt and Delcourt 1997; Dey 2002a; Pyne 1982a), and fire frequency was correlated with Native American occupancy. Fires continued to occur after European settlement and were ignited purposely and accidentally by both settlers and natives. The frequency and extent varied spatially and temporally based on factors such as topography, fuel loads, population levels, land use and fragmentation, and cultural values (for example, see Dey and Guyette 2000; Guyette and others 2002, 2003).

Fire exclusion efforts in the 20th century reduced fire frequency and extent in many fire-adapted plant communities (Shumway and others 2001; Sutherland 1997). On the New England sandplains, for example, several large fires occurred in the early 1900s but relatively few fires have occurred since, and these have been of smaller extent, due in part to suppression activities (Motzkin and others 1996). As forest succession and fire exclusion have proceeded, early-successional habitats have been reduced, and oak-dominated forests are gradually being replaced by forest dominated by a mix of maples and beech (review by Artman and others 2005). In an old-growth forest in western Maryland, for example, the overstory is currently dominated by oaks, but the recruitment layer has shifted from oaks to maple and birch; this shift corresponds with a lack of major fires since 1930 (Shumway and others 2001).

Since the 1980s, the Northeast bioregion has seen increasing use of fire as a management tool. In the last 10 years, prescribed fire, alone or in combination with silvicultural treatments, has been advocated to restore presettlement fire regimes or reference conditions in the Northeast bioregion, particularly in savannas and oak-dominated forests (Brose and others 2001; Healy and McShea 2002; Lorimer 1993; Van Lear and Watt 1993). Today prescribed burns are used routinely on public lands and lands managed by The Nature Conservancy for hazard fuel reduction, maintenance of fire-adapted ecosystems, promotion of oak regeneration, restoration of savannas, retention of early successional vegetation for breeding birds (reviews by, Artman and others 2005; Mitchell and Malecki 2003; Vickery and others 2005), and protection of rare plants (for example, see Arabas 2000; Patterson and others 2005; Trammell and others 2004). Burning may be a useful tool to aid in American chestnut (*Castanea dentata*) recovery in eastern oak forests (McCament and McCarthy 2005).

The actual use of prescribed fire, however, has been limited, and the spatial extent of burning has been relatively small. Nearly 70 percent of forest land in the region is owned by non-industrial private landowners (Smith and others 2001) who seldom use prescribed fire (Artman and others 2005). However, several states (for example, Ohio, Virginia, and North Carolina) have initiated programs to certify public land managers and private citizens in the use of prescribed burns. Thus prescribed burning may be used more frequently on private lands in the future (Artman and others 2005).

Nonnative Plants in the Northeast Bioregion

Current and presettlement vegetation types may have little in common in the Northeast bioregion because most forests were harvested or cleared for agriculture by the early 20th century. European settlers vastly increased the amount of open grassland and introduced many species of nonnative grasses, forbs, and shrubs to the bioregion. Many nonnative plants were introduced as contaminants in crop seed or other imported products, while others were introduced intentionally for agricultural and horticultural purposes. Most plant community types in this bioregion are invaded by nonnative plants in some areas, and

the spread of these species is an increasing problem today (Mehrhoff and others 2003; Richburg and others 2001).

Most large infestations of nonnative species occur in or near settled areas, agricultural lands, roads and trails, or on public lands where they were deliberately introduced (for example, see Barton and others 2004; Ebinger and McClain 1996). "Conservation plantings" previously advocated by federal agencies (Knopf and others 1988) included invasive species such as Japanese barberry (*Berberis thunbergii*), multiflora rose (*Rosa multiflora*), autumn-olive (*Elaeagnus umbellata*), bush honeysuckles (*Lonicera* spp.), and buckthorns (glossy, *Frangula alnus*, and common, *Rhamnus cathartica*). Relatively fewer infestations occur in remote, upland natural areas; however, spread of several nonnative species into more remote areas is facilitated by ongoing development, propagule dispersal along roads, rivers, and other corridors (Barton and others 2004; Buckley and others 2003; Lundgren and others 2004), and especially seed dispersal by birds (for example, see White and Stiles 1992). Many invasive shrubs and vines in the Northeast have bird-dispersed seed (Mack 1996) (fig. 5-1). Nonnative plant species recorded from traps, feces, feeding observations or stomach contents of birds in a study in New Jersey include Japanese barberry, multiflora rose, Oriental bittersweet (*Celastrus orbiculatus*), winged euonymus (*Euonymus alatus*), common buckthorn, European privet (*Ligustrum vulgare*), Japanese honeysuckle (*Lonicera japonica*), Amur honeysuckle (*L. maackii*), and Tatarian honeysuckle (*L. tatarica*) (White and Stiles 1992). Additionally, efforts to assist native wild turkey recovery in the Northeast bioregion include planting nonnative honeysuckles (*Lonicera* spp.) and Oriental bittersweet, seeds of which are subsequently dispersed by wild turkeys (Poole, personal communication 2005).

Figure 5-1—Glossy buckthorn fruits are bird-dispersed and sometimes taken deep into the shady forest (Bradley, Maine). (Photo by Alison C. Dibble.)

Interactions of Fire and Invasive Plants in the Northeast Bioregion

Managers in the Northeast share with other bioregions a concern about the interactions of fire and invasive species. With increasing use of prescribed fire for a variety of management objectives, managers need information on the effects of fire on nonnative plants present in areas to be burned, and on the potential establishment and spread of those plants in the postfire environment.

Of particular concern in the northeast are the effects of nonnative plants on fuel characteristics. Changes in fire regimes due to the presence of nonnative invasive plants in the Northeast bioregion were discussed by Richburg and others (2001), but this topic has otherwise received little attention in the scientific literature, and data are insufficient for making generalizations. Observations and data from other bioregions indicate that changes in fuel characteristics brought about by nonnative species invasions can lead to changes in fire behavior and alter fire regime characteristics such as frequency, intensity, extent, type, and seasonality of fire, and thus impact native plant and animal communities (chapter 3). Invaded forest communities in the Northeast studied by Dibble and others (2003) and Dibble and Rees (2005) often had substantially higher cover of shrubs than uninvaded communities, resulting in increased height and density of surface fuels and suggesting an increased potential for fire to carry into the tree canopy. Additionally, the authors found higher percent cover of nonnative grasses on several invaded forest sites. If nonnative grasses differ in fuel loading, spatial distribution, phenology, or other characteristics from desired native understory species, they may affect fire frequency and seasonal burning window (Dibble and others 2003; Dibble and Rees 2005). Heat content, measured in the cone calorimeter for 42 plant species, differed between some native and nonnative invasive plants found in the Northeast bioregion, with no trend exclusive to one or the other group. For example, plants of fire-adapted ecosystems including black huckleberry (*Gaylussacia baccata*), pitch pine, bear oak (*Quercus ilicifolia*), barberry (*Berberis* spp.), and reindeer lichen (*Cladonia* spp.) had especially high heat content while nonnatives black locust (*Robinia pseudoacacia*), Norway maple (*Acer platanoides*), Japanese stiltgrass (*Microstegium vimineum*), sheep sorrel (*Rumex acetosella*), and glossy buckthorn (*Frangula alnus*) had low heat content (Dibble and others 2007).

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There is growing interest in use of prescribed burning to control nonnative invasive plants (Bennett and others 2003); however, little is known about the effects of fire on nonnative invasives in this bioregion. Additionally, the use of prescribed fire in the Northeast is constrained by a highly reticulated wildland urban interface (WUI) in which the human population is high, and habitat fragmentation and new development are proceeding at a rapid pace. Prescribed burning in the Northeast is also difficult because it can be too moist and cool in most years for prescribed fires to carry or be effective. Additionally, policy restricts use of prescribed fire to particular seasons in many areas. For example, on Nantucket Island, Massachusetts, prescribed fires can only be conducted during the dormant season between October and April (personal communication cited in Vickery and others 2005), despite the fact that growing-season burns are probably more effective for controlling shrubs (Richburg 2005; Rudnicky and others 1997). Use of fire for controlling invasive plants is most effective when combined with other control methods (Bennett and others 2003).

In this chapter, we review the available literature on the interactions between fire and nonnative invasive plants in seven broad vegetation types in the Northeast bioregion: deciduous forest, coniferous forest, mixed forest, grasslands and early successional old fields, fresh wetland, tidal wetland, and riparian zone (fig. 5-2). Established vegetation classifications were not used because the limited data and literature available on fire and invasives in the Northeast bioregion makes using more specific classifications unrealistic. A brief description of the vegetation, presettlement fire regimes, and management issues is presented for each vegetation type. Consult Wade and others (2000) and Duchesne and Hawkes (2000) for greater detail about presettlement fire regimes and fire management considerations in the absence of nonnative invasive plants. The role of fire and/or fire exclusion in promoting nonnative plant invasions, fire regime changes brought about by nonnative plant invasions, and use of fire to control nonnative invasive species is discussed for each vegetation type. The focus is on nonnative species of concern for which some information is available regarding their relationship to fire or their response to other disturbances (table 5-1). This is only a subset of problematic nonnative species in the Northeast bioregion; many nonnatives of concern were excluded from this discussion due to lack of information. Interactions of fire and invasive plants can vary by species, vegetation type, and location, so the information presented in this chapter must be adapted for site-specific applications.
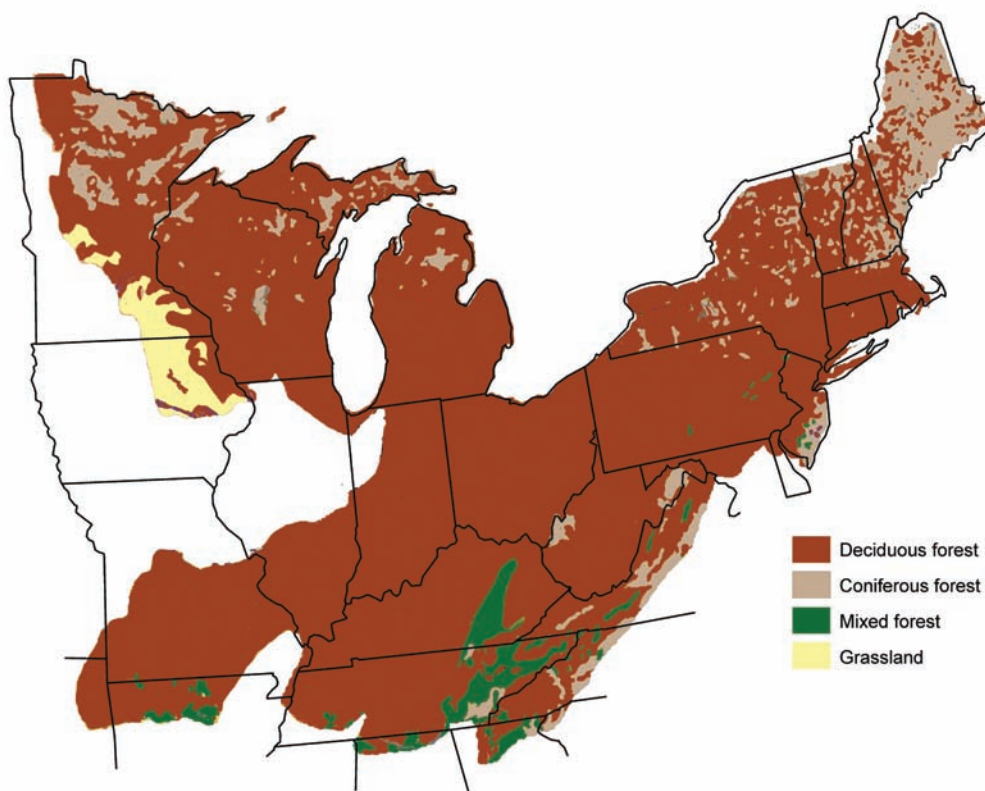

Figure 5-2—Approximate distribution of broad vegetation types in the Northeast bioregion. Riparian areas, wetlands, small grassland patches, and old fields are not shown. (Adapted from Garrison and others 1977.)

Table 5-1—General vegetation types in the Northeast bioregion and perceived threat potential of several nonnative plants in each type (L= low threat, P = potentially high threat, N= not invasive, U = unknown). Designations are **approximations** based on information from state and regional invasive species lists and an informal survey of 17 land managers and researchers in the bioregion.

Species		Plant communities						
Scientific name	Common name	Deciduous forest	Coniferous forest	Mixed forest	Grassland/ old field	Riparian	Fresh wetlands	Tidal wetlands
Acer platanoides	Norway maple	H	P	P	L	H	P	U
Ailanthus altissima	Tree-of-heaven	H	P	P	P	H	P	P
Alliaria petiolata	Garlic mustard	H	H	H	L	H	U	U
Ampelopsis brevipedunculata	Porcelainberry	P	P	P	H	H	P	N
Berberis thunbergii	Japanese barberry	H	H	H	H	H	H	U
Celastrus orbiculatus	Oriental bittersweet	H	H	H	H	H	P	P
Cynanchum louiseae	Black swallow-wort	P	P	P	H	H	P	U
Cynanchum rossicum	Pale swallow-wort	P	P	P	H	H	P	U
Cytisus scoparius	Scotch broom	L	L	L	H	U	U	N
Elaeagnus umbellata	Autumn-olive	P	P	P	H	P	L	N
Euonymus alatus	Winged euonymus	H	H	H	H	H	U	U
Frangula alnus	Glossy buckthorn	H	P	H	H	H	H	H
Glechoma hederacea	Ground-ivy	P	P	P	U	H	P	N
Ligustrum spp.	Privet	H	H	H	P	H	H	U
Lonicera japonica	Japanese honeysuckle	H	P	H	P	H	P	U
Lonicera spp.	Bush honeysuckles	H	H	H	P	H	H	N
Lythrum salicaria	Purple loosestrife	N	N	N	P	H	H	L
Microstegium vimineum	Japanese stiltgrass	H	H	H	H	H	H	U
Paulownia tomentosa	Princesstree	H	H	H	P	P	P	U
Phalaris arundinacea [a]	Reed canarygrass	P	U	U	P	H	H	U
Phragmites australis [a]	Common reed	N	N	N	N	H	H	H
Polygonum cuspidatum	Japanese knotweed	P	P	P	P	H	H	P
Polygonum perfoliatum	Mile-a-minute	P	P	P	H	H	H	U
Pueraria montana var. lobata	Kudzu	P	P	P	H	P	P	P
Rhamnus cathartica	Common buckthorn	H	H	H	H	H	P	N
Robinia pseudoacacia [a]	Black locust	P	H	P	H	P	L	U
Rosa multiflora	Multiflora rose	P	P	P	H	L	P	N

[a] Indicates species that are native in North America, but considered nonnative in all or part of the Northeast bioregion.

Forests _____

Nonnative invasive plants, insects, and pathogens pose a significant threat to forest integrity in eastern North America, especially in conjunction with forest fragmentation and climate change (Luken 2003; Vitousek and others 1997). In pre-colonial times, inland vegetation in the Northeast bioregion was probably dominated by forests with closed canopies and an accumulation of organic matter on the forest floor. These forests had relatively small areas of edge habitat resulting from disturbances such as the death of a large tree, severe storm damage, or fire (Marks 1974, 1983). These openings succeeded rapidly to thickets, young woodland and, in less than a century, forest (Marks 1974). While relatively stable, largely undisturbed forests in the Northeast can be susceptible to establishment and persistence of some shade-tolerant nonnative species such as Japanese stiltgrass, Japanese barberry, privets (*Ligustrum* spp.), bush honeysuckles, Japanese honeysuckle, and Norway maple (Brothers and Spingarn 1992; Ehrenfeld 1997; Fraver 1994; McCarthy 1997; Webb and others 2000). These forests tend to resist invasion by other nonnative plants (Auclair and Cottam 1971; Barton and others 2004; Fraver 1994).

Many nonnatives are restricted to edges and disturbed patches within forests, such as travel corridors including firelanes (Patterson and others 2005), recreation areas (Pyle 1995), sites associated with timber harvest (Buckley and others 2003; Lundgren and others 2004), and areas impacted by severe storm damage (for example, see Taverna and others 2005). These edges and patches typically have a higher abundance of nonnative plant species than forest interiors (Ambrose and Bratton 1990; Brothers and Spingarn 1992; Fraver 1994; Hunter and Mattice 2002; Ranney and others 1981; Robertson and others 1994). Nonnative species that thrive in and can quickly dominate forest edge habitat include Oriental bittersweet, porcelainberry (*Ampelopsis brevipedunculata*), Japanese honeysuckle, kudzu (*Pueraria montana* var. *lobata*), tree-of-heaven (*Ailanthus altissima*), and princesstree (Brothers and Spingarn 1992; McDonald and Urban 2006; Ranney and others 1981; Robertson and others 1994; Saunders and others 1991; Williams 1993). Edges may function as "safe sites" for nonnative invasives, where they can establish, reproduce, and disperse to additional locations including the forest interior (Fraver 1994).

Invasibility of a particular forest site is strongly influenced by its disturbance history, fragmentation of the surrounding landscape, and spatial relationship to propagule sources. Widespread forest clearing for agricultural land use and subsequent abandonment have resulted in secondary forest sites that contain many nonnative plants and propagules (for example, Ashton and others 2005; Bellemare and others 2002; Fike and Niering 1999; Vankat and Snyder 1991). Contemporary forested natural areas and preserves may be an assemblage of forest remnants, abandoned agricultural fields in later stages of succession, woodlots, and streamside corridors embedded in an agricultural and suburban matrix, producing substantial edge habitat (review by Robertson and others 1994). Gaps and edge habitat in northeastern forests are also created by extensive die-off of important canopy trees such as happened with chestnut blight (*Cryphonectria parasitica*) (Myers and others 2004) and Dutch elm disease (*Ophiostoma ulmi*). In canopy gaps resulting from high mortality of eastern hemlock (*Tsuga canadensis*) from the nonnative hemlock wooly adelgid (*Adelges tsugae*), several nonnative species had high cover; these included Japanese stiltgrass, Oriental bittersweet, Japanese barberry, and tree-of-heaven (Orwig and Foster 1998). Similarly, canopy gaps created by wild or prescribed fire could provide seed beds and edge habitat for nonnative invasive populations, although little research is available on this topic. Propagule pressure from existing nonnative invasive populations, coupled with establishment opportunities provided by ongoing disturbances and forest fragmentation, will likely lead to continued spread of these species.

Deciduous and Mixed Forests _____

Background

Two major deciduous forest types in the Northeast bioregion include the maple-beech-birch and oak-hickory ecosystems described by Garrison and others (1977). These types are treated separately here based on differences in fire ecology. The mixed forest type described below is typically dominated by oaks and pines, especially northern red oak and eastern white pine. Maples, birches, beech, and hemlock are common associates.

Maple-beech-birch Forests—Vegetation in this ecosystem includes northern hardwood forests; southward it transitions into mixed mesophytic hardwoods, as discussed by Wade and others (2000). The northern hardwoods occur on mesic and fire protected sites in the Lake States and farther east. The dominant hardwood species include sugar maple, yellow birch (*Betula alleghaniensis*), American beech (*Fagus grandifolia*), and basswood (*Tilia americana*) in the Midwest. Northern hardwoods mix with boreal spruce (including *Picea glauca*, *P. mariana*, and *P. rubens*) and balsam fir to the northeast, and with eastern hemlock, eastern white pine and oaks to the north, south, and west. Mixed mesophytic hardwoods occupy the transition zone between northern hardwood forest and oak-hickory forest, and contain a large diversity of canopy tree species. This type transitions

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into sugar maple-beech-birch forest in northern West Virginia, southwestern Pennsylvania, and southern Ohio in the north, and into the oak-hickory-pine type in northern Alabama in the south (Wade and others 2000). Dominant species include sugar maple, beech, basswood, white oak (*Quercus alba*), northern red oak (*Q. rubra*), buckeye (*Aesculus octandra*), and tulip tree (*Liriodendron tulipifera*) (Küchler 1964).

Presettlement fire regimes in eastern deciduous forests varied among forest types. Charcoal evidence suggests that fires were more common in mixed meso-phytic forests than in northern hardwood forests (Wade and others 2000). Northern hardwood forests are not very flammable and if fires penetrate the forest, they tend to burn as patchy, creeping surface fires. Crown fires are unusual in eastern deciduous forests (Lorimer 1977; Turner and Romme 1994). Fire return intervals are estimated to exceed 1,000 years throughout the northern hardwoods type and are estimated at 35 to over 200 years in the mixed mesophytic type. Both types are characterized by a mixed-severity fire regime (Wade and others 2000); that is, if fire does occur in these forests it would cause selective mortality in dominant vegetation, depending on the susceptibility of differ-ent tree species to fire (Brown 2000). Fires may have occurred more frequently in areas that were burned by Native Americans and where conifers occur as substantial components of the hardwood forests (Wade and others 2000). Although northern hardwood species are generally thought to have little resistance to fire, maple and birch sprout vigorously from the stump, and beech suckers from the root system. See reviews on individual species in the Fire Effects Information System (FEIS) for more information on fire ecology of dominant species in these ecosystems. Also see Wade and others (2000) and Parshall and Foster (2002) for more information on fire regimes in northern hardwood and mixed mesophytic forest types.

Oak-hickory Forests and Oak Savanna—The oak-hickory ecosystem is extensive in the Northeast bioregion, reaching from southern Maine, southwest along the Appalachian Highlands to the northern part of Georgia and Alabama, and westward to the oak savannas and central grasslands. The oak-hickory ecosystem varies from open to closed woods with a sparse to dense understory of shrubs, vines, and her-baceous plants. Associated species vary with latitude and location (Garrison and others 1977). Over three dozen species of oak and almost two dozen species of hickory are possible in the overstory of this ecosystem. It includes oak-hickory and Appalachian oak ecosystems as described by Küchler (1964) and becomes an oak-hickory-pine type in the Mid-Atlantic States, including stands that can be classified as mixed mesophytic forest (Wade and others 2000). At its western extent, this forest type grades into open oak woodlands and oak savannas. Oak savannas are associated with prairies and are generally dominated by prairie grasses and forbs, with widely spaced groves or individual trees (review in LANDFIRE Rapid Assessment 2005a).

According to a review by Wade and others (2000), the presettlement fire regime in the oak-hickory ecosystem was characterized by high frequency, understory fires often ignited by Native Americans. Presettlement fire frequencies are not known but are estimated between 3 and 35 years. Subsequent settlement by Euro-Americans, who used fire for many of the same reasons as Native Americans, increased the frequency and extent of burning in oak-hickory forests. Fire intervals decreased to less than 10 years, and many sites burned annually. Frequent fire maintained open oak-hickory woodlands with large, old, fire-resistant trees and a groundcover of grasses and forbs. Shrubs, understory trees, and woody debris were rare in oak-hickory forests and savannas. Where present, erica-ceous shrubs such as mountain laurel (*Kalmia latifolia*) and rhododendron (*Rhododendron* spp.) could burn with extreme fire behavior, resulting in mixed-severity or stand-replacing fires (Wade and others 2000). Fire regimes in oak savanna are characterized primarily as frequent surface fires occurring at about 4-year intervals (LANDFIRE Rapid Assessment 2005a).

Exclusion of fire has profoundly changed oak-hickory forests and oak savannas and, in many areas, has led to dominance by mixed mesophytic and northern hardwood species and allowed the mid-story canopy to close and shade out herbaceous plants (LANDFIRE Rapid Assessment 2005a; Wade and others 2000). Managers in Virginia note replacement of some oak-hickory stands by maple and beech where fire has been excluded (Gorman, personal communication 2005).

Surface fires enhance regeneration of oak and hickory, and there has been much recent research on use of prescribed fire to promote establishment of oak (Boerner and others 2000a, b; Dey 2002b; Kuddes-Fischer and Arthur 2002; Lorimer 1993; Wade and others 2000). In upland oak-hickory forests at Quantico Marine Base and Fort Pickett Military Reservation, fre-quent fires associated with training activities enhance the oak-hickory community, including an endangered shrub, Michaux's sumac (*Rhus michauxii*) (Virginia Department of Conservation and Recreation 2005). In areas where fire has been excluded and fuel loads are high, reintroduction of fire might need to be phased in with a series of fuel reduction treatments.

Mixed Forests—Presettlement fire regimes in mixed forests were characterized by a range of fire frequencies and fire severities. Estimates given by Wade and others (2000) for mixed forest types range from understory fires with return intervals of less than 10 years in shortleaf pine-oak communities, to mixed-severity fires with return intervals between

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10 and 35 years in Virginia pine-oak communities, to stand-replacement fires with return intervals greater than 200 years in northern hardwoods types with components of spruce or fir. Fire regimes for white pine-red oak-red maple communities are thought to consist of stand-replacement fires with return intervals of less than 35 to 200 years (Duchesne and Hawkes 2000).

Nonnative invasive plants that threaten forests in the Northeast bioregion are similar among deciduous and mixed forests (table 5-1), so they are discussed together here. Distinctions are made among forest types where possible.

Role of Fire and Fire Exclusion in Promoting Nonnative Plant Invasions in Deciduous and Mixed Forests

Managers are concerned about the potential for establishment and spread of nonnative plants after fire. Evidence is sparse regarding postfire response of nonnatives for northeastern deciduous and mixed forests, although inferences may be possible based on life history and reproductive traits (table 5-2). Species such as Japanese honeysuckle and Oriental bittersweet can persist in low numbers in the understory and spread following canopy or soil disturbance, while others such as tree-of-heaven may establish in open areas via long-distance seed dispersal. Several invasive species (for example, tree-of-heaven, autumn-olive, Japanese barberry, privet, honeysuckles, and kudzu) are able to reproduce vegetatively and sprout following top-kill. These species could spread and possibly dominate postfire communities. While there is little hard evidence of seed banking for invasive plants, observations suggest that some species (for example, princesstree, buckthorn, multiflora rose, kudzu, ground-ivy, and Japanese stiltgrass) may establish from the soil seed bank after fire (table 5-2). Some invasive species (for example, common buckthorn, bush honeysuckles) have established and spread in areas such as oak-hickory forests and oak savannas, where fire has been excluded from plant communities adapted to a regime of frequent surface fires.

Many woody invasives in the Northeast bioregion have some traits in common, including the ability to sprout following top-kill (table 5-2). Managers at Virginia-area national parks note that autumn-olive sprouts following aboveground damage, and sprouts are especially vigorous following dormant season burning in oak woodlands (Gorman, personal communication 2005; Virginia Department of Conservation and Recreation 2002a). Similarly, at Fort Devens, Massachusetts, autumn-olive established from both root sprouts and seedlings following a single fire (Poole, personal communication 2005). Observations indicate that a related nonnative species, Russian-olive (*Elaeagnus*

angustifolia), sprouts after fire in the Central and Interior West bioregions (chapters 7 and 8). Russian-olive occurs in northeastern deciduous forests but is considered less invasive than autumn-olive in this bioregion (Mehrhoff and others 2003). While available literature does not describe postfire response of Norway maple, Simpfendorfer (1989) lists it among species that regenerate by coppicing following fire. It is also likely that, if Norway maple saplings and seedlings survive fire, they would respond favorably to gap formation (Munger 2003a, FEIS review). Tree-of-heaven produces abundant root sprouts after complete top-kill from fire (Howard 2004a, FEIS review). Japanese barberry sprouted after cutting and/or burning treatments in a deciduous forest site in western Massachusetts, although total cover of this species was reduced 2 years after burning (Richburg 2005). Observations in a mixed forest in northwestern Georgia indicate that Chinese privet responds to aboveground damage from fire by vigorously sprouting from the root crown (Faulkner and others 1989), and an anecdotal account suggests that Japanese privet (*Ligustrum japonicum*) can "resprout following fire" (Louisiana State University 2001, review). Other privet species are likely to sprout from roots and/or root crowns following fire; however, documentation is lacking (Munger 2003c, FEIS review). Glossy buckthorn sprouted from roots or root crowns after wildfire in a mixed alvar woodland near Ottawa. Sprouts were 3 to 5 feet (1 to 1.5 m) tall after 100 days, but no prefire data were available for comparison. Two nonnative grasses, Canada bluegrass and redtop (*Agrostis gigantea*), occurred in burned areas in these studies at 22 and 18 percent frequency, respectively (Catling and others 2001). While only anecdotal evidence is available suggesting postfire sprouting in multiflora rose (Virginia Department of Conservation and Recreation 2002d), in a deciduous forest in southeastern Ohio germination and recruitment of multiflora rose was higher on open-canopy plots and on plots treated with high-severity prescribed fire than in control plots (Glasgow and Matlack 2007).

Some invasive vines can occur in the forest understory in small numbers and spread via vegetative regeneration or recruitment from the soil seed bank following disturbance. Oriental bittersweet and Japanese honeysuckle often occur under closed canopies, and when disturbance creates canopy gaps, these vines can grow and spread rapidly (Howard 2005a; Munger 2002a, FEIS reviews). Several sources indicate that Japanese honeysuckle sprouts after damage from fire, and postfire sprouting can lead to rapid recovery of preexisting populations. Scattered subpopulations of Japanese honeysuckle can also persist with frequent fire, possibly within fire refugia or via continued recruitment from bird-dispersed seed (Munger 2002a). Kudzu stems and foliage are likely to resist fire damage during

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Table 5-2—Biological and ecological characteristics of nonnative species selected from table 5-1 that occur in forested habitats in the Northeast bioregion. Information is derived from a comprehensive review of available literature for each species. Superscripted letters following plant names refer to citations for biological and ecological information in footnotes. Superscripts in right-hand column indicate information type: * indicates anecdotal information; ** indicates information from experiments; *** indicates a literature review in which additional fire information is available.

Species	Seed dispersal/ establishment	Seed banking	Vegetative reproduction	Shade tolerance	Available fire information
Trees					
Norway maple[a]	Wind-dispersed	No information	Anecdotal evidence of regeneration by coppicing following fire; cut trees can sprout from stumps	Yes	Simpfendorfer 1989*
tree-of-heaven[b]	Abundant, wind-dispersed seeds	Transient	Abundant, rapidly growing sprouts from roots, root crown, and/or bole following complete top-kill or bole damage	Low	Gorman, personal communication 2005*; Hoshovsky 1988*
princesstree[c]	Abundant, wind-dispersed seed (up to 4 km); establishes best on exposed mineral soil with high light levels	Viable seeds found in forest soil seed banks	Sprouts from roots and/or stumps after pulling or cutting, and may also do so after fire	No	Reilly and others 2006*; Langdon and Johnson 1994*
Shrubs					
Japanese barberry[d]	Gravity-, bird- and possibly animal-dispersed seed; high rates of seedling recruitment	No information	Sprouted after cutting and burning in deciduous forest and forested swamp sites; cover substantially reduced two years after cutting and/or burning treatments	Yes	Richburg 2005**; D'Appollonio 2006**
autumn-olive[e]	Abundant bird-dispersed seed	No information	Anecdotal evidence suggests that it can sprout following damage from fire	Low to moderate	Gorman, personal communication 2005*; Virginia Department of Conservation and Recreation 2002a*; Poole, personal communication 2005*

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Table 5-2—(Continued)

Species					
winged euonymus[f]	Large quantities of gravity- and bird-dispersed seed	No information	Anecdotal evidence suggests that it spreads by root suckers and can sprout following damage to aboveground parts	Very	none
buckthorns[g]	Bird- and animal-dispersed seed; recruitment best on sites with ample light and exposed soil	Seed requires scarification for germination. Research on seed banking is not available	Glossy buckthorn sprouts from roots or root crowns after fire; common buckthorn sprouted following prescribed fire in a grassland habitat; in an oak savanna, spring prescribed fire killed common buckthorn seedlings while mature buckthorn were top-killed and sprouted after fire	Low to moderate	Apfelbaum and Haney 1990*; Converse 1984a, TNC review*; Heidorn 1991*; Larson and Stearns 1990*; McGowan-Stinski 2006*; Boudreau and Willson 1992**; Catling and others 2001**; Neumann and Dickmann 2001**; Post and others 1990**; Richburg 2005**
privets[h]	Bird- and animal-dispersed seed	Transient	Anecdotal evidence suggests that privets can sprout following damage (including fire) to aboveground parts; and observations indicate that Chinese privet responds to fire by vigorously sprouting from the root crown	Moderate	Gorman, personal communication 2005*; Louisiana State University 2001* (Japanese privet); Faulkner and others 1989** (Chinese privet)
bush honeysuckles[i]	Abundant, wind-dispersed seed	Not likely to form a persistent seed bank, but information is lacking	Postfire sprouting of bush honeysuckles has been observed; spring prescribed burning may kill bush honeysuckle seedlings and top-kill larger plants, although results have been mixed	Yes	Apfelbaum and Haney 1990*; Kline and McClintock 1994*; Mitchell and Malecki 2003**; Maine Department of Conservation, Natural Areas Program 2004*; Solecki 1997*
multiflora rose[j]	Gravity and bird-dispersed seed	Persistent seed bank	Reproduces by root suckering and layering; sprouting is reported following top-kill, including fire; germination and seedling growth promoted in experimentally burned microsites	Low to moderate	Hruska and Ebinger 1995**; Virginia Department of Conservation and Recreation 2002d*; Glasgow and Matlack 2007**
Woody vines					
porcelainberry[k]	No info	No information	Spreads by "extensive underground growth"; able to reproduce vegetatively from stem or root segments	Low	None

Table 5-2—(Continued)

Oriental bittersweet[l]	Bird-dispersed seed	Conflicting info; may have small, persistent seed bank	Deeply buried perennating buds; capable of sprouting from the root crown, roots, root fragments, and runners; damage to branches, root crowns, runners, or roots encourages sprouting	Yes	None
Japanese honeysuckle[m]	Bird-dispersed seed	Possible, but information is lacking	Sprouts from root crowns after damage from fire. Repeated burning can have negative impacts	Yes	Munger 2002a***; Faulkner and others 1989**; Schwegman and Anderson 1986**
kudzu[n]	No information	Mature seeds are dormant and require scarification for germination so seeds are likely to persist, although information on longevity is lacking	Sprouts from the root crown after fire; seed dormancy may be broken by high temperatures	No	Munger 2002b***; Rader 2000**; Miller 1988*
Herbaceous vines					
swallow-worts[o]	Abundant gravity- and wind-dispersed seed	No information	Mature plants of both species sprout readily from perennating buds on the root crown occurring below the soil surface and likely to be protected from fire; fire thought to be ineffective for controlling these species	Populations on shaded sites may persist for years and expand rapidly if a gap is created	Lawlor 2002, TNC review*; Lawlor 2000*; DiTommaso and others 2005*; Sheeley 1992*; Richburg 2005**
mile-a-minute[p]	Bird-, animal-, and water-dispersed seed	Seeds can remain dormant in the soil seed bank for at least three years	No info; annual species	Tolerates light shade	None
Herbs					
garlic mustard[q]	Primarily short-distance, gravity and ballistic dispersal; (long-distance dispersal by animals is possible)	Mostly transient, though some seeds may persist for several years	Top-killed when exposed to fire, but can survive by sprouting from the root crown	Moderate	Hintz 1996**; Luken and Shea 2000**; Munger 2001***; Nuzzo and others 1996**

Table 5-2—(Continued)

		Seeds likely to persist in soil seed bank			
ground-ivy[r]	No information		Spreads by stolons with roots at each node; displayed rapid vegetative growth after a spring prescribed fire in a mixed forest	Somewhat	Chapman and Crow 1981*
Japanese stiltgrass[s]	Water-, wind- and animal-dispersed seed	Soil-stored seed may remain viable for 3 to 5 years	An annual plant that can grow back from tillers and stolons following top-kill from early-season fire; can establish from the soil seed bank after fire; germination and seedling growth promoted in experimentally burned microsites	Very; but shows significant increase in biomass following canopy removal	Barden 1987*; Gorman, personal communication 2005*; Tu 2000, TNC review*; Glasgow and Matlack 2007**
reed canarygrass[t]	No information	Extensive germination from the soil seed bank after fire and herbicide application	Vigorous, rapidly spreading rhizomes	No	Howe 1994b**; Apfelbaum and Sams 1987*; Hutchison 1992b*; Preuninger and Umbanhowar 1994*

[a] Munger 2003a; Webb and others 2001; Martin and Marks 2006.
[b] Howard 2004a, FEIS review; Hutchinson and others 2004; Marsh and others 2005.
[c] Williams 1993; Dobberpuhl 1980; Hyatt and Casper 2000; Johnson, K. 1996, review.
[d] Mehrhoff and others 2003; Rhoads and Block 2002, TNC review; Ehrenfeld 1997; Ehrenfeld 1999; Richburg and others 2001, review; Johnson, E. 1996; Silander and Klepeis 1999; D'Appollonio 2006.
[e] Munger 2003b, FEIS review.
[f] Mehrhoff and others 2003, review; Miller 2003, review; Ebinger 1983; Martin 2000, TNC weed alert; Ma and Moore 2004.
[g] Converse 1984a, TNC review; Chapter 7, this volume.
[h] Mehrhoff and others 2003; Munger 2003c, FEIS review; Miller 2003, review.
[i] Munger 2005a, FEIS review; Bartuszevige and Gorchov 2006.
[j] Munger 2002c, FEIS review; Szafoni 1991; Richburg 2005.
[k] Antenen 1996, review; Antenen and others 1989; Yost and others 1991; Robertson and Antenen 1990.
[l] Howard 2005a, FEIS review.
[m] Munger 2002a, FEIS review.
[n] Munger 2002b, FEIS review; Susko and others 2001.
[o] Lawlor 2002, TNC review; Cappuccino and others 2002; Lumer and Yost 1995; Smith and others 2006; Sheeley and Raynal 1996.
[p] Kumar and DiTommaso 2005.
[q] Munger 2001, FEIS review.
[r] Hutchings and Price 1999; Mitich 1994.
[s] Cole and Weltzin 2004; Cole and Weltzin 2005; Flory and others 2007; Howard 2005c, FEIS review; Oswalt and others 2007; Winter and others 1982 .
[t] Hoffman and Kearns 2004; Leck and Leck 2005; Snyder 1992b, FEIS review.

the growing season because they typically maintain high water content and are relatively unflammable except after frost-kill in autumn (Wade, personal communication 2005). When large kudzu plants do burn, they can sprout from the root crown after top-kill and reestablish soon after dormant-season fire, returning to prefire abundance by the second postfire growing season. Additionally, soil heating by fire may promote kudzu seed germination by scarifying the seedcoat. However, dormant season fire can kill root crowns of small, newly established kudzu plants (Munger 2002b, FEIS review).

Fire information is available for only a few species of the nonnative herbs occurring in northeastern forests. Researchers report that garlic mustard can establish and persist after fire in northeastern deciduous forests. Establishment of garlic mustard from the soil seed bank may be facilitated by postfire conditions (Munger 2001, FEIS review). Repeated fall burning (2 to 3 annual burns) did not reduce abundance or relative importance of garlic mustard in an eastern mesophytic forest understory in Kentucky (Luken and Shea 2000). In a white pine-mixed deciduous forest in New Hampshire, ground-ivy displayed rapid vegetative growth after a spring prescribed fire but did not occur on fall-burned plots (Chapman and Crow 1981). Addtionally, some evidence suggests that ground-ivy seeds might survive in the soil seed bank for a number of years (Hutchings and Price 1999) and therefore may be capable of recruitment after fire. Japanese stiltgrass can establish in forest understories (for example, see Dibble and Rees 2005; Ehrenfeld 2003), in gaps created by overstory mortality (Orwig and Foster 1998), and in disturbed areas created by hurricanes (Taverna and others 2005). In oak-hickory woodland sites, Japanese stiltgrass established after mechanical thinning and prescribed fire in southern Illinois (Anderson and others 2000) and established from the soil seed bank after an accidental fire on a North Carolina floodplain (Barden 1987). In deciduous forest in southeastern Ohio, germination and growth of Japanese stiltgrass was higher in open-canopy plots and in plots treated with high-severity prescribed fire than in untreated controls (Glasgow and Matlack 2007). Observations indicate that it also grows back from tillers and stolons following top-kill from early-season fire (Tu 2000, TNC review).

Fire exclusion affects forest types differently depending on the extent to which they are dominated by fire-adapted species or have gap formation. Many nonnative invasive species in the Northeast bioregion are shade-tolerant (table 5-2) and thus may invade forests and savannas where fire has been excluded. Hobbs (1988) suggested that common buckthorn, known for its shade tolerance, may have spread in northeastern deciduous forests in part because of fire exclusion, as well as in gaps that occurred with the demise of American elm. Common buckthorn and bush honeysuckles have infested oak savanna remnants where fire has been excluded. In a study of 24 oak savanna remnants in northern Illinois, Indiana, and southern Wisconsin, Apfelbaum and Haney (1990) suggested that, without fires about every 35 to 100 years, native and nonnative woody species, including common buckthorn and bush honeysuckles, establish in oak understories and interfere with oak regeneration. In 24 oak savannas that varied in soil type and management history, periodic fires afforded some control of mesic shrub infestations and promoted oak regeneration (Apfelbaum and Haney 1990). Glossy buckthorn is typically associated with unburned woodland in mixed forests (Catling and Brownell 1998; Catling and others 2002). A relatively shade-tolerant herbaceous species, garlic mustard, occurred in areas of low ambient light where reduced fire frequency resulted in increased tree canopy cover in a northern Illinois oak savanna remnant (Bowles and McBride 1998). Invasive populations of Japanese honeysuckle apparently do not occur in communities with frequent, low-severity fires (Munger 2002a). More data are needed to uncover whether this pattern can be related to fire exclusion.

Fire exclusion from fire-seral communities such as oak-hickory forests or oak savannas diminishes opportunities for maintenance of the dominant species. Instead, establishment and growth of shade-tolerant species are enhanced, and fire-seral species can be replaced. Once shade-tolerant nonnative species establish in closed-canopy forests, they may persist, spread, and possibly dominate the understory. Evidence suggests that Norway maple establishes, persists, and grows in forest understories in the absence of fire or other stand-level disturbances. In a New Jersey piedmont forest, for example, Norway maple, American beech, and sugar maple are gradually replacing white oak, northern red oak, and black oak (*Q. velutina*), which were formerly dominant (Webb and Kaunzinger 1993; Webb and others 2000). Nonnative grasses (sweet vernal grass (*Anthoxanthum odoratum*), fineleaf sheep fescue (*Festuca filiformis*), Japanese stiltgrass (Dibble and Rees 2005), and Canada bluegrass (*Poa compressa*) (Swan 1970)) often occur in the understory of deciduous forests and seem to spread in the absence of fire. Of these, Japanese stiltgrass is the most studied and is one of the most invasive grasses in forests and riparian areas, especially along trails and roadsides but also in undisturbed, shaded sites (Cole and Weltzin 2004). It can invade woodlands with incomplete canopy closure (Winter and others 1982) and persist after the canopy closes completely (Howard 2005c). If fire is reintroduced to these communities, restoration of native plant communities is not certain; shade-tolerant nonnatives that can sprout after top-kill or establish from the seed bank may dominate the postfire community.

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Effects of Nonnative Plant Invasions on Fuel Characteristics and Fire Regimes in Deciduous and Mixed Forests

Nonnative plant invasions may change fuel properties, fire behavior, and possibly fire regimes in several ways (Richburg and others 2001). Individual species could affect fire behavior due to differences from native species in heat content (Dibble and others 2007), moisture content, volatility, fuel packing, and phenology. Nonnative grasses have the potential to increase biomass and continuity of fine fuels on invaded sites (Dibble and Rees 2005). Invasive shrubs and vines may affect biomass and flammability of the shrub and herb layers or act as ladder fuels. While increased fuel loads due to nonnative invasive plants might not be a concern to fire managers in wet years, hazard fuels must be considered if drought occurs. However, properties of individual species might be less important than fuel moisture, topography, and wind velocity (Ducey 2003) during a wildfire.

There is concern that encroachment by nonnative grasses, vines, and shrubs could increase flammability and fuel continuity in deciduous forests. Fuels in five invaded mid-successional deciduous forest stands dominated by oak-hickory (Maryland), poplar (Maine), oak-bigtooth aspen (Maine), oak-yellow poplar (New Jersey), and mixed hardwoods (Vermont) were studied by Dibble and others (2003) and by Dibble and Rees (2005). Invaded stands were compared to nearby uninvaded stands. Under invaded conditions, graminoid and shrub cover were greater because of the frequency and height of the nonnative plants (including fineleaf sheep fescue, sweet vernal grass, Japanese stiltgrass, bush honeysuckles (fig. 5-3), Japanese barberry, and others). If fire occurs in invaded stands, patches of fine

fuels represented by nonnative grasses and shrubs could increase fire intensity (Dibble and Rees 2005). For example, Japanese stiltgrass forms large (≥ 0.2 ha), dense patches with hundreds to thousands of stems per square foot (Dibble, unpublished data 2005; Dibble and Rees 2005). This species produces large amounts of litter and fine fuels, and stems lie down soon after they die in autumn, creating a continuous fuelbed of matted straw (Barden 1987) that may constitute an increase in biomass and continuity of fine fuels compared to uninvaded sites. More information on fuel properties of several native and nonnative grasses is available (Dibble and others 2007).

Invasive vines such as Oriental bittersweet (fig. 5-4), Japanese honeysuckle, kudzu, Chinese wisteria (*Wisteria sinensis*), porcelainberry, and English ivy (*Hedera helix*) have potential to alter fuel characteristics of invaded communities. They could increase fuel loading and continuity by growing up and over supporting vines, shrubs, and trees, and by killing the vegetation beneath them. Invasive vines could increase the likelihood of crown fire, especially under drought conditions, by acting as ladder fuels. Such changes have not been quantified. In the southern Appalachians, Oriental bittersweet contributes substantial vine biomass (Greenberg and others 2001). It can also support later-successional vines and lianas (Fike and Niering 1999), possibly enabling other species to become ladder fuels

Figure 5-3—Bush honeysuckle in the understory of a mixed forest that developed on an abandoned agricultural field (Bradley, Maine). (Photo by Alison C. Dibble.)

Figure 5-4—Oriental bittersweet has completely overtaken this eastern white pine (Rockland, Maine). (Photo by Alison C. Dibble.)

(Howard 2005a). In a deciduous forest in New York, several gaps were occupied by porcelainberry growing over Amur honeysuckle. In some quadrats, the cover of these two species combined was well over 100 percent, and few tree seedlings and herbs grew beneath the tangled canopy (Yost and others 1991).

In deciduous forests where invasive plants are prevalent, the morphology and stand structure of invasive Japanese barberry may alter fuel characteristics (Dibble and Rees 2005). Individual plants consist of multiple stems originating from the root collar and varying in length and morphology. Stems die after a few years, as new stems sprout from the base (Ehrenfeld 1999; Silander and Klepeis 1999). Japanese barberry populations can become dense, nearly impenetrable thickets within 15 years of initial establishment (Ehrenfeld 1999), even under closed canopies. Populations may become so thick that they shade out understory species (Johnson, E. 1996, review).

In some cases, nonnative vegetation might decrease the potential for ignition and spread of fire, although there are no studies documenting this in northeastern deciduous forests. Mile-a-minute (*Polygonum perfoliatum*), Japanese barberry, privet, kudzu, Japanese honeysuckle, and Oriental bittersweet are thought to reduce flammability on some oak-hickory sites in Virginia-area national parks. For example, mile-a-minute vine produces a dense mass of succulent, almost nonflammable vegetation. Where mile-a-minute dominates, managers are concerned that the use of prescribed fire to promote regeneration of desirable native species may not be possible. Japanese barberry and privet displace native, flammable ericaceous species including mountain laurel and blueberry (*Vaccinium* spp.), and there is concern that dominance by nonnative species may reduce flammability of the invaded community (Gorman, personal communication 2005). In dense thickets of Chinese privet in northwestern Georgia, prescribed fire was spotty and erratic. Lack of fire spread in privet infestations might be explained by moist and compacted privet litter or by the affinity of Chinese privet for moist, low-lying soils (Faulkner and others 1989). Because kudzu stems and foliage maintain high water content, flammability of invaded sites may be reduced even during drought, when desired native plants become susceptible to fire due to desiccation (Munger 2002b). Similarly, it has been suggested that dense stands of garlic mustard may inhibit the ability of a forest understory to carry surface fire (Nuzzo 1991).

Differences in phenology between native and non-native species could theoretically affect fire seasonality, rendering a community more or less flammable during particular seasons. For example, buckthorns (Converse 1984a, TNC review) and bush honeysuckles (Batcher and Stiles 2000) leaf out earlier than native vegetation and retain their leaves later into autumn. This topic deserves further study.

Use of Fire for Controlling Nonnative Invasives in Deciduous and Mixed Forests

Fire alone is probably not sufficient to control most invasive species in deciduous forests of the Northeast bioregion because high fuel moisture and insufficient fuel accumulation limit both fire severity and the frequency with which burning can be conducted. In general, a long-term commitment and some combination of control treatments will likely be more effective for controlling invasive species than any single approach (Bennett and others 2003). Additionally, the usefulness and effectiveness of prescribed fire differ among forest types and depend to some extent on the fire types and frequency to which native plant communities are adapted. In this sense, prescribed fire is more likely to be an effective tool for controlling invasive species and promoting native vegetation in oak-hickory forests and oak savannas than in other forest types.

In maple-beech forests, a lack of dry surface fuels and/or a brief weather window for burning make the use of prescribed fire difficult. Additionally, if managers seek to maintain an overstory of fire-intolerant species such as maple and beech, burning under conditions where fires are severe enough to kill nonnative species will likely kill desired species as well. Using prescribed fire to control Norway maple, for example, would probably be detrimental to sugar maple and American beech.

Repeated prescribed burning may be more effective in oak-hickory forests and oak savannas than in maple-beech forests because dominant native species in these plant communities are adapted to relatively frequent fires. In this case, fire may be appropriate where management goals include controlling nonnative species or reducing fuels, accompanied by maintenance of native seral species.

Where conditions are appropriate for carrying a surface fire, nonnative invasive trees such as Norway maple and tree-of-heaven may be top-killed by fire, but both species can sprout following top-kill (Howard 2004a; Webb and others 2001). Observations indicate that seedlings of tree-of-heaven are killed by fire, but larger individuals tend to survive and sprout after fire (Gorman, personal communication 2005), even following heat-girdling (Hoshovsky 1988, TNC review). No experimental information is available on the effects of fire on Norway maple; however, cutting Norway maple resulted in sprouting from both seedlings and larger trees the following summer (Webb and others 2001). Additionally, removal of Norway maple from the canopy of a mixed maple forest in New Jersey resulted in a dramatic floristic and structural change in some

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areas, with establishment of both native and nonnative plant species not previously seen in the forest. Among the new arrivals were tree-of-heaven, Japanese barberry, winged euonymus, Japanese honeysuckle, wineberry (*Rubus phoenicolasius*), black locust, and garlic mustard. It is unclear whether these species established from the seed bank or from off-site sources. Removal of Norway maple seedlings also resulted in a large pulse of Norway maple recruitment (Webb and others 2001).

Sprouting is reported in many invasive woody species following top-kill (table 5-2). Prescribed fire during the dormant season is generally ineffective for controlling invasive shrubs in the Northeast bioregion. These fires reduce shrub cover temporarily and may kill seedlings and smaller plants, but populations are not controlled as shrubs resprout (Richburg 2005; Richburg and others 2001). However, on a study site in a mature deciduous forest in western Massacusetts, cover of Japanese barberry was significantly reduced 2 years after both cutting and burning (conducted in April and November), with the greatest reductions in areas where treatments were combined (Richburg 2005). Results of a study on the use of fire to control Japanese honeysuckle, Chinese privet, and native poison ivy in an oak-hickory-pine forest in northwestern Georgia are relevant to forests in the Northeast bioregion. Both fall and winter burns significantly ($P < 0.05$) reduced Japanese honeysuckle biomass. However, sprouting from buds protected by unburned litter was evident as early as 1 month following fire. Chinese privet showed no significant response to fire or season of burning, and many plants sprouted from root crowns. The response of privet to fire was unclear because fire did not spread well in privet thickets (Faulkner and others 1989).

It has been suggested that repeated prescribed fire may be effective for controlling species such as bush honeysuckles (Munger 2005a, FEIS review; Nyboer 1992), privets (Batcher 2000a, TNC review), and multiflora rose (Virginia Department of Conservation and Recreation 2002d). However, little empirical evidence is available to support these suggestions for northeastern forests. Additionally, repeated burns may be limited by insufficient fuel accumulation to carry fires that are scheduled close together (for example, see Richburg 2005). Repeated prescribed fire has been used with some success for controlling nonnative shrubs such as common buckthorn in oak savannas (chapter 7).

Japanese honeysuckle was reduced by repeated prescribed burning in North Carolina shortleaf pine forest and in an Illinois barren remnant (Munger 2002a). However, cessation of prescribed fire treatments, even after multiple consecutive or near-consecutive years of burning, can lead to reinvasion (Schwegman and Anderson 1986). Anecdotal evidence suggests that kudzu may also be controlled by repeated prescribed fire under certain conditions. Managers in Virginia observed that 3 to 4 years of prescribed fire late in the growing season can eliminate kudzu in the treated area (Gorman, personal communication 2005).

Little information is available on the use of fire to control invasive grasses in deciduous forests. It has been suggested that Japanese stiltgrass is not controlled by spring burning or mowing in oak-hickory forests because seeds germinate from the soil seed bank after treatment, and plants may grow rapidly enough to set seed that same year (Virginia Department of Conservation and Recreation 2002b). More effective control of Japanese stiltgrass might be achieved by timing prescribed fire before seeds ripen but late enough in the season to prevent a second flush of seed production (Gorman, personal communication 2005). In an oak savanna in Wisconsin, early April burning was not effective for controlling reed canarygrass (*Phalaris arundinacea*); fire appeared to enhance its spread. Burning in mid to late May weakened reed canarygrass and prevented seed production, though it did not eliminate the infestation and was detrimental to desired native herbs such as shooting star (*Dodecatheon media*) (Henderson 1990). For more information on control of reed canarygrass using prescribed fire, see the "Riparian and Wetland Communities" section, page 82.

Prescribed fire can be used to temporarily control garlic mustard under some conditions. However, garlic mustard has a moderately persistent seed bank and rapid population growth, and some individuals are likely to survive understory and mixed-severity fires in deciduous and mixed forests due to the patchiness of these fires (Munger 2001). Three consecutive years of prescribed burning in a central Illinois black oak forest failed to eradicate garlic mustard. One reason was that individuals survived in protected, unburned microsites such as the lee of a downed log or a patch of damp litter, and these survivors were successful in producing seed (Nuzzo and others 1996). Additionally, removal of garlic mustard may lead to proliferation of other undesirable species, so caution is warranted to avoid interventions that may be detrimental to the native community (McCarthy 1997).

Coniferous Forests _____

Background

Coniferous forests in the Northeast bioregion include white-red-jack pine ecosystems in the Great Lakes area; pitch pine communities in parts of the New England coast, the New Jersey Pine Barrens, and upstate New York; and spruce-fir (*Picea-Abies*) ecosystems in the Lake and New England States and at high elevations in the Appalachian Mountains. Other coniferous forest types include Virginia pine (*Pinus virginiana*), shortleaf pine (*Pinus echinata*), and Table Mountain pine

(*Pinus pungens*) within the loblolly-shortleaf pine and oak-pine ecosystems described by Garrison and others (1977). Some of these coniferous forest types also occur in the Southeast bioregion. More information is available on interactions between fire and invasive species in pine forests and savannas in chapter 6.

The white-red-jack pine ecosystem occurs on plains and tablelands of the northern Lake States and parts of New York and New England. In the Lake States these forests are used principally for timber and recreation, while large urban areas fragment this ecosystem in the northeast (Garrison and others 1977). Prior to Euro-American settlement, eastern white pine and red pine associations were generally fire-maintained seral types and existed occasionally as self-perpetuating climax under mixed fire regimes in the Great Lakes area. Fire exclusion can alter plant community structure and composition in these forest types, with shade-tolerant species becoming widespread. These stands may respond well to prescribed burning; however, understory invasion by shade-tolerant species could make burning difficult by developing a layer of less flammable surface material (Duchesne and Hawkes 2000). No information was found regarding nonnative species invasions in these ecosystems.

Pitch pine is well adapted to frequent fire, with presettlement fire regimes characterized by surface fires at intervals less than 10 years where burning by Native Americans was common and mixed-severity fires at intervals of about 10 to 35 years (Wade and others 2000). In the absence of disturbance, pitch pine is replaced by various hardwoods, especially oak and hickory, or by eastern white pine if present. Fire exclusion has also led to conversion of pitch pine forests to black locust-dominated stands (Dooley 2003). Black locust is an early-successional tree that colonizes old fields and burned areas in its native range (Converse 1984b, TNC review), from Pennsylvania southward. It is considered nonnative to the north and east (Fernald 1950). Today prescribed fire is used to reduce fuel loads and maintain or restore fire-adapted vegetation in some pitch pine communities (Patterson and Crary 2004).

Virginia pine and shortleaf pine types are estimated to have relatively frequent presettlement fire-return intervals (~2 to 35 years). Table Mountain pine fire regimes are characterized by stand-replacement fires at intervals of <35 to 200 years (Wade and others 2000). Little information is available regarding invasive species in these forest types. At Manassas National Battlefield Park in northeastern Virginia, some old agricultural fields have succeeded to Virginia pine and support spreading populations of Japanese stiltgrass, Japanese honeysuckle, privet, and winged euonymus (Dibble and Rees 2005). Princesstree occurs in Table Mountain pine-pitch pine forests in the southern Appalachian Mountains (Williams 1998). With the exception of princesstree, there is no information in the literature regarding invasive species and fire in these forest types.

Northeastern spruce-fir forests are characterized by a presettlement fire regime of stand-replacement fires with long return intervals (35 to over 200 years) (Duchesne and Hawkes 2000; Wade and others 2000). Spruce-fir stands are presumed to be less vulnerable than some other vegetation types to encroachment by nonnative invasive plants, though exceptions can occur where seed sources are available. In Maine, spruce-fir stands can have persistent, spreading populations of bush honeysuckles, Norway maple, Japanese barberry, and/or winged euonymus (Dibble and Rees 2005). Loss of dominant trees to nonnative insect pests such as balsam wooly adelgid and hemlock wooly adelgid (Dale and others 1991, 2001) provides openings for establishment of Japanese barberry, Oriental bittersweet, tree-of-heaven, and Japanese stiltgrass (Orwig and Foster 1998).

Role of Fire and Fire Exclusion in Promoting Nonnative Plant Invasions in Coniferous Forests

Few studies in northeastern coniferous forests discuss the establishment or spread of nonnative invasive species following fire or a period of fire exclusion.

Princesstree is widely planted in the eastern United States as an ornamental and a source of high-value export lumber. It is an early successional species that produces large numbers of wind-dispersed seed. It is not shade-tolerant, and seed germination and seedling establishment are restricted to disturbed areas such as exposed mineral soil, where light levels are high and leaf litter is absent (Williams 1993). Princesstree established after wildfire in forests dominated by Table Mountain pine, pitch pine, and Virginia pine in the southern Appalachians (Reilly and others 2006). Managers report postfire establishment of princesstree after "several wildfires" in Great Smoky Mountains National Park (Langdon and Johnson 1994). Invasion of native forests by princesstree is facilitated by other large-scale disturbances such as timber harvest, construction, gypsy moth defoliation, hurricanes, and floods (Johnson, K. 1996, review; Miller 2003; Williams 1993). In debris avalanches following Hurricane Camille in Virginia, princesstree established at densities ranging from 75 to 310 stems/ha on 3 of 4 study sites. Other species that occupied these sites included Japanese honeysuckle and tree-of-heaven, but these species were rarely found in the canopy of a mature forest (Hull and Scott 1982). Viable princesstree seeds have been found in the soil seed bank of some forest communities (Dobberpuhl 1980; Hyatt

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and Casper 2000). This includes the Pine Barrens of southern New Jersey (Matlack and Good 1990), where prescribed fire is sometimes used (Patterson and Crary 2004). Managers should be aware of the possibility of princesstree establishment from the seed bank. Princesstree sprouts from roots and/or stumps after pulling or cutting (Johnson, K. 1996) and may also do so after fire.

Sheep sorrel might be promoted by fire in conifer stands that have developed in agricultural openings or in openings over bedrock. In a stand of white pine on an old field in southeastern Nova Scotia that was clearcut and burned in June, sheep sorrel had a high stem density relative to other vegetation 1 year after fire (Martin 1956). In a 4-year study of a spruce-fir stand in southwestern New Brunswick, Canada, which was clearcut and burned twice, Hall (1955) found that sheep sorrel established immediately after the first burn and persisted after the second, though at a low stem count.

Responses of nonnative buckthorn species to fire may vary depending on frequency of burning. Four years after a low-severity spring burn in white and red pine plantations of Michigan, common and glossy buckthorns less than 0.8 inch (2 cm) DBH were present on plots burned only once but absent from plots burned three times in 5 years. Larger buckthorns (0.8 to 2.3 inches (2.0 to 5.9 cm) DBH) occurred on unburned plots but not on any burned plots, suggesting that the larger size class had been eliminated by fire (Neumann and Dickmann 2001). It is not clear to what extent the two nonnative buckthorn species differed in their response to repeated burning in the pine plantations.

Effects of Nonnative Plant Invasions on Fuel Characteristics and Fire Regimes in Coniferous Forests

There is no documentation that fire regimes have been changed by nonnative invasive species in northeastern coniferous forests, although studies suggest that fuel characteristics may be altered on invaded sites.

A study by Dibble and Rees (2005) suggests that nonnative species have altered fuel characteristics in coniferous forests in southern Maine. Invaded stands support a shrub layer dominated by nonnative honeysuckle species, Japanese barberry, Oriental bittersweet, common buckthorn, and/or glossy buckthorn, and have significantly greater shrub cover and frequency than nearby, relatively uninvaded stands. Fineleaf sheep fescue and wood bluegrass (*Poa nemoralis*, native to Eurasia and recently added to Maine's list of nonnative invasive plants) are abundant in the herb layer and may increase fine fuel loads and continuity on invaded conifer sites (Dibble and Rees 2005).

Black locust has been observed to reduce potential fire spread in pitch pine stands on sandy outwash plains, especially on old farm fields. Black locust litter on the forest floor tends to lie flat and stay relatively damp due to closed-canopy conditions created by black locust clones. The higher live-to-dead fuel ratios and higher fuel moistures effectively slow surface fires compared to uninvaded pitch pine stands (Dooley 2003). Native plants and animals in these fire-dependent plant communities can be adversely impacted by black locust dominance. When black locust encroached in dunes of Indiana, decline in native plant diversity was often accompanied by an increase in nonnative cheatgrass (Peloquin and Hiebert 1999). Larvae of the federally endangered Karner blue butterfly (*Lycaeides melissa sameulis*) of northeastern and upper midwestern North America feed solely on blue lupine (*Lupinus perennis*) in fire-adapted pitch pine woodlands and oak savannas (King 2003; Kleintjes and others 2003), which may be degraded by black locust invasion. Note that the host plant is not bigleaf lupine (*Lupinus polyphyllus*) of the Pacific Northwest, though that species dominates some roadsides and openings in Maine and is listed as a nonnative invasive species in that state.

Use of Fire to Control Nonnative Invasive Plants in Coniferous Forests

Fire is not typically used to control nonnative invasive plants in coniferous forests in the Northeast bioregion where fire suppression is usually the fire management priority. Prescribed fire is sometimes used to maintain fire-adapted ecosystems such as pitch pine, where black locust may be controlled by frequent, severe burning in late spring (Dooley 2003). However, it is difficult to obtain fires of sufficient severity to kill black locust, which typically responds to burning, cutting, and girdling by resprouting and suckering (Converse 1984b).

Grasslands and Early-Successional Old Fields _____

Background

True grasslands in the Northeast bioregion are sparse and discontinuous compared to their counterparts to the west and south. This section includes barrens and sandplains (as described by Curtis 1959 and Dunwiddie and others 1996) and early-successional old fields—areas that were initially cleared for agriculture and are currently maintained in early successional stages (Richburg and others 2004). While material covered here also pertains to disjunct populations of prairie communities extending east as far as Pennsylvania, Kentucky, and

Tennessee (Transeau 1935), discussion of prairie ecosystems is presented more fully in chapter 7.

The grasses and forbs that comprise northeastern grasslands include unique plant assemblages and numerous rare plants and animals (for example, see Dunwiddie 1998; Mitchell and Malecki 2003) and thus are of particular concern to managers. The ecology and much of the fire history of northeastern grasslands are summarized in Vickery and Dunwiddie (1997). Fire, primarily anthropogenic in origin, has been identified as one factor contributing to the origin and persistence of these plant communities (for example, Niering and Dreyer 1987; Parshall and Foster 2002; Patterson and Sassaman 1988; Transeau 1935; Winne 1997). Prescribed fire is currently used in some areas to maintain early successional species. For example, in a sandy outwash plain in southern Maine where native grassland has persisted for more than 900 years (Winne 1997), fire is used to maintain habitat for grasshopper sparrow and a large population of a rare herb, northern blazing star (*Liatris scariosa* var. *novae-angliae*) (Vickery 2002). Xeric blueberry barrens in southeastern Maine have been an open grassland-pine/shrub type for at least 1,700 years; many of these areas are now maintained by burning in alternate years (Winne 1997).

The area of old fields in the Northeast bioregion is extensive. From the early days of the colonial period, forests were converted to pasture and cropland; by the mid-19th century, less than 40 percent of Vermont, Massachusetts, Rhode Island, and Connecticut was forested. Conversion of forest to agriculture resulted in intentional and accidental introduction of many nonnative plants. When agricultural fields were abandoned, seeds of nonnative plants were no doubt present, and other species may have been introduced with "conservation plantings" (Knopf and others 1988). Thus propagules of nonnative invasive plants are likely to be more abundant in old fields than in grasslands of other origins. As agriculture declined in the Northeast during the late 19th and 20th centuries, forests reclaimed much of the landscape (review by Vickery and Dunwiddie 1997), while some areas are maintained in early succession for bird habitat (for example, see Vickery and others 2005) or other conservation purposes.

Grasslands and old fields in the Northeast bioregion are early successional communities capable of supporting woody vegetation. The most problematic invasives are woody species, both native and nonnative, which alter the structure as well as the species composition of these habitats. Techniques for controlling woody invasives in grasslands typically include cutting or mowing, herbicides (Barnes 2004), and/or fire (for example, Dunwiddie 1998). Grassland burns are commonly conducted in fall or spring, though control of woody species may be more effective if burns occur during the growing season (Mitchell and Malecki 2003, Richburg 2005). In the wildland urban interface, early successional vegetation is more commonly maintained by mowing than by burning. Because these habitats require disturbance to remain in an early successional stage, they may be especially vulnerable to establishment and spread of nonnative plants (for example, see Johnson and others 2006).

Nonnative species that seem especially problematic in northeastern grasslands include Scotch broom (*Cytisus scoparius*), multiflora rose, porcelainberry, and swallow-wort (*Cynanchum louiseae, C. rossicum*). Numerous other nonnative species are invasive or potentially invasive in northeastern grasslands (table 5-1), but research on their relationship to fire in grasslands or old fields is lacking. Mehrhoff and others (2003) state that most nonnative species of concern in this bioregion are common in old fields, but many are also problematic in forest or riparian communities, and some of these are discussed in other sections of this chapter.

Role of Fire and Fire Exclusion in Promoting Nonnative Invasives in Grasslands and Old Fields

Nonnative herbs may increase after fire in northeastern grasslands and old fields. Swan (1970) quantified vegetation response to wildfires of the early 1960s in goldenrod-dominated fields in south-central New York. Two years after fire, three nonnative species showed higher relative frequency on burned than unburned areas: Canada bluegrass (81 percent vs. 56 percent), redtop (39 percent vs. 25 percent), and sheep sorrel (44 vs. 33 percent). Burning appears to enhance germination of sheep sorrel, possibly by removing the litter layer (Kitajima and Tilman 1996). However, Dunwiddie (1998) reports no effect of fire on sheep sorrel, and Niering and Dreyer (1989) report equivocal results: In Connecticut old fields dominated by little bluestem, relative frequency of sheep sorrel decreased after 17 years of annual burning but also decreased on unburned plots. Fire alone increased stem density of spotted knapweed for 3 years after spring burning in old fields in Saratoga National Historical Park, Saratoga Springs, New York (Gorman, personal communication 2005).

Nonnative shrubs are likely to survive all but severe, growing-season fires, though information specific to northeastern grasslands is limited. Most research on Scotch broom comes from the Pacific Northwest and is summarized briefly here; this species is covered more thoroughly in chapter 10. Scotch broom spreads from abundant seeds and can sprout from stumps or root crowns following damage to aboveground parts. Scotch

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broom seeds can survive in the soil for at least 5 years and possibly as long as 30 years. Laboratory studies and postfire field observations indicate that heat scarification induces germination (Zouhar 2005a, FEIS review). Prescribed fires in a northeastern grassland (on one site, August cut, then burn; on another, April burn) reduced Scotch broom cover significantly, but numerous seedlings established (Richburg 2005).

Fire can top-kill bush honeysuckle plants and is likely to kill seedlings and stressed plants. However, perennating tissues on roots and root crowns are often protected by soil from fire damage, so there is potential for postfire sprouting. Sprouting of Bell's honeysuckle (*Lonicera × bella*) was observed after spring and late-summer fires at the University of Wisconsin Arboretum (Munger 2005a). After spring burning in an oak forest, sprouts of Bell's honeysuckle were described as "not very vigorous" (Kline and McClintock 1994).

Multiflora rose is a shrub that can reproduce by root suckering and layering, and forms dense thickets that displace herbaceous plants, especially in early successional habitats such as old fields. Both multiflora rose and common buckthorn produced sprouts following stem removal by cutting, with or without burning, in the dormant and growing season (Richburg 2005). Native rose species are typically top-killed by fire; with increasing fire severity, they may be subject to root crown and rhizome damage (Munger 2002c, FEIS review). Thus it seems likely that multiflora rose will survive fire and, because seeds remain viable in the soil for 10 to 20 years (Szafoni 1991), possibly regenerate from seed after fire.

Several nonnative vines are invasive in northeastern grasslands; however, information on responses of nonnative vines to fire in these communities is lacking. We could find no peer-reviewed accounts of invasion or spread of porcelainberry or Oriental bittersweet after fire. However, the ability of these vines to regenerate vegetatively, produce abundant seed (table 5-2), and establish in openings suggests that fire may favor their spread. Black swallow-wort cover increased after dormant season cutting and burning of invasive shrubs in a New York grassland (Richburg 2005), although significance of differences from untreated sites was not reported. Lawlor (2002, TNC review) reports that swallow-wort recovered and reproduced the season following prescribed fires in New York and Wisconsin (see "Use of Fire to Control Nonnative Invasive Plants in Grasslands and Old Fields," page 81).

In the absence of fire or other disturbance (for example, mowing), woody species generally increase in northeastern grasslands. Two studies illustrate how the presence and abundance of nonnative species in old fields may change over time without disturbance. In New Jersey, 40 years of vegetation data from old fields were used to evaluate changes in nonnative species abundance and diversity over time. Invasions were initially severe, with nonnative species comprising over 50 percent of the cover and species in each field. After 20 or more years of abandonment, the abundance and richness of nonnative species had declined significantly without management intervention. As woody cover increased, many nonnative herbaceous species that had dominated earlier in succession, particularly annuals and biennials, became much less abundant. Some shade-tolerant invasive species (garlic mustard, bush honeysuckles, Norway maple, Japanese stiltgrass, and Japanese barberry) are currently increasing on these sites and may present the next invasion challenge to the managers of the grassland community (Meiners and others 2002). A site in southeastern Connecticut that was abandoned and burned 40 years earlier became partially dominated by Oriental bittersweet, which increased in cover along with Japanese honeysuckle, Morrow's honeysuckle, and multiflora rose during the last decades of the study (Fike and Niering 1999).

Effects of Nonnative Plant Invasions on Fuel Characteristics and Fire Regimes in Grasslands and Old Fields

Presettlement and even postsettlement fire regimes for northeastern grasslands are not well described in the literature (but see Vickery and Dunwiddie 1997), so departure of current patterns from past fire regimes is difficult to determine, and the influence of nonnative species on fire regime changes is difficult to estimate. Old fields have no reference fire regime because they are a recent anthropogenic vegetation type. It is more fruitful, in this section, to discuss how nonnative species may alter fuels in northeastern grasslands and thereby influence the fire regimes desired for maintenance of these plant communities or for protecting property in the wildland urban interface. Scotch broom is the only nonnative species for which information on fuel characterisitcs and fire behavior in northeastern grasslands has been published (Richburg 2005).

Scotch broom establishes in old fields and grasslands, where it can eventually replace native plants with a dense, monospecific stand. As Scotch broom stands age, the ratio of woody to green material increases and dead wood accumulates (Waloff and Richards 1977). During experimental fires intended to control this species in old fields on Naushon Island, Massachusetts, where the effects of cutting and burning were studied, Scotch broom was observed to be highly flammable, even when green (Richburg 2005; Richburg and others 2004). Cutting reduced non-woody fine fuels and increased 1-hour and 10-hour woody fuels. Fuel bed depth did not change. Subsequent burns showed flame lengths of approximately 20 feet (6 m) on uncut plots burned with a headfire in April, and 3 feet (1 m) on

cut plots burned in August with a backfire under "very dry conditions."

Due to their growth form and habits, nonnative vines can affect fuel load and distribution, as described above in the section on deciduous and mixed forests. Porcelainberry vines, for example, can cover the ground in sunny openings, such as old fields, and grow up into trees and shrubs at the forest edge. These vines can eventually kill the supporting vegetation (Yost and others 1991), and festooned trees are also susceptible to wind damage, further increasing mortality of supporting species. Similarly, pale swallow-wort and black swallow-wort, which are viny and twining herbs (Gleason and Cronquist 1991), can form large, monospecific stands in open areas and can over-top and smother shrubs (Lawlor 2002).

Use of Fire to Control Nonnative Invasive Plants in Grasslands and Old Fields

The nonnative shrubs and vines that are most problematic in northeastern grasslands are all able to sprout after fire, at least to some extent, and several persist in the soil seed bank (table 5-2). While fire may be a desirable tool for promoting desired grasses and forbs in grasslands, its effectiveness may be confounded by the ability of nonnative species to survive and thrive after fire. Research in grasslands in south-central New York and Naushon Island, Massachusetts, indicated that a single dormant-season burn is unlikely to reduce nonnative woody species. Combining growing-season prescribed fire with other treatments may improve control. A treatment that reduced common buckthorn and Scotch broom in grasslands consisted of a late spring mowing, allowing cut fuels to cure, and then a late summer burn. Growth rate of common buckthorn sprouts was slower on August-burned plots than on unburned plots or spring-burned plots. Effects may be short-lived, however. Nonstructural carbohydrates in common buckthorn and multiflora rose declined after cutting, mowing, or burning treatments but recovered within 1 year (Richburg 2005).

Scotch broom may be susceptible to heat damage from fire, but regeneration from the seed bank complicates the use of fire to control this species. Several researchers provide evidence that Scotch broom seed germination is stimulated by fire (Zouhar 2005a), although results vary among locations (for example, see Parker 2001). Chapters 4 and 10 cover this species in some detail, but one study is relevant here. In old fields being maintained as grasslands on Naushon Island, Massachusetts, Richburg (2005) found that prescribed fires, whether in the dormant season or growing season, killed Scotch broom but led to copious recruitment of Scotch broom germinants from the soil seed bank and/or from nearby untreated plants. Cover of native graminoids and herbs was low

within Scotch broom patches and decreased within a year after burning. When prescribed fire is used to stimulate Scotch broom germination from the seed bank, follow-up treatments such as subsequent controlled burns, spot burning, revegetation with fast growing native species, herbicide treatments, grazing, and hand-pulling can be used to kill seedlings and thus reduce the seed bank (Zouhar 2005a).

Multiflora rose seems able to survive fire but does not usually increase immediately after burning. Thus repeated fires may be useful in controlling this species. In a savanna restoration project on an old agricultural field in Illinois, Hruska and Ebinger (1995) significantly reduced stem density of multiflora rose and autumn-olive following March fires in 2 successive years. They were concerned that desired native oak seedlings were adversely impacted. In plant communities comprised of fire-adapted grasses and forbs, periodic prescribed burns will likely retard multiflora rose invasion and establishment (Munger 2002c). The Virginia Department of Conservation and Recreation (2002d) recommends spring prescribed fire to reduce cover of multiflora rose, with follow-up burns in subsequent years for severe infestations.

Bush honeysuckle species (Morrow's honeysuckle, Bell's honeysuckle, and others) may be controlled with prescribed fire in fire-adapted grassland or old-field communities. According to several sources, spring prescribed burning may kill bush honeysuckle seedlings and top-kill larger plants, although results have been mixed (Munger 2005a). Morrow's honeysuckle was not reduced by dormant-season prescribed fire in old fields of western New York, but growing-season fires preceded by growing-season mowing reduced this species (Mitchell and Malecki 2003). The Maine Department of Conservation, Natural Areas Program (2004) recommends burning during the growing season. Regardless of season, a single prescribed fire is usually not sufficient to eradicate bush honeysuckles. Annual or biennial burns may be needed for several years (Munger 2005a). Solecki (1997, review) recommends annual or biennial spring burning for 5 or more years to control bush honeysuckles in prairie ecosystems.

Fire may be useful for controlling Japanese honeysuckle in grasslands, but only with repeated use and long-term commitment to monitoring and follow-up treatments. Cessation of prescribed fire treatments, even after multiple consecutive or near-consecutive years of burning, often leads to reinvasion. Following spring burns in 4 out of 5 years, fire was excluded from a southern Illinois barren. Japanese honeysuckle frequency decreased following the fires. However, shade increased during fire exclusion years, and 11 years after the last fire, frequency of Japanese honeysuckle was nearly four times preburn levels (Schwegman and Anderson 1986).

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Numerous sources agree that fire is not effective for reducing swallow-wort populations (DiTommaso and others 2005; Lawlor 2000, 2002; Sheeley 1992). Perennating buds on the root crowns generally occur a centimeter or more below the soil surface and are thus likely to be protected from fire (Sheeley 1992; Lawlor 2000, 2002). At Montezuma National Wildlife Refuge, western New York, a large swallow-wort infested area was burned in late spring to reduce woody debris in grasslands (Lawlor 2002). The swallow-worts recovered and reproduced as usual the following season. Similar results were observed after prescribed burning in Wisconsin. Lawlor (2002) suggests that burning or flaming could be used to control seedlings after mature growth has been killed with herbicides; swallow-wort seedlings lack the well-developed root crown of more mature plants.

Riparian and Wetland Communities _____

Background

Riparian and wetland communities in the Northeast bioregion vary in native plant community composition, site characteristics, and fire regime; however, several species of nonnative invasive plants are common among these community types, so they are discussed together here.

Riparian plant communities in the Northeast bioregion may be dominated by hardwoods, conifers, or mixed stands, and a dense layer of shrubs and vines can occur beneath the tree canopy. Many native plants grow almost exclusively in riparian areas and may be adapted to intense disturbance from seasonal flooding and scour by water and ice. Disturbance by fire is unusual in Northeastern riparian communities, so riparian plants may not be fire-adapted; however, adaptations that allow these plants to recover after flooding and scour could aid in their recovery after fire (chapter 2).

The bottomland hardwood vegetation type described by Wade and others (2000) includes the elm-ash-cottonwood ecosystem (*sensu* Garrison and others 1977) that occurs in riparian areas along major streams or scattered swamp areas throughout the eastern United States and includes several forest cover types. The historical role of fire in these ecosystems is unclear, although many of the dominant riparian species are sensitive to fire and especially intolerant of repeated burning. Presettlement fire regimes were thought to be of mixed-severity or stand-replacement types, with intervals of about 35 to 200 years. Fuel loads were generally low due to rapid decomposition, so large, severe fires probably occurred only during extended drought or in heavy fuels caused by damaging wind storms (Wade and others 2000). Conditions in spring and fall are often too wet for prescribed burning.

Freshwater wetlands in the Northeast bioregion include forested wetlands such as red maple swamps, silver maple (*Acer saccharinum*) floodplain forests, alder thickets, conifer bogs, Atlantic white cedar (*Chamaecyparis thyoides*), black gum (*Nyssa sylvatica*), and bay forests; and also fens and marshes dominated by sedges and grasses (Garrison and others 1977). Forested wetlands such as conifer bogs are probably only susceptible to fire in severe drought years due to their typically humid environment. Ground fires are possible with severe drought or drainage; with strong winds, conifer bogs can sustain crown fires. Presettlement, stand-replacement fire intervals are estimated between 35 and 200 years (Duchesne and Hawkes 2000). Prescribed fire is probably not appropriate in forested wetlands, and fire is typically excluded from these communities.

Wet grasslands in the Northeast bioregion include freshwater and salt or brackish tidal wetlands along the Atlantic coast, as well as freshwater inland marshes. Frost (1995) provides information on dominant vegetation along gradients of salinity and fire frequency. Consistent differences in species composition and fire behavior occur between saltwater and freshwater wetlands. Freshwater wetlands support a high diversity of species and a variety of plant associations. These are typically dominated by herbaceous species but may also support woody associations, although woody plant development is impeded by factors including ice scour, wave action, and periodic fires. Saltwater wetlands of the Northeast include the northern cordgrass prairie described by Küchler (1964), which is dominated by cordgrasses (*Spartina* spp.), saltgrass (*Distichlis* spp.), and rushes (*Juncus* spp.) (Wade and others 2000). Sedges (especially *Scirpus,* but also *Schoenoplectus* and *Bolboschoenus*) are also common. Forbs may be present where fresh water mixes into the system. Woody plants are typically intolerant of the salinity and the twice-daily inundation that characterize tidal wetlands, but they may occupy hummocks or outcrops. Since presettlement times, the assumption is that woody plants have extended into the marsh, vegetation is taller, and native plants have been displaced by tall, dense stands of common reed (*Phragmites australis*) (LANDFIRE Rapid Assessment 2005b).

Most information on fire regime characteristics in herbaceous wetlands comes from the southeastern United States (for example, Frost 1995), with relatively little information on wetlands in the Northeast bioregion. Fires are common in southeastern wetlands, which support large quantities of flammable, herbaceous vegetation that is well-adapted to frequent fires. Occurrence of woody plants can alter fire behavior, and groundwater levels influence both fire behavior

and fire effects on soils and vegetation. In freshwater marshes, flammability varies due to the large diversity of plant communities, but species such as sawgrass (*Cladium*), cattail (*Typha*), common reed, maidencane (*Amphicarpum purshii, Panicum hemitomon*), and switchgrass (*Panicum virgatum*) provide flammable fuels that can support continuous, intense fires. Cordgrass species that dominate tidal marshes are also quite flammable. Presettlement fire frequency for northern cordgrass prairie communities is estimated at 1- to 3-year intervals (Wade and others 2000).

Prescribed fire is used more extensively in salt marshes than in freshwater marshes. Conditions in spring and fall are often too wet for prescribed burning in freshwater wetlands, although fire is sometimes used to reduce fuel loads, control invasive plants, and promote native species. Prescribed fire is frequently used in saltwater grasslands to enhance productivity and to reduce plant cover, fuel loadings, and woody species (Wade and others 2000).

Few nonnative invasive plants pose a high threat potential in tidal wetlands (table 5-1). Along margins and in areas where tidal influence or salinities have been altered by land use and development, woody species such as tree-of-heaven (Kiviat 2004) and Oriental bittersweet (Bean and McClellan 1997, review) may be invasive. Common reed, a large, perennial, rhizomatous grass with nearly worldwide distribution, is the invasive species of most concern in tidal wetlands (for example, see Leck and Leck 2005; Niering 1992; Weis and Weis 2003) and also invades fresh wetlands and riparian areas in the Northeast bioregion (table 5-1). Literature reviews

(for example, D'Antonio and Meyerson 2002; Marks and others 1993, TNC review) suggest that, although common reed is native to North America, invasive strains may have been introduced from other parts of the world; and while there is evidence that common reed is native in the Northeast bioregion, many marshes are occupied by a European genotype (Saltonstall 2003). Common reed is regarded as aggressive and undesirable in parts of the eastern United States, but it may also be a stable component of a wetland community that poses little or no threat in areas where the habitat is undisturbed. Examples of areas with stable, native populations of common reed include sea-level fens in Delaware and Virginia and along Mattagodus Stream in Maine. In areas where common reed is invasive, large monospecific stands may negatively impact native plant diversity and create a fire hazard (D'Antonio and Meyerson 2002; Marks and others 1993).

Riparian areas often support more invasive species than upland habitats (for example, see Barton and others 2004; Brown and Peet 2003). This is attributed to high levels of propagule pressure (that is, abundance of seeds or vegetative fragments), a high-frequency disturbance regime, and water dispersal of propagules (Barton and others 2004; Robertson and others 1994). Several nonnative invasive plant species occur in fresh wetlands and/or riparian areas in the Northeast bioregion (table 5-1), including widespread species such as Japanese stiltgrass, garlic mustard, tree-of-heaven, Norway maple, Japanese barberry, bush honeysuckles, privets, multiflora rose, common buckthorn, Oriental bittersweet, and ground-ivy, which are covered in more detail in other sections of this chapter. Species that may be common in old fields and other areas of anthropogenic disturbance, such as porcelainberry and swallow-worts (covered in the "Grasslands and Early-Successional Old Fields" section, page 78), can be invasive along rivers and streams where scouring spring floods occur. Swallow-worts, for example, occur in areas subject to hydrologic extremes such as alvar communities of the eastern Lake Ontario region or New England coastal areas (Lawlor 2002). Species that seem to have a particular affinity for wetland and riparian communities include common reed, reed canarygrass, purple loosestrife (fig. 5-5), glossy buckthorn, Japanese knotweed (*Polygonum cuspidatum*), and mile-a-minute.

Figure 5-5—Reed canarygrass and purple loosestrife grow in dense patches on the typically rocky shore of the Penobscot River (Eddington, Maine). (Photo by Alison C. Dibble.)

Role of Fire and Fire Exclusion in Promoting Nonnative Invasives in Riparian and Wetland Communities

There is little published literature on the role of fire or fire exclusion in promoting plant invasions in riparian communities or wetlands in the Northeast bioregion. However, managers should be alert to the possibility of invasion by nonnative species after wild or prescribed fires, and the possibility that wetland areas adapted to frequent fires could be invaded in the absence of fire by nonnative woody species such as glossy and common buckthorn (Moran 1981).

Studies in the north-central United States and adjacent Manitoba, Canada, indicate that common reed is not typically damaged by fire because it has deeply buried rhizomes that are often under water, and the heat from most fires does not penetrate deeply enough into the soil to injure them. When fire consumes the aboveground foliage of common reed, new top growth is initiated from the surviving rhizomes. Rhizomes may be damaged by severe fire when the soil is dry and humidity low (Uchytil 1992b, FEIS review). Fires of this severity are likely to occur only under conditions of artificial drainage and/or severe drought.

Reed canarygrass is a cool-season, rhizomatous grass that can form dense, monotypic stands in marshes, wet prairies, wet meadows, fens, stream banks, and swales (Hutchison 1992b) (fig. 5-6). It is native to North America and also to temperate regions of Europe and Asia (Rosburg 2001; Solecki 1997). In the United States, cultivars of the Eurasian ecotype have been developed for increased vigor and thus may be more invasive than native ecotypes (Wisconsin Department of Natural Resources 2004). Reed canarygrass is considered a threat to native wetlands because of its rapid early growth, cold hardiness, and ability to exclude desired native plants (Hutchison 1992b; Lyons 1998, TNC review; Wisconsin Department of Natural Resources 2004). Anthropogenic disturbance and alteration of water levels encourage its spread (Wisconsin Department of Natural Resources 2004). Reed canarygrass seems well adapted to survive and reproduce after fire, but its response to wildfire has not been described in the literature. Burning of a Minnesota wetland followed by repeated herbicide application led to extensive germination of reed canarygrass from the seed bank, probably because of increased light at the soil surface (Preuninger and Umbanhowar 1994).

Glossy buckthorn is similar to common buckthorn (see table 5-2) in its reproductive biology and sometimes invades similar woodland habitats, but it more commonly invades moist to wet sites that are not fully flooded (Andreas and Knoop 1992; Frappier and others 2003; Taft and Solecki 1990). Reviews indicate that it grows best in drier parts of wetlands, in wetlands where some drainage has occurred, and possibly where fires have been excluded (Converse 1984a; Larson and Stearns 1990). Glossy buckthorn recruitment is most successful with ample light and exposed mineral soil. Burning to maintain vigor of the native plant community may prevent glossy buckthorn seedling establishment; however, if seed sources occur near burned areas, seedlings can establish readily on exposed soils. Glossy buckthorn also sprouts from roots or the root crown after fire (Catling and others 2001; Post and others 1990). In a calcareous fen in Michigan burned in the fall, glossy buckthorn stem density was twice as great the summer after burning as the summer before burning, and stems were one-third the height of preburn stems (unpublished report cited in review by Converse 1984a). On a prairie site in northwest Indiana, prescribed fire in October resulted in complete top-kill of glossy buckthorn, yet 1 year after fire there was a 48 percent increase in total stems of glossy buckthorn. The site was burned again the following April and sampled the following September with similar results. Overall stem numbers increased 59 percent. The authors suggest that prescribed burning may be used to prevent seed set but that plants will resprout (Post and others 1990).

Purple loosestrife is one of the most invasive species of freshwater wetlands and riparian areas in North America. It is an herbaceous perennial forb with buds that overwinter on the root crown about 0.8 inches (2 cm) below the soil surface (DiTomaso and Healy 2003). Surface fires are unlikely to provide enough heat or burn long enough to cause substantial damage to roots or the root crown of purple loosestrife (Munger

Figure 5-6—Reed canarygrass quickly filled this low conifer forest when the hydrology changed and the overstory died. The forest had previously contained only a sparse understory layer. (Photo by Alison C. Dibble.)

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2002d, FEIS review; Thompson and others 1987). Information describing interactions between purple loosestrife and fire are lacking, although it is likely that purple loosestrife can survive fire by sprouting from buds located below the soil surface. Fire may also lead to recruitment of purple loosestrife seedlings due to exposure of bare substrate containing a substantial seed bank (Munger 2002d).

Two members of the buckwheat family (Polygonaceae), Japanese knotweed and mile-a-minute, are especially invasive in riparian and freshwater wetland communities. Japanese knotweed is an herbaceous perennial that is widely distributed in much of the eastern United States, where it spreads primarily along river banks (fig. 5-7) but also occurs in wetlands, along roadways, and in other disturbed areas (Seiger 1991, TNC review). Japanese knotweed reproduces from seed (Bram and McNair 2004; Forman and Kesseli 2003) and perennial rhizomes that can extend 18 inches (46 cm) below ground, are 50 to 65 feet (15 to 20 m) long, and can survive repeated control attempts. It can also establish from rhizome and stem fragments. Once established, Japanese knotweed spreads via rhizomes

Figure 5-7—Japanese knotweed spread from a nearby house site to occupy at least 50 m along both sides of this stream in Blue Hill Falls, Maine. (Photo by Alison C. Dibble.)

to form virtual monocultures (Child and Wade 2000) that are extremely persistent and difficult to control (Seiger 1991). Given its extensive root system and its response to repeated cutting, it seems likely to survive even frequent, severe fire, though no peer-reviewed reports are available on this topic.

Mile-a-minute is a prickly, annual, scrambling vine that is especially prevalent along roadsides, ditches, stream banks, wet meadows, and recently harvested forest sites (Virginia Department of Conservation and Recreation 2002c). A review by Kumar and DiTommaso (2005) indicates that mile-a-minute grows best in sunny locations on damp soil but can also tolerate light shade. The prickly stem and leaves allow it to climb over neighboring vegetation and to form dense, tangled mats that cover small trees and shrubs to a height of about 26 ft (8 m) along forest edges. Mile-a-minute reproduces by seed that is dispersed by birds and mammals, as well as by water transport. Seeds can remain dormant in the soil seed bank for at least 3 years (Kumar and DiTommaso 2005). Since mile-a-minute thrives in gaps and disturbed areas and its seed is widely dispersed by birds (Okay 2005, review), fire could contribute to its increase; however, this has not been documented.

Effects of Nonnative Plant Invasions on Fuel Characteristics and Fire Regimes in Riparian and Wetland Communities

We found no studies that specifically address changes in fuel characteristics and fire behavior in riparian or wetland communities in the Northeast bioregion. Existing reports do not indicate that nonnative plant invasions have altered the fire regimes in these communities. The discussion of fuel properties here is based on morphology and phenology of nonnative invasives.

Common reed is perceived as a fire hazard where it occurs in dense stands in wetlands. It produces substantial amounts of aboveground biomass each year, and dead canes remain standing for 3 to 4 years (Thompson and Shay 1985). It has been suggested that common reed colonies increase the potential for marsh fires during the winter when aboveground portions of the plant die and dry out (Reimer 1973). Thompson and Shay (1989) observed that, even when common reed stands are green, the typically abundant litter allows fires to burn. Additionally, head fires in common reed stands may provide firebrands that ignite spot fires more than 100 feet (30 m) away (Beall 1984, as cited by Marks and others 1993).

Glossy buckthorn branches profusely from the base, with dead stems often found among smaller, live stems (Taft and Solecki 1990). Herbaceous fuels are usually sparse beneath large glossy buckthorn shrubs

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or in dense thickets (Packard 1988). Where glossy buckthorn invasion has reduced fine herbaceous fuels and increased dead woody fuels, fire behavior may be altered.

Purple loosestrife is difficult to burn, based on reports from managers who attempted to use prescribed fire to control it (Munger 2002d). Such attempts are commonly described as being confounded by moist soil conditions and patchy fuel distribution. A persistent stand of purple loosestrife could alter fuel conditions and fire behavior if it displaces native vegetation that is more flammable, and could thus further alter plant community composition. There is, however, currently no empirical evidence of such effects from purple loosestrife invasion.

Use of Fire to Control Nonnative Invasive Plants in Riparian and Wetland Communities

Prescribed fire is not likely to be a useful control measure for invasive species in plant communities where fires are typically rare and native species are not fire-adapted. Many forested wetlands, for example, are typically too wet to burn except during drought. Conversely, herbaceous wetlands commonly support native species that are adapted to frequent fire (Frost 1995; Wade and others 2000), and prescribed fire may be useful for controlling nonnative invasives in these communities.

Prescribed fire and herbicides are often used, alone and in combination, to manage common reed in wetlands in the Northeast bioregion and adjacent Canada. For example, fire has been used to reduce common reed in marshes at Prime Hook National Wildlife Refuge near residential areas of Delaware (Vickers 2003). However, little published quantitative information is available regarding the efficacy of fire. Prescribed burning alone removes accumulated litter and results in a temporary decrease in aboveground biomass of common reed, but fire does not kill plants unless rhizomes are burned and killed. This seldom occurs because the rhizomes are usually covered by a layer of soil, mud, and/or water (Marks and others 1993).

Season of burning may influence postfire response of common reed. Researchers in Europe found that burning common reed in winter caused little damage, while burning during the emergence period killed the majority of common reed shoots (Toorn and Mook 1982). Spring burning at the Delta Marsh in Manitoba removes litter and promotes a dense stand of even-aged canes, whereas summer burning results in stunted shoots and may control vegetative spread (Thompson and Shay 1985; Ward 1968). Both spring and fall burning of common reed resulted in greater shoot biomass, and summer burns resulted in lower shoot biomass in comparison with controls, while total shoot density on all burned plots was higher on controls. Similarly, belowground production in common reed was higher by mid-September on spring and fall burns than on controls, but was not higher on summer burns. Summer burns resulted in increased species diversity, richness, and evenness, while these community characteristics were not altered by spring and fall burns (Thompson and Shay 1985).

Burning is sometimes used in conjunction with herbicide treatments and manipulation of water levels to control common reed. Clark (1998) found that herbicide applied late in the growing season, followed by dormant season prescribed fire and a second herbicide application the following growing season, was more effective than spraying alone. A significant decrease in density and frequency of common reed was recorded in spray-burn treatments compared to pretreatment measures, untreated controls, and spray only treatments (Clark 1998). At Wertheim National Wildlife Refuge in New York, common reed was eliminated from a freshwater impoundment that was drained in the fall, burned the following winter, and then reflooded. Common reed remained absent for at least 3 years following treatment (Parris, personal communication cited by Marks and others 1993). The same TNC review presents several additional case studies documenting attempts to control common reed using prescribed fire.

Reed canarygrass is difficult to control because it has vigorous, rapidly spreading rhizomes and forms a large seed bank (Hoffman and Kearns 2004, Leck and Leck 2005); in addition, control efforts could reduce native ecotypes of this species or harm other native species (Lavergne and Molofsky 2006, review; Lyons 1998). Effects of prescribed fire on reed canarygrass vary. Moist bottomlands in Wisconsin undergoing restoration from agriculture to tallgrass prairie were burned on a 3-year rotation—one group of plots in late March and another group in mid-July. Neither frequency nor cover of reed canarygrass changed significantly in any of the treatments (no burn, spring burn, summer burn) (Howe 1994b). A review by Apfelbaum and Sams (1987) included an account of burning of wet prairie in Illinois every 2 to 3 years. This treatment appeared to restrict reed canarygrass to disturbed sites and prevent spread into undisturbed wetland. The effects of burning reed canarygrass at different seasons have not been studied for wetlands. Hutchison's (1992b) management guidelines suggest that late spring or late autumn burning for 5 to 6 consecutive years may produce "good control" of reed canarygrass in wetlands, but that treatment will be ineffective unless desired species are present or seeded in. Prescribed fires may be difficult to conduct in stands dominated by reed canarygrass because of high water levels and vegetation greenness. Management guidelines from

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the Wisconsin Department of Natural Resources (2004) suggest that treatment with glyphosate could make fall burning more feasible. Because reed canarygrass alters water circulation, increases sedimentation, and may increase the uniformity of wetland microtopography (Zedler and Kercher 2004), restoration of invaded ecosystems is likely to require restoration of physical structure of the habitat and seeding.

The literature on glossy buckthorn, largely anecdotal, suggests that prescribed fire may be used to control this species, especially in communities adapted to frequent fire (for example, Heidorn 1991; Larson and Stearns 1990). According to a management guideline by Heidorn (1991), regular prescribed fire (annual or biennial burns for 5 or 6 years or more) can control both glossy and common buckthorns in communities adapted to frequent fire such as fens, sedge meadows, and marshes. A review by Converse (1984a, TNC review) suggests glossy buckthorn can be reduced by cutting in the spring at leaf expansion and again in the fall, followed by spring burning the next 2 years. Postfire sprouts of glossy buckthorn may be more susceptibile to herbicides (Converse 1984a) or other control measures. McGowan-Stinski (2006, review) indicates that the season after mature buckthorn shrubs have been removed from an area, large numbers of seedlings are likely to germinate; in addition, untreated saplings and/or resprouts are likely to occur. He suggests controlling seedlings, saplings and sprouts by burning them with a propane torch in the first growing season after removal of adults. It is most efficient to torch seedlings and saplings at the stem base until wilting occurs. Repeat treatment could be needed. Seedlings are usually not capable of resprouting if torched before August (McGowan-Stinski 2006).

In a forested swamp dominated by white ash and red maple in the Berkshire Hills of western Massachusetts, Morrow's honeysuckle dominated a dense shrub understory, and Japanese barberry was a common associate. Richburg (2005) compared treatments to reduce the ability of these species to store root carbohydrates. A growing season cut followed by fall burning and a cut the next year had the greatest effect on reducing nonstructural carbohydrates for Morrow's honeysuckle. For Japanese barberry, this treatment and the dormant-season cut led to the lowest root carbohydrate levels, which was interpreted as a decrease in plant vigor. Plots cut in the dormant season had taller sprouts and greater growth rates by late summer 2003 than plots treated during the growing season.

For three additional species, use of fire as a control method is ineffective or not well-known. The use of fire as a control measure for purple loosestrife has been largely dismissed as ineffective. Attempts to burn residual biomass following cutting or herbicide treatments may merely result in recruitment of purple loosestrife seedlings where burning exposes soil containing a substantial seed bank (Munger 2002d). Burning Japanese knotweed when it is actively growing is not recommended as an effective control method according to a control manual published in England (Child and Wade 2000). There is no information in the literature regarding the use of fire to control mile-a-minute. Control efforts should focus on eliminating or reducing seed output, especially near waterways, and avoiding disturbance and the creation of gaps in existing vegetation (Okay 2005).

Emerging Issues in the Northeast Bioregion _____

As nonnative invasive plants continue to spread into previously uninvaded areas and managers gain experience with their control, questions and concerns about the relationship of invasive species to fire will also change. Some of the following matters are under active discussion in the region:

Fuel Properties of Invaded Northeastern Plant Communities and Influences on Fire Regimes

Ducey (2003) pointed out the inadequacy of fuels information specific to the Northeast, especially regarding heat content of dead fuels. Currently fuel models must be extrapolated from models developed in western vegetation types. The Photo Series for the Northeast (http://depts.washington.edu/nwfire/dps/) will be an important resource but will not focus on invaded fuel beds. Dibble and others (2007) assessed the relative flammability of native versus nonnative fuels for 42 species, but more research is needed.

Fuel accumulations that may exceed reference conditions in forested areas have resulted from fire exclusion, extensive mortality of dominant tree species, severe weather events such as the region-wide ice storm in January 1998, and encroachment by nonnative invasive plants. In some locations in the Northeast, nonnative invasive grasses form a more continuous fine surface fuel layer than occurred in nearby uninvaded conditions (Dibble and Rees 2005). At other locations, invasive vines have become common and can act as ladder fuels. However, it is not known if these changes in fuel bed characteristics will result in an increase in fire size, frequency, and/or severity.

Vulnerability of Forest Gaps to Invasion

Just as millions of American chestnut trees succumbed in the 1900s to chestnut blight (*Cryphonectria parasitica*), so we are likely to see continued tree

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mortality in the northeastern bioregion associated with insects and diseases. Agents of tree mortality include:

- Hemlock wooly adelgid (*Adelges tsugae*)
- Gypsy moth (*Lymantria dispar*)
- Sudden oak death (*Phytophthora ramorum*)
- White pine blister rust (*Cronartium ribicola*)
- Balsam wooly adelgid (*Adelges piceae*)
- Dutch elm disease (*Ophiostoma ulmi*)
- Spruce budworm (*Choristoneura fumiferana*)
- Asian long-horned beetle (*Anoplophora glabripennis*)
- European wood wasp (*Sirex noctilio*)
- Emerald ash borer (*Agrilus planipennis*)

Of these, only spruce budworm is native to North America. The impact of high tree mortality probably exceeds the impact of fire in promoting invasive plants in this bioregion. Additionally, the effects of salvage operations following insect kill, and timber harvest in general, may introduce and promote invasive species. In salvage operations, log yards and skid trails are often the sites and corridors for new infestations of nonnative invasive plants. Tree-of-heaven has invaded harvested stands in Virginia (Call and Nilsen 2003, 2005) and West Virginia (Marsh and others 2005), and Japanese stiltgrass established after timber harvest in eastern Tennessee (Cole and Weltzin 2004). Openings in infested stands might be invaded by nonnative honeysuckles, Oriental bittersweet, Japanese barberry, tree-of-heaven, invasive grasses, or other nonnative plants. The presence of these species could alter fuelbed structure and possibly biomass and seasonal drying patterns.

Global Climate Change

Population expansions by nonnative plants in the Northeast are likely to be facilitated by a warming climate, which is expected to continue to increase the frequency and intensity of disturbances and thus opportunities for invasion. Recent models (Adger and others 2007; Intergovernmental Panel on Climate Change 2001) indicate that climate change will reduce snowfall and alter streamflow in eastern forests. These effects would be accompanied by greater uncertainty in weather. Ice storms, hurricanes, and episodes of drought are expected to increase in frequency, intensity, or duration. These events may result in more frequent wildfire, accompanied by increases in nonnative invasives favored by fire.

Interactions Between Nonnative Invasive Plants, Fire, and Animals

Information is needed on changes in wildlife nutrition that come about when nonnative plants are burned in the Northeast bioregion. Lyon and others (2000b) reviewed changes in nutritional content of wildlife foods when vegetation is burned, but they focused on native plant communities.

Invertebrate species may affect the relationships between fire and plant communities. In New Jersey hardwoods (oaks, yellow poplar, maple), areas invaded by Japanese barberry and Japanese stiltgrass differed from uninvaded areas not only in plant composition and structure but also in forest floor properties. Invaded areas had higher pH, thinner litter and organic layers (Kourtev and others 1998), and higher nitrate concentrations accompanied by greater nonnative earthworm density (Kourtev and others 1999). Earthworms are an important wildlife food (for example, for American robin, and woodcock) and their abundance could lead to altered behavior and habitat use. Because they consume the litter layer, they may influence the potential for surface fires.

Conclusions

The highly fragmented landscape, proximity of the wildland urban interface, and large number of nonnative species that occur in the Northeast bioregion complicate land management decisions, including fire and fuel management. A relative lack of peer-reviewed literature on the relationships between fire and invasive plants for this bioregion further challenges the manager to make informed decisions. Managers must consider the possibility of nonnative species establishing or spreading after wild or prescribed fire. Ideally, monitoring for invasive species and far-sighted mitigation will be included in their fire management plans. Available information suggests that some nonnative invasive plants have potential to alter fuel characteristics and that these differ from reference conditions (for example, Dibble and Rees 2005).

When planning prescribed fire with the objective of controlling invasives, fire impacts on all species must be considered and efforts made to prescribe a fire or fire regime that will favor native vegetation over invasive plants. Use of fire to control invasives in a plant community where the fire is outside reference conditions could produce undesired effects on the native community. Additional considerations for the use of prescribed fire for controlling invasive plants in the Northeast bioregion include:

1. It is important to prioritize safety and compliance with air quality and other regulations within the wildland-urban interface.
2. Cooperation among adjoining landowners is key.
3. Multiple control methods and repeated treatments are likely to be needed to reduce most invasive plant populations.

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4. Implementation of high-quality, long-term monitoring, archiving of data, and information sharing are essential components of a successful control project.

Resources Useful to Managers in the Northeast Bioregion

Because peer-reviewed literature is limited regarding the relationship between fire and nonnative invasive plants in the Northeast bioregion, information sharing by managers can be especially effective:

- A listserve maintained by the Mid-Altantic Exotic Pest Plant Council enables managers to relate their successes and failures using control treatments, including prescribed burning (www.ma-eppc.org).
- Spread of nonnative invasive plants in six New England states is tracked by county in an online atlas (Invasive Plant Atlas of New England, http://invasives.uconn.edu/ipane/), based on herbarium specimens. Information on weed control is also included (Mehrhoff and others 2003).

- The Virginia Native Plant Society offers fact sheets about nonnative invasive plants at http://www.dcr.state.va.us/dnh/invlist.htm. These fact sheets cover use of prescribed fire as a management tool, though fire effects are rarely noted and few references are given.
- Rapid assessment reference condition models are available for several "potential natural vegetation groups" in the Northeast bioregion through the LANDFIRE website (http://www.landfire.gov). Model descriptions can be downloaded and compared to existing conditions. This can aid in estimating fuel loads and fire regime characteristics that are desirable in restoration projects and hazard fuels management.

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Notes