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Chapter 10:

Fire and Nonnative Invasive Plants in the Northwest Coastal Bioregion

Introduction

This chapter discusses the relationship between fire (natural and prescribed) and nonnative plant species within major vegetation communities of the Northwest Coastal bioregion, and specifically addresses the role of fire in promoting nonnative species invasions, the effects of nonnative species on fire regimes, and usefulness of fire as a management tool for controlling nonnative species. Four plant communities of western Washington and Oregon will be covered: coastal Douglas-fir forests, montane forests and meadows, riparian forests, and Oregon oak woodlands. Three plant communities of Alaska will also be examined: coastal hemlock-spruce forests, boreal forests, and tundra (fig. 10-1). Table 10-1 provides a list of important nonnative species in the Northwest Coastal bioregion and their estimated impact within these plant communities.

Conifer forests dominate much of the landscape of the Northwest Coastal bioregion. In this densely forested environment, fire promotes nonnative species establishment by creating open-canopy conditions for these predominantly shade-intolerant plants and exposing mineral soils for ruderal seedling establishment.

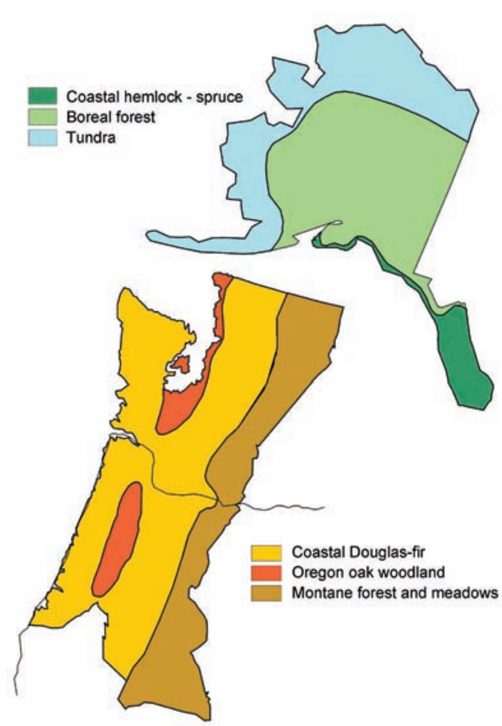

Figure 10-1—Approximate distribution of major plant communities in the Northwest Coastal bioregion. Riparian forests are not shown.

Table 10-1—Nonnative invasive plant species of concern in the Northwest Coastal bioregion and their approximate threat potential in each of the major plant communities covered in this chapter. L= low threat, H = high threat, P = potentially high threat, U = unknown threat, N= not invasive

		Plant communities						
		Pacific Northwest				Alaska		
Scientific name	Common name	Coastal Douglas-fir forests	Montane forests and meadows	Riparian forests	Oak woodlands	Hemlock-spruce forests	Boreal forests	Tundra
Anthoxanthum aristatum	Annual vernal grass	N	N	U	H	N	N	N
Arrhenatherum elatius	Tall oatgrass	N	U	U	H	N	N	N
Brachypodium sylvaticum	False brome	P	U	U	P	N	N	N
Cytisus scoparius	Scotch broom	H	N	L	H	L	N	N
Hedera helix	English ivy	L	N	H	U	N	N	N
Holcus lanatus	Common velvetgrass	P	U	U	P	N	N	N
Hypericum perforatum	Common St. Johnswort	L	U	L	P	L	N	N
Hypochaeris radicata	Hairy catsear	U	L	U	H	L	N	N
Linaria vulgaris	Yellow toadflax	N	N	N	N	P	L	N
Melilotus alba, M. officinale	Sweetclover	N	N	L	L	H	H	N
Phalaris arundinacea	Reed canarygrass	N	U	H	P	L	N	N
Phleum pratense	Timothy	U	U	L	U	L	H	N
Polygonum cuspidatum, P. sachalinense	Knotweed species	N	N	H	N	H	N	N
Rubus discolor, R. lacinatus	Blackberry species	P	U	H	H	N	N	N
Rumex acetosella	Common sheep sorrel	U	U	L	P	L	N	N
Ulex europeus	Gorse	P	N	U	N	L	N	N
Vicia cracca	Bird vetch	N	N	N	P	H	H	N

Throughout this region, nonnative plant species are largely restricted to early seral environments that follow disturbance and do not persist into later stages of forest succession. Persistent populations of nonnative plants tend to be restricted to naturally open environments, such as woodlands and meadows, or locations subject to repeated disturbance, such as riparian corridors or roadsides.

Coastal Douglas-fir Forests

At low to mid elevations west of the Cascade Range in Washington and Oregon, humid, maritime forests composed of evergreen conifers are the dominant vegetation type. Two distinct forest zones are recognized within the greater coastal Douglas-fir region: Sitka spruce (*Picea sitchensis*) and western hemlock (*Tsuga heterophylla*) (Franklin and Dyrness 1973). Sitka spruce zone forests dominate the western edge of this region, forming a narrow band along the coast from southern Oregon north to southeast Alaska. Inland from the coastal margin, western hemlock zone forests are the most extensive vegetation type west of the Cascade Crest in Washington and Oregon. Western hemlock zone forests are dominated by coast Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*) during the predominant subclimax periods, with succession to western hemlock and western redcedar (*Thuja plicata*) only at later-seral (climax) stages. Broadleaf trees are of lesser importance than conifers in both forest types, though red alder (*Alnus rubra*) is abundant in disturbed locations and several hardwood species are common in riparian areas. Together, Sitka spruce and western hemlock zone forests make up the coastal Douglas-fir region (Garrison and others 1977); both forest zones are discussed in this section.

Humid forests of the coastal Douglas-fir region are characterized by infrequent fire (Arno 2000, table 5-1, pg. 98) and remain largely unaffected by fire exclusion policies. In Sitka spruce zone forests, where precipitation averages 80 to 120 inches (200 to 300 cm) per year and low clouds lead to high rates of fog drip, climate limits the ability of wildfires to burn through forest stands. In this zone, wind is the primary disturbance type, and fire intervals are on the order of hundreds to thousands of years (Agee 1993; Arno 2000, pg. 98). However, when fires do occur in coastal forests, they spread over large areas and are typically stand-replacing.

Presettlement fire regimes are highly variable for western hemlock zone forests. In drier regions of the Cascade foothills, mixed-severity fires occurred roughly every 100 years, while high-severity fires occurred about every 130 to 150 years or more; in mesic forests of the Olympic Peninsula, fire-return intervals greater than 750 years are common (Agee 1993). Large, intense, high-severity fires in both Sitka spruce and western hemlock forest zones may be associated with severe fire weather, regardless of fuel conditions (Agee 1997). Low- and moderate-severity fires in western hemlock zone forests, however, are related to a complex combination of weather, seral vegetation stage, and fuel characteristics (Agee 1997; Wetzel and Fonda 2000).

Forests of the coastal Douglas-fir region were extensively harvested during the 20th century. This practice continues to be the case, particularly on private industrial forestlands. As a result, coastal Douglas-fir forests are generally immature with scattered pockets of mature and old-growth stands. Logging has become the dominant disturbance agent, and prescribed fire has been used by forest managers for disposing of logging slash and preparing sites for reforestation.

Role of Fire in Promoting Nonnative Plant Invasions in Coastal Douglas-fir Forests

Nonnative species in the coastal Douglas-fir region are associated with high-light environments and disturbance (DeFerrari and Naiman 1994; Heckman 1999; Parendes and Jones 2000). Therefore, opportunities for nonnative species establishment are created by disturbances that open the forest canopy, such as fire, forest thinning and harvest, and road building and maintenance. Gradually, intense competition from residual native species and regenerating conifers and eventual shading by developing forest stands can eliminate some plants, including nonnative species, from understory plant communities (Halpern and Spies 1995; Oliver and Larson 1996), especially when recently harvested sites are densely planted with Douglas-fir (Schoonmaker and McKee 1988) or other conifers. For example, abundance and cover of nonnative plant species are negatively correlated with time since canopy opening disturbance (successional age) ($r = -0.61$, $P < 0.001$) and canopy density ($r = -0.37$, $P < 0.05$) in clearcut timber harvest units located in Sitka spruce and western hemlock zone forests of the Olympic Peninsula (DeFerrari and Naiman 1994). Nonnative species that establish after fire or other canopy disturbances must originate from the soil seed bank, be transported to the site by logging or fire-suppression equipment, or be dispersed from populations located along nearby roads, in riparian corridors, or in open habitats.

Few studies have been conducted on the effects of fire on nonnative species within the coastal Douglas-fir region, perhaps due to relatively long fire-return intervals. Neiland (1958) conducted a vegetation analysis in the Tillamook Burn more than a decade after a huge stand-replacing fire (1933) and subsequent smaller fires (1939, 1945) burned on the western slope of the Oregon Coast Range. Several nonnative herbaceous species were found in quadrats within the burned area but not in adjacent unburned forest (table 10-2). Since Neiland's study took place several years after the fires, it is not clear whether nonnative species established and spread in response to fire or were introduced by fire suppression and reforestation efforts and persisted and spread due to open-canopy conditions.

The ecological effects of slash and broadcast burns after clearcut timber harvests have been well studied (Dyrness 1973; Halpern 1989; Halpern and Spies 1995; Halpern and others 1997; Kraemer 1977; Lehmkuhl 2002; Morris 1958; Schoonmaker and McKee 1988; Stein 1995; Stewart 1978; West and Chilcote 1968). However, several critical topics remain unaddressed. Although slash fires are typically set in both spring and fall, the effects of season of burn on nonnative species' responses has not been examined. In addition, nonnative plant species' responses to natural fire and slash burning have not been compared, though similarities are assumed. Finally, the effects of mechanical disturbance associated with harvest have not been separated from the effects of burning.

Slash burning on clearcuts promotes the establishment and temporary dominance of both native and nonnative herbaceous species in the coastal Douglas-fir forest region (Kraemer 1977). Though disposing of

Table 10-2—Frequency of nonnative plant species observed in 200, 1-m^2 quadrats located within the perimeter of the Tillamook Burn, Oregon (Neiland 1958).

Scientific name	Common name	Frequency (%)
Hypochaeris radicata	Hairy catsear	25
Senecio vulgaris	Common groundsel	5
Cirsium vulgare	Bull thistle	4
Poa trivialis	Rough bluegrass	1
Elymus repens	Quackgrass	1
Cirsium arvense	Canada thistle	[a]

[a] Species present in area but not in quadrat.

logging slash without the use of fire may reduce the establishment and spread of nonnative invasive species (Lehmkuhl 2002), untreated logging slash increases overall fire danger (Graham and others 1999). Native and, to a lesser extent, nonnative ruderal herbs are the dominant vegetation during the first 4 to 5 years after slash burning; native residual species typically regain dominance after this point (Dyrness 1973; Halpern 1989; Halpern and Spies 1995; Schoonmaker and McKee 1988). Severe slash fires may kill residual native species (Schoonmaker and McKee 1988), thereby increasing the temporal window available to nonnative species to establish, reproduce, develop soil seed banks, and disperse seeds to neighboring locations.

The nonnative winter annual woodland groundsel (*Senecio sylvaticus*) is particularly prominent in slash burns following clearcuts in western hemlock zone forests of the Cascade and Coast Ranges (Dyrness 1973; Halpern 1989; Halpern and others 1997; Kraemer 1977; Morris 1958, 1969; Schoonmaker and McKee 1988; Stewart 1978; West and Chilcote 1968). Woodland groundsel rapidly increases in abundance and cover 1 to 2 years after broadcast burning (fig. 10-2) and then, just as quickly, declines to negligible amounts (Dyrness 1973; Halpern and others 1997; Kraemer 1977; Morris 1958; Schoonmaker and McKee 1988; Stewart 1978; West and Chilcote 1968). Woodland groundsel seed does not survive broadcast burning (Clark 1991); therefore, fast increases in woodland groundsel populations are related to its abundant production of wind-dispersed seed along with its life history traits of fall germination, rapid early growth, and annual lifecycle (Halpern and others 1997). The transient nature of woodland groundsel has been attributed to a high soil fertility requirement (West and Chilcote 1968) that is met by soil conditions associated with recent burns; and to poor competition with perennials for soil nutrients (van Andel and Vera 1977). However, Halpern and others (1997) demonstrated that interspecific competition was not responsible for this pattern and questioned the hypothesis that woodland groundsel populations decline after soil nutrients have been depleted. The timing of harvest and slash burning has a strong effect on the timing of population booms of woodland groundsel (Halpern 1989; Halpern and others 1997), indicating the importance of timing of disturbance to its establishment and initial growth. Fall broadcast burns consume wind-dispersed and buried seed and result in low population densities the following year, though dramatic increases in population densities typically occur during the second growing season (Halpern and others 1997). Similarly, logging activities that occur after seed dispersal in late summer and early fall prevent seedling establishment and also result in low initial population densities the following year (Halpern 1989).

In addition to woodland groundsel, the invasion and short-term abundance of several other nonnative herbs are associated with clearcut timber harvest followed by broadcast burning in coastal Douglas-fir forests. For example, in the Oregon Coast Range, the frequency and abundance of tansy ragwort (*Senecio jacobaea*) is related to the amount of site disturbance after clearcutting of western hemlock and Sitka spruce zone forests, being greatest with site preparation treatments such as broadcast burning and spraying of herbicides (Stein 1995). In areas with persistent seed banks of tansy ragwort—a function of the historic abundance of mature plants—invasions are triggered by localized disturbances that remove existing vegetation (McEvoy and Rudd 1993; McEvoy and others 1993).

Broadcast-burned clearcuts within western hemlock zone forests of the Cascade Range may also be invaded by nonnative bull and Canada thistles (*Cirsium vulgare, C. arvense*) (Dyrness 1973; Halpern 1989; Schoonmaker and McKee 1988), St. Johnswort (*Hypericum perforatum*) (Schoonmaker and McKee 1988), prickly lettuce (*Lactuca serriola*) and wall-lettuce (*Mycelis muralis*) (Dyrness 1973; Schoonmaker and McKee 1988). In a study of secondary succession in the western Oregon Cascades, severely burned (surface litter completely consumed) microsites located within broadcast burned harvest units were colonized and dominated by native fireweed (*Chamerion* spp.), though nonnative bull thistle (Dyrness 1973) and Canada thistle (Halpern 1989) also established in severely burned as well as lightly burned (surface litter charred but not completely removed) microsites. These two thistle species peaked

Figure 10-2—Woodland groundsel dominating a clearcut and broadcast burned site 1 year after disturbance, as is typical for this species in western hemlock forests. (Photo by Vegetation Management Research Cooperative, Oregon State University.)

in relative abundance 3 to 5 years after broadcast burning and then declined (Dyrness 1973; Halpern 1989). Similarly, in a survey of understory vegetation in broadcast burned clearcuts in the western Oregon Cascades (Schoonmaker and McKee 1988), bull and Canada thistles were several times more abundant in stands that had been harvested and burned 5 years previous than in younger or older stands. No statistical analysis was presented. Fire Effects Information System (FEIS) literature reviews indicate that bull and Canada thistles establish after fire in other regions of the Northwest (Zouhar 2001d, 2002b); however, it has not been determined whether establishment occurs from soil seed banks or from wind-dispersed seed. In an experiment examining the effects of heat and soil moisture treatments on bull thistle seeds collected from an old-growth Douglas-fir forest seed bank (Clark 1991; Clark and Wilson 1994), the authors determined that soil temperatures typical of even a low-severity fire could kill bull thistle seed in the soil seed bank. This observation suggests that bull thistle establishment in broadcast burned harvest units is achieved with seeds dispersed from mature plants located in nearby unburned locations.

St. Johnswort also establishes in clearcut and broadcast-burned harvest units of the western Cascade Range, Oregon (Schoonmaker and McKee 1988). Similar to the thistle species previously described, St. Johnswort cover was several times greater in stands that had been harvested and broadcast burned 5 years prior to Schoonmaker and McKee's study than in younger or older harvest units examined. Statistical analysis was not provided. Fire may stimulate sprouting and seed germination of St. Johnswort (Zouhar 2004, FEIS review) and, therefore, may have contributed to its establishment in the broadcast-burned harvest units examined. Alternatively, establishment and spread of St. Johnswort may have been related to soil disturbances associated with mechanical timber harvest and open-canopy conditions.

Finally, broadcast-burned clearcuts in western hemlock zone forests of the Cascade Range are frequently invaded by prickly lettuce (Dyrness 1973) and wall-lettuce (Schoonmaker and McKee 1988). Within two broadcast-burned clearcuts in the western Oregon Cascades, microsites with low-severity burns were invaded by prickly lettuce, while severely burned microsites were not (Dyrness 1973). In another western Oregon Cascade Range study examining the understory composition of clearcut and broadcast burned Douglas-fir stands, Schoonmaker and McKee (1988) observed that wall-lettuce was several hundred times more abundant in stands that had been clearcut and broadcast burned 5 years prior to the study than in younger or older stands. Again, statistical analysis was not presented. The influence of broadcast burning, mechanical disturbance, or open-canopy conditions on the establishment and spread of these species was not explored.

Seeding of nonnative grasses and legumes in slash-burned clearcuts (Lehmkuhl 2002), after wildfires (Agee 1993; Beyers 2004), and along forest roads (Dyrness 1975) and firelines (Beyers 2004) has been widely practiced in coastal Douglas-fir forests in order to increase herbaceous forage for ungulate populations, reduce browsing on conifer seedlings, suppress undesirable species, and reduce soil erosion (Beyers 2004; Lehmkuhl 2002). Seeding of nonnative herbaceous species may influence the establishment and growth of nonnative invasive species and/or alter successional development of native plant communities. In an experiment examining the ecological effects of spring broadcast burning coupled with seeding of common nonnative forage species (orchard grass (*Dactylis glomerata*), white clover (*Trifolium repens*), perennial ryegrass (*Lolium perenne*), annual ryegrass (*L. perenne* ssp. *multiflorum*), and birdsfoot trefoil (*Lotus corniculatus*)) after clearcut timber harvest in the coastal forests of western Washington, the total number and cover of nonnative species were significantly greater in burned plots than in unburned plots ($P = 0.018$ and $P = 0.094$, respectively), suggesting that broadcast burning contributed to the establishment of nonnative species (Lehmkuhl 2002). However, since both seeded forage species and invasive species were included in nonnative species counts and assessments, it is not clear whether burning encouraged the establishment of forage species, invasive species, or both. In addition, the number of nonnative species in burned plots, both forage and invasive, increased during the first 3 years after site treatment ($P = 0.054$), while the number of nonnatives in unburned plots did not change significantly over the same time period. Seeding of nonnative forage species had no effect on the total number of nonnative species observed in burned and unburned plots even though introduced forage species were included in counts, suggesting that introduced forage species may have displaced nonnative invasive species. In contrast, the cover of forage and invasive nonnative species was significantly greater in seeded versus unseeded plots ($P = 0.038$) and broadcast burning more than doubled the annual production of forage grasses for 3 years after treatment. Though the author concluded that seeding of nonnative forage species had "little long-term apparent effect on native plant communities" (Lehmkuhl 2002, pg. 57), the initially high cover of introduced forage species may have reduced the cover of both nonnative invasive species and native ruderal species that establish after fires (Beyers 2004). Therefore, seeding of forage species for invasive nonnative species control must be weighed against potential impacts to native early-seral plant communities.

Silvicultural thinning of dense, young forest stands is frequently used in the coastal Douglas-fir region to encourage structural and species diversity in the understory community (Halpern and others 1999; Thysell and Carey 2001a) and, to a lesser extent, to reduce the severity of future wildfires (Graham and others 1999). Forest thinning in the coastal Douglas-fir forest region stimulates germination of seeds in the soil seed bank, including seeds of nonnative species (Bailey and others 1998; Thysell and Carey 2001a). To assess potential nonnative species response to silvicultural thinning in the coastal Douglas-fir region, Halpern and others (1999) examined the composition of soil seed banks in 40- to 60-year-old Douglas-fir and Sitka spruce stands on the Olympic Peninsula, Washington. All plots were located in stands that originated after clearcut logging. Almost 30 percent of all species represented in soil seed banks and 50 percent of all germinants from litter and soil samples were nonnative species. These species may have originally invaded the stand after clearcut timber harvest. Nonnative species observed in this study represented two basic life histories: short-lived herbaceous species that establish after clearcut logging and common weeds of agricultural areas, waste places, and roadsides. The authors concluded that silvicultural thinning of young stands may provide a temporary window for the re-establishment of nonnative species (Halpern and others 1999). The maintenance of open stand conditions in order to decrease the threat or severity of wildfire may allow the persistence of nonnative plant species in forest understories.

Common roadside weeds of the coastal Douglas-fir region may spread into burns and timber harvest units located adjacent to road networks and along firelines. At the landscape scale, Parendes (1997) tracked the invasion status of woody nonnative Scotch broom (*Cytisus scoparius*) and Himalayan blackberry (*Rubus discolor*) as well as herbaceous nonnative species such as Canada thistle, bull thistle, tansy ragwort, and St. Johnswort on the H.J. Andrews Experimental Forest in the western Oregon Cascades. Of these species, Scotch broom was closely associated with disturbance, as it was more frequent (no statistical analysis presented) on roads adjacent to timber harvest units. St. Johnswort and bull thistle were present on more than 70 percent of the road network; Canada thistle, Scotch broom, and tansy ragwort were present on 10 percent to 30 percent of the road network; Himalayan blackberry was present in only a few isolated locations (Parendes 1997). The fire ecology of Canada thistle, bull thistle, tansy ragwort, and St. Johnswort were discussed previously. A review of Scotch broom and Himalayan blackberry fire ecology follows.

In a literature review published by The Nature Conservancy, the author notes that Scotch broom does not grow well in forest understories but rapidly invades after fire or logging disturbance throughout the coastal Douglas-fir region where it forms dense thickets, spreads into native vegetation, and prevents or slows reforestation (Hoshovsky 1986, TNC review). Seed germination of Scotch broom is increased by soil heating (Regan 2001) and broadcast burning (Parker 1996), suggesting that fire may facilitate invasion of this species in the coastal Douglas-fir region.

Himalayan blackberry is an aggressive invader within the coastal Douglas-fir region, invading wet sites that have been disturbed and abandoned by humans and forming impenetrable thickets in young forest plantations and riparian areas (fig. 10-3). Similar to Scotch broom, Himalayan blackberry grows poorly in forest understories, requiring high light levels for seedling survival and fruit production. However, rapid invasion of cleared forestland suggests that Himalayan blackberry seed may remain viable in the soil for many years (Hoshovsky 1989, TNC review). Himalayan blackberry probably sprouts from root crowns after fire. Its seeds may also survive fire, explaining observations of rapid seedling establishment of many blackberry species after fire (Tirmenstein 1989a, FEIS review). Fire in the coastal Douglas-fir region may encourage invasion by this species.

Similarly, in a multi-scale assessment of nonnative plants on the Olympic Peninsula, Washington, DeFerrari and Naiman (1994) identified several nonnative plant species in clearcuts and, to a much lesser extent, young forest understories. Nonnative species recorded

Figure 10-3—Himalayan blackberry establishing in a disturbed site within the coastal Douglas-fir region. (Photo by Jed Colquhoun, Extension Weed Specialist, Oregon State University.)

in this study include species that have already been discussed: Scotch broom, Himalayan blackberry, Canada thistle, bull thistle, woodland groundsel, and tansy ragwort. In addition, cutleaf blackberry (*Rubus laciniatus*), perennial ryegrass, and common dandelion (*Taraxacum officinale*) were observed. Cutleaf blackberry shares many of the ecological and life history traits of Himalayan blackberry (Tirmenstein 1989b, FEIS review) and invades after logging and slash burning in the coastal Douglas-fir region (Steen 1966). Fire may encourage the production of reproductive tillers in perennial ryegrass (Sullivan 1992b, FEIS review) and facilitate a short-term increase in common dandelion abundance (Esser 1993b, FEIS review).

Effects of Nonnative Plant Invasions on Fuels and Fire Regimes in Coastal Douglas-fir Forests

Two nitrogen-fixing shrubs, Scotch broom (fig. 10-4) and, in coastal environments, gorse (*Ulex europaeus*), may influence fire behavior and/or fire regimes in western hemlock and Sitka spruce zone forests. According to literature reviews, both species develop seed banks that may remain viable for up to 30 years (Zielke and others 1992) and are stimulated by fire to germinate (Parker 1996; Washington State Noxious Weed Control Board 2005; Zielke and others 1992). Both species rapidly invade disturbed areas and can prevent or slow reforestation by forming dense populations (Hoshovsky 1986; Huckins 2004; Washington State Noxious Weed Control Board 2005; Zielke and others 1992). Literature reviews note that, without further disturbance, populations of Scotch broom and gorse degrade after 6 to 8 years and senesce at 10 to 15 years, allowing later-seral plant species to gradually reoccupy the site (Huckins 2004; Zielke and others 1992). As populations of gorse and Scotch broom age, they have been observed to create large amounts of litter (Hoshovsky 1986; Zielke and others 1992). Gorse leaves a center of dead vegetation as it grows outward and its stems have high oil content (Washington State Noxious Weed Control Board 2005), increasing its flammability.

Though it has not been demonstrated that the dominance of either species can increase fire spread or intensity in the coastal Douglas-fir region, the characteristics of both species indicate that such a relationship may exist. In literature reviews, Scotch broom (Hoshovsky 1986; Huckins 2004; Zielke and others 1992) and gorse (Washington State Noxious Weed Control Board 2005; Zielke and others 1992) are described as fire hazards. Early-seral stages of forest succession in coastal Douglas-fir forests are the most flammable (Agee 1997). By slowing reforestation, Scotch broom and gorse prolong these flammable early-seral stand

Figure 10-4—Scotch broom (the shrub with yellow flowers) invades disturbed areas and forms dense populations. (Photo by Jed Colquhoun, Extension Weed Specialist, Oregon State University.)

conditions. Furthermore, dense populations of senescing Scotch broom or gorse provide continuous fuels that may increase the spread of surface fires. For example, Zielke and others (1992) describe a fire that spread rapidly through a gorse understory across 2,500 acres (1,000 ha) of New Zealand forestland. Similarly, a fire that quickly overran the coastal town of Bandon, Oregon in 1936 was attributed to dense populations of gorse found in neighborhood yards (Huber 2005). The density of litter associated with Scotch broom and gorse populations and the flammable oils found in gorse stems may increase fire intensity, though comparisons with native shrubs have not been made to support this assumption. Finally, fire stimulates seed germination of Scotch broom and gorse; therefore, if these shrubs are capable of increasing the frequency of wildfire, they create the conditions necessary for their continued recruitment and dominance.

False brome (*Brachypodium sylvaticum*), a nonnative perennial bunchgrass, is rapidly invading low- to mid-elevation western hemlock zone forests of Oregon's Coast and western Cascade Ranges (fig. 10-5); it is not

Figure 10-5—False brome, a nonnative bunchgrass that invades forest understories and roadsides in western Oregon, and forms dense, continuous populations. (Photo by Tom Kaye, Institute for Applied Ecology, Corvallis, Oregon.)

currently found in Washington. False brome invades roads, clearcuts, open habitats, and understories of both young and mature undisturbed mixed conifer stands, where it dominates the herbaceous layer and forms dense, continuous populations that exclude most native species (False Brome Working Group 2002, 2004; Kaye 2001). Members of the False Brome Working Group have speculated that false brome may alter fire regimes (False Brome Working Group 2002). False brome may increase biomass of fine fuels capable of carrying late-season fires, particularly in well established stands of false brome that have accumulated a heavy build-up of thatch. Alternatively, populations of false brome may decrease understory fire spread of early- and mid-season fires because it has been observed to stay green until late fall (False Brome Working Group 2002). False brome reproduces rapidly from seed but is not rhizomatous (Kaye 2001). Observations indicate that false brome is not controlled by burning (False Brome Working Group 2003).

Use of Fire to Manage Invasive Plants in Coastal Douglas-fir Forests

Though broadcast burning is regularly used to prepare timber harvest units for reforestation, fire is not commonly used for nonnative species control within the coastal Douglas-fir region.

Upper Montane Conifer Forests and Meadows

Above the extensive low- to mid-elevation western hemlock zone forests of Washington and Oregon, upper montane slopes of the Olympic Range and the western Cascade Range are dominated by cold, wet conifer forests. Two forest zones are represented in this region: Pacific silver fir (*Abies amabilis*) at middle to high elevations and mountain hemlock (*Tsuga mertensiana*) in subalpine environments. Wet meadows are found in both forest zones but are more common within the mountain hemlock zone, while dry grassy balds occur along high-elevation ridges of the Olympic Range (Franklin and Dyrness 1973).

Infrequent stand-replacing fires with return intervals of 200 or more years typically characterize the fire regimes of upper montane environments of the western Cascade and Olympic Ranges (Arno 2000, pg. 99). In the western Cascade Range, extensive areas of mountain hemlock-zone forest burned during the latter half of the 19th and early 20th centuries (Franklin and Dyrness 1973). Past fires probably contributed to the establishment and maintenance of high elevation meadows in both the Cascade and Olympic Ranges (Henderson 1973, as cited by Franklin and Dyrness 1973). Fire exclusion in the latter part of the 20th century may be allowing the gradual succession of mountain meadows to forest. Consequently, land managers are experimenting with the use of prescribed fire in order to prevent forest encroachment and maintain or restore meadow composition and structure (Halpern 1999).

Upper montane environments in the Northwest Coastal bioregion support fewer nonnative species than lower elevation sites (DeFerrari 1993; Parendes 1997; Sarr and others 2003). For instance, in the Olympic Mountains, nonnative plant diversity is lower in mature forests located in high-elevation protected wilderness areas than in lower elevation forests (DeFerrari and Naiman 1994). This trend also holds true along roadsides; in the western Oregon Cascades, many roadside nonnative species decrease in abundance with increased elevation (Parendes 1997). Microclimatic conditions found at high elevations may limit the spread of nonnative species; however, several other factors that influence nonnative species establishment and spread, such as precipitation, density and ages of roads, and proximity to source populations, are confounded with elevation and may play an even larger role in limiting invasions.

Though nonnative plant abundance is generally low in upper montane forests of the Northwest Coastal bioregion, nonnative species are common invaders along roadsides and after disturbances, such as landslides and debris flows. In a decommissioned parking lot located in the northeast corner of Olympic National

Park and within a subalpine meadow dominated by Idaho fescue (*Festuca idahoensis*), established populations of dandelion and timothy (*Phleum pratense*) maintained cover over an 8-year study period, while Kentucky bluegrass (*Poa pratensis*) continued to spread into heavily impacted areas (Schreiner 1982). On the debris flow that followed the eruption of Mt. St. Helens in 1980, several wind-dispersed nonnative species established, including Canada thistle (Dale 1989, 1991), woodland groundsel (Dale 1989; Dale and Adams 2003), and hairy catsear (*Hypochaeris radicata*) (Dale and Adams 2003; del Moral and others 1995). Hairy catsear and woodland groundsel were abundant in the post-eruption seed rain (Wood and del Moral 2000), along with lesser quantities of bull thistle, Canada thistle, common dandelion, common groundsel (*Senecio vulgaris*), and wall-lettuce. All of these species have wind-dispersed seed and are frequently observed invading recently disturbed environments. The presence of nonnative plants in roaded and disturbed environments may increase the likelihood of establishment in burned areas.

Though these nonnative species may invade burns in upper montane environments of the Northwest Coastal bioregion, their ability to do so has not been demonstrated. However, populations of Canada thistle (Zouhar 2001d, FEIS review), bull thistle (Zouhar 2002b, FEIS review), and common dandelion (Esser 1993b) sometimes increase after fire in mountain environments of the Interior Northwest. Fire has also been observed to stimulate tiller production and growth in timothy (Esser 1993a, FEIS review). In addition, woodland groundsel is closely associated with slash burns in coastal Douglas-fir forests of the Northwest Coastal bioregion.

Role of Fire in Promoting Invasions of Nonnative Plant Species in Upper Montane Communities

Past research suggests that burned areas in upper montane environments of the Cascade Range may be largely free of nonnative species. Fire effects studies conducted in wilderness areas and National Parks located along the crest of the Washington and Oregon Cascades have found no evidence of nonnative species in burned or unburned plots (Douglas and Ballard 1971; Fahnestock 1977; Hemstrom and Franklin 1982; Miller and Miller 1976). However, these studies were conducted more than 2 decades ago. Follow-up research is needed to determine whether nonnative plants have invaded these areas or other burned areas more recently. Whether this conspicuous absence of nonnative species is due to a lack of local seed source or to environmental barriers to establishment has also not been explored.

Agee and Huff (1980) examined the effects of the Hoh fire, which burned through both lower and upper montane forests of the western slope of the Olympic Range. One year after the blaze, burned plots supported three wind-dispersed nonnative species–hairy catsear, wall-lettuce, and tansy ragwort. These species were absent from adjacent undisturbed forest. The authors did not indicate whether these species were observed in lower montane environments, upper montane environments, or both.

There is some concern that reintroduction of fire into alpine and subalpine meadow communities to prevent forest encroachment and restore meadow community composition and structure may inadvertently facilitate invasion of nonnative species (Halpern 1999). Prior to the application of prescribed fire in a subalpine bunchgrass meadow in the western Oregon Cascades, Halpern (1999) noted that nonnative plant species contributed very little to the diversity of species or the vegetative cover of herbaceous plants in the meadow. Five nonnative species were observed in the meadow community prior to the prescribed burn: quackgrass (*Elymus repens*), creeping bentgrass (*Agrostis stolonifera*), Kentucky bluegrass, common sheep sorrel (*Rumex acetosella*), and yellow salsify (*Tragopogon dubius*). Although these investigators did not assess postfire vegetation responses, low-severity, late fall burns such as the one conducted in this study would be unlikely to affect the spread of Kentucky bluegrass (Uchytil 1993, FEIS review) or common sheep sorrel (Esser 1995, FEIS review), but might reduce the spread of quackgrass (Snyder 1992a, FEIS review).

Effects of Nonnative Plant Invasions on Fuels and Fire Regimes in Upper Montane Communities

There is no indication that nonnative species are changing the fire regimes of upper montane environments of the Pacific Northwest. Nonnative plant abundance is low throughout this region and, therefore, unlikely to cause significant changes to fuel characteristics. Furthermore, the wet, cold climate of upper montane environments prevents the ignition and limits the spread of fire. Fires in this region are closely associated with periods of extreme fire weather, regardless of fuel characteristics that may be influenced by the presence and abundance of nonnative species.

Use of Fire to Manage Invasive Plants in Upper Montane Communities

Nonnative plant species are not presently posing a serious threat to upper montane plant communities of the Northwest Coastal bioregion. Prescribed fire is not considered to be a useful management tool in subalpine forests as fires do not spread under controllable conditions, and native forest dominants tend to be fire avoiders (Agee 1993). However, there is growing interest in using prescribed fire to maintain or restore subalpine meadows that are gradually being lost to forest succession (Halpern 1999). Management activities in these areas should be designed to prevent the accidental introduction of nonnative species' propagules and the promotion of nonnative species establishment and spread.

Riparian Forests

Low- and mid-elevation riparian forests of the coastal region of Washington and Oregon are covered in this section. Dominant trees of riparian corridors which dissect coastal Douglas-fir forests include red alder, bigleaf maple (*Acer macrophyllum*), Sitka spruce, western redcedar, and western hemlock. Riparian forests of low-elevation valleys, such as the Willamette Valley of Oregon and the lower Columbia River Valley of western Oregon and Washington, are dominated by black cottonwood (*Populus trichocarpa*), Oregon ash (*Fraxinus latifolia*), bigleaf maple, red alder, white alder (*Alnus rhombifolia*), and Oregon white oak (*Quercus garryana*). Riparian corridors of the Coast Range and western Cascades are frequently located in steep ravines, while those of the western Olympic Peninsula are typically located on river terraces. Riparian forests in low-elevation valleys have been extensively cleared for agriculture and urban development.

Riparian communities are highly susceptible to invasion by nonnative plants because of frequent natural (floods and landslides) and anthropogenic disturbances. Because of the down gradient movement of soils and plant propagules, riparian communities are often subject to the cumulative ecological damage sustained by entire watersheds (Naiman and others 2000), including the effects of road building, timber harvesting, slash burning, grazing, mining, fire suppression activities, and water withdrawals. These activities have led to a loss of native species (Naiman and others 2000) and increased vulnerability to invasion of nonnative plants. Nonnative plants typically contribute 20 to 30 percent, and can contribute up to 75 percent, of total richness in riparian communities in the Northwest Coastal bioregion (DeFerrari and Naiman 1994; Naiman and others 2000; Planty-Tabacchi and others 1996). In general, the proportion of total richness composed of nonnative species increases from headwaters to piedmont (the transition zone between mountains and lowlands) and remains high along the lower reaches of streams and rivers where human impacts and development are concentrated (Planty-Tabacchi and others 1996). Young, disturbed patches of riparian vegetation are considerably more invaded by nonnative species than more stable, mature patches (DeFerrari and Naiman 1994; Parendes and Jones 2000; Planty-Tabacchi and others 1996). Similarly, riparian communities that dissect coastal Douglas-fir forests are more invaded than adjacent conifer forest (DeFerrari and Naiman 1994; Planty-Tabacchi and others 1996). High abundance of nonnative plants in riparian communities of the Northwest Coastal bioregion highlights the importance of riparian habitats as source populations and dispersal corridors for these species (DeFerrari and Naiman 1994; Naiman and others 2000; Parendes and Jones 2000).

The ecological role of fire in riparian forests of the Northwest Coastal bioregion is only beginning to be explored. Riparian environments tend to be cooler and moister than adjacent uplands; therefore, the flammability of riparian vegetation may be lower than that of vegetation in upland forests (Olson and Agee 2005). High moisture levels coupled with fewer ignitions due to lower slope positions suggest lower fire frequency and higher fire severity in riparian forests than in upland forests (Minnich 1977, as cited by Olson and Agee 2005). Thus, riparian corridors can serve as fire refugia. However, deviations from this trend have also been observed. In a study of historic fires in riparian and upland forests in the Umpqua River drainage of the southern Oregon Cascades, Olson and Agee (2005) found no difference in fire-return intervals between riparian and upland forests. Furthermore, when severe fire weather affects Northwest Coastal bioregion forests (Agee 1997), riparian forests may be as likely to burn as adjacent upland forests.

Riparian vegetation tends to recover rapidly after fire disturbance (Beschta and others 2004). However, fire exclusion in riparian habitats may alter the successional development of plant communities (Gregory 1997). After years of fire exclusion and timber harvest in both upland and riparian locations, the ecosystem impacts of fire may be severe and recovery may be slow or incomplete (Beschta and others 2004).

Role of Fire in Promoting Nonnative Plant Invasions in Riparian Forests

Fire in riparian forests of the Northwest Coastal bioregion may encourage the spread of nonnative plants (DeFerrari and Naiman 1994; Parendes and Jones 2000; Planty-Tabacchi and others 1996; Thompson 2001). Himalayan blackberry (Tirmenstein 1989a), cutleaf blackberry (Termenstein 1989b, FEIS review), St. Johnswort (Zouhar 2004), Canada thistle (Zouhar 2001d), bull thistle (Zouhar 2002b), common sheep sorrel (Esser 1995), perennial ryegrass, Scotch broom, white sweetclover (*Melilotus alba*), and common dandelion (Esser 1993b) have all been observed to establish or increase abundance after fire. The mechanism of response to fire varies by species; for example, fire stimulates the production of reproductive tillers in perennial ryegrass (Sullivan 1992b) and seed germination of Scotch broom (Regan 2001) and white sweetclover (Uchytil 1992a, FEIS review).

Fires in upslope forests may indirectly affect nonnative plant invasions in riparian forests by affecting the timing and magnitude of soil erosion (Bisson and others 2003). Forested slopes of the Cascade, Olympic, and Coast Ranges are steep, and landslides and debris flows are common on both disturbed (fire, logging, road building) and undisturbed slopes in these areas (Miles and Swanson 1986). Landslides and storm runoff deliver soil, wood, and plant propagules from upland sources to riparian communities. For example, the invasion of Scotch broom and foxglove into western Cascade riparian habitats of Oregon may be limited by upland seed sources, such as roads and clearcuts, and the distributions of both species in this area are consistent with down gradient movement of seed from upslope locations to riparian communities (Watterson 2004). In contrast, riparian corridors may not be important sources of nonnative propagules for upland forested locations (DeFerrari and Naiman 1994). Nonnative species are more common on landslides caused by road building and clearcutting than on landslides located on undisturbed slopes. Common nonnatives found on landslides located near such disturbances include species that are frequently sown on road embankments for erosion control (Miles and Swanson 1986).

Seeding of nonnative plant species in postfire recovery projects alters successional pathways and compounds fire-related impacts (Beschta and others 2004). Postfire seeding of nonnative grasses and forbs is often done to reduce soil erosion, prevent landslides, and protect riparian and aquatic habitats (Beyers 2004). However, the establishment of a dense cover of nonnative vegetation can inhibit the regeneration of native woody species and eliminate native ruderal herbs from postfire ecosystems (Beschta and others 2004). Furthermore, several of the nonnative species commonly sown to slow soil erosion, such as timothy, orchard grass, and tall fescue (*Lolium arundinaceum*), have invaded riparian habitats (DeFerrari and Naiman 1994; Planty-Tabacchi and others 1996). Postfire recovery projects should ideally be aimed at enhancing reestablishment of native vegetation. Unfortunately, it is often difficult to get native seed for postfire recovery projects when and where it is needed (Beyers 2004).

Effects of Nonnative Plant Invasions on Fuels and Fire Regimes in Riparian Forests

Despite increasing concern over the effects of nonnative plants in riparian communities, their effect on fire regimes has received limited attention. Nonnative species that may influence riparian fire regimes include forage grasses used in postfire seeding projects, English ivy (*Hedera helix*), and knotweed (*Polygonum* spp.). However, the magnitude and direction of these species' effects on fuel characteristics and fire regimes are unknown.

Postfire seeding of nonnative grasses to slow soil erosion and protect riparian and aquatic habitats may increase the flammability of burned sites. Seeded grasses can form continuous fuel beds with high surface to volume ratios that encourage rapid rates of fire spread. Furthermore, many nonnative grasses are dry and flammable during the late summer and early fall fire seasons (Beschta and others 2004). More research is needed to assess the relationship between fire recovery activities and fuels.

Given its growth habit, the invasion of English ivy may influence fire behavior. English ivy is an aggressive nonnative vine that poses a threat to nearly all forest habitats in the Northwest Coastal bioregion below 3,000 feet (900 m), but is especially problematic in moist and riparian forests and urban and suburban areas. English ivy grows both along the ground, where it covers over native vegetation, and up trees, where it attaches to tree bark with root-like structures and rapidly climbs into the canopy. Soll (2004c, TNC review) found that host trees have low vigor and, within a few years, are killed and/or vulnerable to tip-over and blow down. After tree canopies are destroyed by the invasion of English ivy, shade-intolerant nonnative species, such as Himalayan blackberry, may become established (Soll 2004c). Though dense populations of English ivy clearly affect the structure of surface and crown fuels, their impact on fire behavior is unknown. In the moist forests where English ivy occurs, extreme fire weather may be a more important driving force of fire intensity and severity than fuel characteristics (Agee 1997); therefore, even if English ivy causes marked changes in fuel characteristics it may have little or no influence on local fire regimes. However, given English ivy's abundance near populated areas, further research may be warranted.

Another group of invasive nonnative species that has spread rapidly in Pacific Northwest riparian communities and may affect fire regimes includes Japanese, giant, and cultivated knotweeds (*Polygonum cuspidatum*, *P. sachalinense*, *P. polystachyum*). Knotweeds are fast growing and invade recently disturbed soils where they quickly outgrow and suppress native vegetation (Soll 2004a, TNC review). It has been suggested that Japanese knotweed (*Polygonum cuspidatum*) populations pose a fire hazard during the dormant season due to dense accumulations of dead plant material (Ahrens 1975). However, tissues of Japanese knotweed have relatively low heat content (Dibble and others 2004), so fires in these populations may be of relatively low intensity and severity. More research is needed to determine whether knotweed populations may influence fire behavior, severity, or frequency.

Use of Fire to Manage Invasive Plants in Riparian Forests

There is little information available regarding the use of fire in Northwest riparian communities to control invasive plants, with the exception of reed canarygrass (*Phalaris arundinacea*). Reed canarygrass invades and dominates valley wetland and riparian communities in the Northwest Coastal bioregion. It typically forms dense monotypic stands in wetlands, moist meadows, and riparian communities, excluding native vegetation. Altered water levels and human-caused disturbances appear to facilitate the invasion of reed canarygrass (Lyons 1998, TNC review).

Reed canarygrass's response to prescribed burning is mixed. Hutchison (1992b) found that prescribed burning is an effective method of controlling reed canarygrass in productive sites containing seed banks of native, fire-adapted species, such as wet prairie habitats. In fact, some native wetland species may be unable to compete with reed canarygrass without prescribed fire (Hutchison 1992b). However, Lyons (1998) states that fire does not always kill mature reed canarygrass and may even stimulate stem production unless the fire burns through the sod layer to mineral soil. The high temperature required at the soil surface may be difficult to achieve as reed canarygrass stays green late into the fire season and "so does not burn very hot" (Tu 2004, pg. 6). According to a TNC regional management report (Tu 2004), prescribed fires are typically applied in the fall in the Pacific Northwest and, therefore, may not be severe enough to kill mature reed canarygrass when used in this region. Herbicide treatment prior to prescribed fire may help increase fuel loads and reed canarygrass mortality. Successive burn treatments may not be a control option, as it is frequently impossible to burn stands of reed canarygrass several consecutive years in a row due to a lack of fine fuels after only one burn. Overall, in Northwest Coastal bioregion riparian areas, prescribed fire may be more effective as a pretreatment before other types of control efforts, such as tillage, shade cloth, or herbicide application (Tu 2004).

Oregon Oak Woodlands and Prairies

Oregon oak woodlands and prairies comprised the historic vegetation of the Willamette Valley of Oregon and the Puget Lowlands of Washington. Prairie vegetation is also found on the San Juan Islands, in locations on the western Olympic Peninsula, on coastal headlands in Oregon, and along the shores of Puget Sound and the Straits of Georgia. Oregon oak woodlands were historically composed of native grasses and forbs with Oregon white oak either in open-grown stands or as solitary trees, and other low-stature broadleaved trees and shrubs. Prior to Euro-American settlement and widespread conversion of Oregon oak woodlands and prairie habitats to agriculture and urban development, American Indians burned Oregon oak woodland habitat every year to every several years in order to increase the production of desired plants and herd game (Boyd 1986; Johannessen and others 1971; Norton 1979; Wray and Anderson 2003). Presettlement fire regimes of Oregon oak woodlands were characterized by low-severity understory fires occurring every 35 years or less (Arno 2000, pg. 98). The dominant tree species, Oregon white oak, persisted due to its resistance to low-severity fire (Agee 1996b).

Today, due to extensive livestock grazing and agricultural and urban development, Oregon oak woodlands and prairies are severely fragmented and degraded. Less than 1 percent of presettlement condition Oregon oak woodland habitat remains (Crawford and Hall 1997; Kaye and others 2001; Pendergrass 1996). Nonnative species abound, competing with native plants and often dominating the herbaceous vegetation. Fire exclusion and cessation of burning by Native Americans allowed the invasion and establishment of woody species, such as native Douglas-fir and Oregon ash, as well as nonnative Scotch broom and Himalayan blackberry in previously open woodlands and prairies (Johannessen and others 1971; Pendergrass 1996; Thilenius 1968; Thysell and Carey 2001b; Towle 1982). Increasingly dense and widespread shrub layers are associated with a decreasing abundance of native forbs (Parker and others 1997; Thilenius 1968). The loss of Oregon oak woodland habitat endangers many species such as Bradshaw's lomatium (*Lomatium bradshawii*) (Kaye and others 2001; Pendergrass and others 1999), Curtis aster (*Symphyotrichum retroflexum*) (Clampitt 1993; Giblin 1997), Kincaid's lupine (*Lupinus organus* var. *kincaidii*), Fender's blue butterfly (*Icaricia icarioides fenderi*) (Schultz and Crone 1998), and Oregon silverspot butterfly (*Speyeria zerene hippolyta*) (Pickering and others 2000).

Reserves and wildlife refuges in this region are experimenting with the reintroduction of low-severity fire to maintain and restore Oregon oak woodland habitats and species. Prescribed fires are most often set in the fall, in keeping with the presettlement fire regime. However, with agricultural development and widespread livestock grazing, nonnative plant species now dominate the herbaceous and shrub layers of these plant communities, complicating restoration efforts.

Competition between native and nonnative species may also alter community structure and composition in the postfire environment, even when the reintroduced fire regime is similar to the presettlement regime (Agee 1996a).

Role of Fire in Promoting Nonnative Plant Invasions in Woodlands and Prairies

Though fire is reintroduced into Oregon oak woodlands and prairies by land managers with the dual goals of decreasing nonnative plants and increasing native plants, the diversity of nonnative species established in Oregon oak woodland habitats ensures that at least some will respond favorably to fire. Fire increases the reproduction, germination, establishment, and/or growth of a number of nonnative species that invade these communities (table 10-3).

Herbaceous species—Prescribed fires in Oregon oak woodland habitats have increased the establishment, frequency, and cover of a number of nonnative herbaceous species (table 10-3). Annual fires, in particular, appear to strongly favor nonnative ruderal herbaceous species over native grasses. For instance, in remnant Puget Lowland prairies located at Fort Lewis, Washington, 50 years of annual burning have resulted in the complete replacement of the dominant native species, Idaho fescue, with nonnative forbs and annual grasses, such as hairy catsear and annual vernal grass (*Anthoxanthum aristatum*) (Tveten 1997; Tveten and Fonda 1999). Researchers concluded that native prairie communities, while adapted to frequent, low-severity fires, are not adapted to prolonged annual burning. If prescribed fire is introduced too frequently, land managers may inadvertently encourage the invasion and dominance of nonnative plant species in these communities.

When fire is introduced less frequently or introduced after an extended period of fire-free conditions, impacts on the composition of Oregon oak woodland communities are less obvious and more complicated. In general, prescribed fire encourages the establishment and spread of nonnative ruderal herbaceous species in Oregon oak woodlands. Herbaceous species observed to establish or increase in frequency and/or cover after prescribed fire include annual vernal grass (Clark and Wilson 2001), colonial bentgrass (*Agrostis capillaris*) (Parker 1996), little quakinggrass (*Briza minor*) (Pendergrass 1996), garden cornflower (*Centaurea cyanus*) (Maret 1997), common velvetgrass (*Holcus lanatus*) (Agee 1996a; Dunwiddie 2002; Pickering

Table 10-3—Nonnative plant species' responses to prescribed fire in lowland prairies of the Northwest Coastal bioregion. Statistical significance provided where available.

Species	Burn season	Effect	Direction	Significance	Authors	Notes
Anthoxanthum odoratum sweet vernal grass	Fall	Flowering	+	*P* = 0.02	Clark and Wilson 2001	
Agrostis capillaris colonial bentgrass	Fall	Cover	+		Parker 1996	Correlated with burn temperature
	Spring	Frequency	–	Not significant	Tveten 1997 Tveten and Fonda 1999	
Briza minor little quakinggrass	Fall	Cover	+		Pendergrass 1996	
Carex pensylvanica Penn sedge	Fall	Frequency	–	*P* < 0.05	Tveten 1997 Tveten and Fonda 1999	
Centaurea cyanus garden cornflower	Fall	Establishment	+	*P* < 0.005	Maret 1997	
Cytisus scoparius scotch broom		Germination	+		Parker 1996	
			+		Regan 2001	Greenhouse study
			+		Agee 1996a	
		Cover	–		Agee 1996a	
	Fall		–	*P* < 0.05	Tveten 1997 Tveten and Fonda 1999	
	Fall	Density	–	*P* < 0.05	Tveten 1997 Tveten and Fonda 1999	
	Spring	Seedling density	–	*P* < 0.05	Tveten 1997 Tveten and Fonda 1999	

Table 10-3—(Continued)

Species	Burn season	Effect	Direction	Significance	Authors	Notes
Holcus lanatus common velvetgrass	Fall	Cover	+		Dunwiddie 2002	1st and 2nd burns in 15 yrs
	Fall		–		Dunwiddie 2002	3rd burn in 15 yrs
	Fall		–		Schuller 1997	
	Fall	Flowering	–	$P = 0.01$	Clark and Wilson 2001	
	Summer	Frequency	+		Schuller 1997	
	Fall		+	$P < 0.1$	Pickering and others 2000	
		Establishment	+		Agee 1996a, b	Positively associated with severely burned microsites
Hypericum perforatum St.Johnswort	Fall	Cover	+		Pendergrass 1996	
	Fall		–	$P < 0.01$	Clark and Wilson 2001	
	Fall	Frequency	+	$r > 0.50$	Streatfeild and Frenkel 1997	Correlated with time since burn
Hypochaeris radicata hairy catsear	Fall	Cover	+		Pendergrass 1996	
	Spring, Fall		–	$P < 0.05$	Tveten 1997	
					Tveten and Fonda 1999	
	Summer	Frequency	+		Schuller 1997	
	Fall	Establishment	+	$P < 0.005$	Maret 1997	
	Fall		+	$P < 0.05$	Tveten 1997 Tveten and Fonda 1999	
Luzula congesta heath woodrush	Spring, Fall	Frequency	+	$P < 0.05$	Tveten 1997 Tveten and Fonda 1999	
Leontodon hirtus rough hawkbit	Fall	Cover	+	Pendergrass 1996		
Parentucellia viscosa yellow glandweed	Fall	Frequency	+	$r > 0.5$	Streatfeild and Frenkel 1997	Time since burn
Poa pratensis Kentucky bluegrass	Fall	Cover	–	$P < 0.05$	Tveten 1997 Tveten and Fonda 1999	
		Frequency	+	$r > 0.50$	Streatfeild and Frenkel 1997	Burn history
Pyrus communis common pear	Fall	Sprouting	+	Not significant	Pendergrass and others 1998	
	Fall	Stem height	–		Pendergrass 1996	
Rosa eglanteria sweetbriar rose	Fall	Sprouting	+	Not significant	Pendergrass and others 1998	
	Fall	Stem height	–	Not significant	Pendergrass 1996	
	Fall		–		Streatfeild and Frenkel 1997	
Rubus discolor Himalayan blackberry	Fall	Establishment	+	Not significant	Pendergrass 1996	
Rubus laciniatus evergreen blackberry	Fall	Establishment	+	Not significant	Pendergrass 1996	
Rumex acetosella common sheep sorrel	Fall	Cover	+		Dunwiddie 2002	1st and 2nd burns in 15 yrs
	Fall	Cover	–		Dunwiddie 2002	3rd burn in 15 yrs
	Spring, Fall	Frequency	+		Tveten 1997	
	Spring, Fall		+	$P < 0.1$	Pickering and others 2000	
Senecio jacobaea tansy ragwort		Establishment	+		Agee 1996a	Positively associated with severely burned microsites

and others 2000; Schuller 1997), St. Johnswort (Pendergrass 1996; Streatfeild and Frenkel 1997), hairy catsear (Maret 1997; Pendergrass 1996; Schuller 1997; Tveten 1997; Tveten and Fonda 1999), heath woodrush (*Luzula congesta*) (Tveten 1997; Tveten and Fonda 1999), yellow glandweed (*Parentucellia viscosa*) (Streatfeild and Frenkel 1997), Kentucky bluegrass (*Poa pratensis*) (Tveten 1997; Tveten and Fonda 1999), common sheep sorrel (Dunwiddie 2002; Pickering and others 2000; Tveten 1997), and tansy ragwort (Agee 1996a). Results from regional fire effects studies are summarized in table 10-3.

Despite this trend, nonnative plants rarely respond predictably and consistently to prescribed fire in this region. Many species have been observed to respond both positively and negatively to fire, perhaps related to differences in environment, community composition, season of burn, or fire frequency. Short-term responses may not be predictive of long-term trends. For example, cover of velvetgrass is usually reduced by fire (Clark and Wilson 2001; Dunwiddie 2002; Schuller 1997), while its frequency is increased (Pickering and others 2000; Schuller 1997). Though fire damages mature plants, it strongly favors seedling establishment. A short-term reduction in cover may, therefore, give way to a long-term increase in population density and cover.

Due to the extensive invasion of nonnative plant species that has occurred over the last century or more, the environmental impact of fire has also been fundamentally altered in Oregon oak woodland and prairie communities. Many prairie and woodland habitats are no longer dominated by native bunchgrasses and instead support nonnative thatch-forming grasses. Because of this transition, prescribed fires in habitats extensively invaded by nonnative grasses may create microsites favorable for seedling establishment of herbaceous species, both nonnative and native, primarily through removal of accumulations of litter (Maret 1997). For example, in a study conducted in the Willamette Valley, fall broadcast burning prior to sowing of common nonnative species significantly increased establishment of garden cornflower ($P < 0.005$) in a prairie dominated by nonnative annual grasses, and significantly increased establishment of hairy catsear ($P < 0.005$) in a prairie dominated by nonnative tall oatgrass (*Arrhenatherum elatius*). The author speculated that burning may not create favorable microsites for seedling establishment in communities dominated by native bunchgrasses because litter accumulations are much less (Maret 1997). This observation points to a fundamental change that has occurred to the composition, structure, and dynamics of Oregon oak woodland communities since Euro-American settlement, one that affects nonnative and native species' responses to fire and cannot be simply undone by reintroducing presettlement fire regimes.

Woody species—Prescribed fire increases the abundance of several nonnative woody species in Oregon oak woodland habitats (table 10-3), primarily through sprouting of underground parts or seed scarification. For example, prescribed fires increase the stem density of several nonnative woody species that sprout from underground parts in response to disturbance, including sweetbriar rose (*Rosa eglanteria*) and common pear (*Pyrus communis*) (Pendergrass and others 1998), though stem heights are reduced (Pendergrass 1996; Streatfeild and Frenkel 1997). Burned prairie sites have also been associated with the establishment and increase of nonnative blackberries (*Rubus discolor, R. laciniatus*) (Pendergrass 1996).

Scotch broom is another woody species that often increases after prescribed fire. Though fire reduces the cover and density of mature plants (Agee 1996a; Tveten 1997; Tveten and Fonda 1999), burning and soil heating increase germination of Scotch broom by scarifying seed coats (Parker 1996; Regan 2001). In a greenhouse study conducted with soils from Fort Lewis, Washington, Regan (2001) found that Scotch broom germination greatly increased with soil heating (140 °F (60 °C) for 10 minutes), leading him to conclude that prescribed burns would increase the germination of Scotch broom. Field studies have confirmed this hypothesis (Agee 1996a; Parker 1996).

Scotch broom has extensively invaded prairies of the Puget Lowlands, forming monotypic stands in some locations (Parker 1996; Tveten 1997). Though Scotch broom invasion is typically associated with fire or other disturbance, a unique situation exists in some prairie communities of the Puget Lowlands of Washington. Unlike other prairie vegetation in the Northwest Coastal bioregion, prairies of the Puget Lowlands are usually found on glacial outwash and characterized by the presence of a biological soil crust composed of algae, lichens, and liverworts. Parker (1996, 2001) conducted a seeding experiment to assess the importance of seedbeds to the invasion of Scotch broom in these prairies. Significantly more Scotch broom seedlings emerged in untreated control plots than in any other treatment ($P = 0.01$), including broadcast burning before and after seeding. The author suggests that, in prairies located on nutrient-poor glacial outwash, biological soil crusts may be facilitating the invasion and establishment of Scotch broom. Though Parker (1996) noted that broadcast burning after seeding increased Scotch broom seed germination, she concluded that prescribed fire does not necessarily increase the success of Scotch broom seedling establishment in some Puget Lowland prairies. Rather, Scotch broom is more likely to establish in undisturbed prairies than ones that are regularly burned (Parker 1996).

Effects of Nonnative Plant Invasions on Fuels and Fire Regimes in Woodlands and Prairies

Due in part to invasion by native and nonnative woody plants, the fire regime of lowland prairies has shifted from a low-severity regime, maintained by frequent, anthropogenic ignitions and fueled by grasses and herbaceous vegetation, to a mixed-severity regime with lengthened fire-return intervals and accumulations of woody fuels. Over the last century, forest succession in Oregon oak woodlands and prairies has been associated with the spread of Douglas-fir and broadleaved trees into previously open habitats. Fire-resistant, open-grown Oregon white oak stands have grown increasingly dense with small, clustered stems. Native trees such as Douglas-fir, bigleaf maple, and Oregon ash, and nonnative trees such as sweet cherry (*Prunus avium*), common pear, paradise apple (*Malus pumila*), and oneseed hawthorn (*Crataegus monogyna*) have established in open prairies and oak understories and given oak stands a distinctly two layered appearance (Thilenius 1968). Canopy closure of Douglas-fir, which grows considerably taller than Oregon white oak, eventually kills overtopped oaks and may contribute to an accumulation of large woody fuel.

Shrub layers composed of native and nonnative plants have also thickened and spread due to fire exclusion (Chappell and Crawford 1997; Thilenius 1968). Invading nonnative shrubs include Himalayan and cutleaf blackberry, Scotch broom, and sweetbriar rose, all of which form dense, impenetrable thickets (Hoshovsky 1986, 1989; Pendergrass 1996; Soll 2004b). Himalayan blackberry (Soll 2004b) and Scotch broom (Hoshovsky 1986) populations are associated with accumulations of dead plant material. Dense stands of small oaks and other trees, and thick understories of Scotch broom and other nonnative shrubs provide fuels for intense, high-severity fires that sweep into the crowns of large oaks (Thysell and Carey 2001b). While resistant to low-severity fire, Oregon white oaks are vulnerable to high-severity fires fueled by native and nonnative woody plants.

The widespread replacement of native bunchgrasses with thatch-forming nonnative grasses such as tall oatgrass and false brome may change the behavior and severity of surface fires from historic conditions. No research is currently available on this topic.

Use of Fire to Manage Invasive Plants in Woodlands and Prairies

Several studies of lowland prairie restoration have examined the effectiveness of prescribed fire to both control nonnative invasive plants and encourage the establishment and growth of native plants (Clark and Wilson 2001; Ewing 2002; Maret and Wilson 2000; Parker 1996; Pendergrass 1996; Pickering and others 2000; Streatfeild and Frenkel 1997; Wilson and others 2004). In plant communities adapted to low-severity fire regimes, native plants are usually not killed by fire unless fuel buildup is excessively high or native plants have low vigor prior to burning (Agee 1996a). Both conditions are prevalent in lowland prairie communities of the Pacific Northwest (Dunwiddie 2002). Several native plants of lowland prairies respond favorably to prescribed fire (Agee 1996a; Clark and Wilson 2001; Kaye and others 2001). Others, such as Idaho fescue (Agee 1996a; Ewing 2002), Roemer's fescue (*Festuca roemeri*) (Dunwiddie 2002), and tufted hairgrass (*Deschampsia caespitosa*) (Clark and Wilson 2001) are more sensitive to fire but can be replanted after fires aimed at eradicating nonnative species (Agee 1996a) or creating impoverished soil conditions. Reducing soil nutrients and organic matter through application of fire may give some native species a competitive advantage over nonnative invasive species (Ewing 2002).

Restoration projects in lowland prairies are frequently based on the assumption that prescribed fires promote or maintain native herbaceous species, and that fire inhibits nonnative herbaceous and woody species because they are not fire-adapted. Studies of prairie restoration have either weakly supported or weakly refuted this assumption. For example, in a short-term replicated experiment conducted in the southern Willamette Valley, the effects of 2 years (1994, 1996) of prescribed burns (conducted in September-October) were compared with other restoration treatments in a remnant patch of wetland prairie extensively invaded by nonnative grasses and forbs and native and nonnative woody plants. Burning significantly reduced the cover of nonnative forbs as a group ($P = 0.03$) and significantly increased the cover of native forbs ($P = 0.04$), supporting the hypothesis that prescribed fires tend to favor native species over nonnative species. In particular, St. Johnswort cover and the flowering of common velvetgrass were reduced after fire (Clark and Wilson 2001); however, both of these species also responded favorably to fire in other regional studies (table 10-3).

In contrast, another study of Willamette Valley wet prairie restoration indicated that prescribed burns were effective at increasing native forbs but ineffective at controlling nonnative plant species. Prior to the burns, community species richness was dominated by native forbs, while nonnative perennial grasses dominated vegetation cover. Broadcast burns were conducted in fall of 1988 and 1989 with strip-head burning techniques and were reported to reach lethal temperatures at the soil surface, with short residence times. After the burns, the frequency of native annual and perennial forbs increased in most of communities sampled (four out of five, and three out of five communities sampled, respectively); however, cover of native annual forbs decreased in two communities as well. Though the frequency of nonnative perennial graminoids decreased in most communities sampled, the total cover of nonnative species increased in all but one community (Pendergrass 1996).

Similarly, at Oak Patch Natural Area Preserve, Washington, Oregon white oak regeneration, as well as establishment of nonnative herbaceous species, increased after prescribed burning (Agee 1996a, b). The site had been extensively invaded by Douglas-fir, and most of the mid-sized Douglas-fir were cut and removed prior to the burn. Oregon white oak regeneration was associated with high-severity burn patches where small logs had burned and most soil organic matter had been consumed. However, these same high-severity burn patches were also associated with establishment of nonnative invasive species such as tansy ragwort and common velvetgrass. No statistical analysis was presented (Agee 1996a, b).

In other situations, the reintroduction of fire appears to have little immediate impact, positive or negative, on the composition of degraded prairie communities. For example, 1 year after implementation of a prescribed fire program at the W.L. Finley National Wildlife Refuge, Willamette Valley, Oregon, Streatfeild and Frenkel (1997) found little difference in the relative proportion of native and nonnative plant species in treatment areas, regardless of fire history. In the study area, 20 plots were burned in September of the previous year; six plots were burned 4 to 6 years prior to the study; and ten plots were unburned controls. The authors concluded that prescribed fires were not (yet) achieving the management goals of reducing the cover of nonnative plant species or increasing the cover of native perennial herbaceous species (Streatfeild and Frenkel 1997). Whether the continued application of prescribed fire at the refuge will eventually favor native or nonnative species remains uncertain.

Though the initial reintroduction of fire may have a pronounced effect on community composition, individual applications of a frequent fire program may have few observable impacts and serve to maintain established community characteristics. For instance, a program of spring-applied prescribed fire was begun in 1978 and applied on a 3 to 5 year rotation on 7,500 acres (3,035 ha) of fescue grasslands and oak woodlands at Fort Lewis, Washington. The effects of 1 year (1994-95) of prescribed fire were examined within this management area. Prescribed fires were set in spring or fall under the following conditions: 50 to 68 °F (10 to 20 °C) ambient temperatures, 20 to 50 percent relative humidity, and wind speeds <3 miles/hour (4.8 km/hour). Flame heights were <3 feet (0.9 m). In the fescue grassland community, the majority of native and nonnative herbaceous species had no significant response to fire treatments. Nevertheless, the cover of one nonnative species, hairy catsear, decreased after both spring and fall burns, though its frequency did not change significantly due to dense postfire germination. Likewise, prescribed fire in the oak woodland community examined had little effect on herbaceous species. Though spring burns significantly decreased the frequency of colonial bentgrass ($P < 0.05$) and fall burns significantly decreased the frequency of Penn sedge ($P < 0.05$) (*Carex pensylvanica*), neither treatment reduced the cover of either nonnative graminoid (Tveten 1997; Tveten and Fonda 1999).

Similarly inconclusive results were obtained from a study of herbaceous species response to summer and fall burns conducted from 1985 to 1992 on the Mima Mounds Natural Area, Washington. Prescribed fires had, for the most part, mixed results with few significant effects on the frequency of nonnative or native species. The only lasting effect observed was a 3-year increase in the frequency of hairy catsear after a single summer burn. Though fall burns reduced the frequency of common velvetgrass, declines were also observed in the unburned control area, limiting interpretation of the results (Schuller 1997).

Prescribed fire is also applied to lowland prairies to control invading native and nonnative woody plants (Parker 1996; Thysell and Carey 2001b; Tveten and Fonda 1999). Though prescribed fire can reduce the spread of these species, it is not always an effective method of control. Fire exclusion allows Scotch broom to invade lowland prairies and oak woodlands of southern Washington (Tveten and Fonda 1999), and prescribed fire is commonly used to control this species in this region (Parker 2001; Tveten and Fonda 1999). Though fire has proven useful for this purpose, it must be applied frequently enough to prevent the buildup of fuels, which threaten oak overstories, but not so frequently that nonnative herbaceous species are favored over native ones (Thysell and Carey 2001b; Tveten and Fonda 1999). Scotch broom is least likely to sprout if treatments are applied during mid-summer, though care must be taken to avoid spreading mature seeds (Ussery and Krannitz 1998). Care must also be taken during dry conditions due to the volatile oils in Scotch broom foliage, which are capable of producing high-intensity fires (Huckins 2004). Though a single, severe fire can greatly reduce the cover of Scotch broom, it may also stimulate seed germination from the soil seed bank. A second, less intense fire roughly 2 or 3 years later, before Scotch broom seedlings begin to flower, is required to achieve long-term control (Agee 1996a). Spot treatment, such as using a flamethrower during winter months, can remove remaining Scotch broom seedlings that are not killed by prescribed fires (Agee 1996b).

In the prescribed fire program at Fort Lewis previously described, the effects of prescribed fires (spring and fall treatments) were examined in Scotch broom communities. Due to fuel conditions, fires were patchy, and flame heights were <6.5 feet (2 m). Fall burns caused more mortality and resulted in less sprouting of Scotch broom than spring burns. Fall fires caused a reduction in Scotch broom density and cover, but had little effect on seedling density. In contrast, spring burns reduced seedling density but had no effect on mature plants. Fall burns also reduced total fuels in Scotch broom thickets while spring burns did not. The authors concluded that several cycles of prescribed fire will be required to "restore the balanced fire regime" to Fort Lewis prairies (Tveten and Fonda 1999, p. 156).

Prescribed fire can help control the spread of Himalayan blackberry in lowland prairies but may not eliminate it from an area (Agee 1996a). In fact, fire may promote the spread of Himalayan blackberry in some prairie communities (Pendergrass and others 1998). Several burn treatments are necessary to control this species. According to a TNC management report (Soll 2004b), fire is not completely effective on its own but may be used to remove mature plants over large areas. The use of herbicides prior to burning may desiccate aboveground vegetation so that fires can take place during safe weather conditions. In addition, to ensure that fires carry, aboveground vegetation may need to be chopped or mown prior to ignition. For long-term control, burning may need to be followed by herbicide treatment, repeated burning or mowing to exhaust the soil seed banks and rhizome carbohydrate reserves, and/or planting of fast-growing or shade-tolerant native species. Prescribed fire may be most effective on slopes and in locations where grasses can help carry the fire (Soll 2004b).

Similar to the variable response of nonnative herbaceous species to prescribed fire, studies examining the effectiveness of prescribed fire for controlling nonnative woody plant species in lowland prairies have yielded mixed results. For example, prescribed fire has been shown to be largely ineffective at controlling sweetbriar rose, common pear, and nonnative blackberries in some locations. After 2 consecutive years of experimental prescribed burns in a wet Willamette Valley prairie, burned plots were associated with increased sprouting of sweetbriar rose and common pear, indicating that prescribed burns were not severe enough to kill belowground meristematic tissues of these species. Furthermore, burned plots were more invaded by Himalayan and cutleaf blackberry than unburned plots (Pendergrass and others 1998). Similarly, a program of frequent prescribed fire in a Willamette Valley refuge was deemed ineffective at controlling sweetbriar rose (Streatfeild and Frenkel 1997). In another prairie restoration experiment conducted in the Willamette Valley, the effects of 2 years of fall burns were compared with other restoration treatments in a remnant patch of wetland prairie extensively invaded by native and nonnative woody species, including Scotch broom and Himalayan blackberry. Though burning did significantly reduce the number of surviving native and nonnative shrubs ($P = 0.03$), other results were inconclusive, as the response was variable, perhaps due to variable fire severity and species' abilities to sprout after fire (Clark and Wilson 2001). Results from these studies suggest that the ecological changes caused by a century of fire exclusion, forest succession, and other human impacts are unlikely to be reversed by one or two low-severity broadcast burns (Pendergrass and others 1998).

Burning prior to the direct seeding of native plants may improve their establishment by removing accumulations of litter and destroying competing vegetation. However, burning may also increase the establishment of nonnative plants from the soil seed bank. In a seeding experiment conducted in the Willamette Valley, seeds of common native and nonnative grasses and forbs were planted in fall broadcast burned and unburned plots. Plots were located in three prairies distinguished by the relative dominance of different herbaceous vegetation types: annual nonnative grasses, perennial nonnative grasses, and native bunchgrasses. Burned seedbeds located in communities dominated by nonnative grasses had greater seedling establishment of native species (100 percent and 75 percent of species sown) than nonnative species (13 percent and 33 percent of species sown). In contrast, a greater proportion of nonnative species (50 percent) was favored by burned seedbeds than native species (25 percent) in the relatively pristine site dominated by native bunchgrasses. In general, broadcast burning prior to direct seeding of native species improved seedling establishment in low-quality, highly-invaded prairie habitats. However, broadcast burning in relatively pristine prairie communities may have created conditions that favored establishment of nonnative ruderal species present in the soil seed bank (Maret and Wilson 2000).

In conclusion, the utilization of fall burns for lowland prairie restoration is a "mixed bag" in terms of native and nonnative plant species response (Pickering and others 2000). Fire stimulates many nonnative species while controlling others and can have both negative and positive effects on native vegetation. Many nonnative species respond both favorably and unfavorably to fire, making community responses difficult if not impossible to predict. In addition, many studies have only examined the immediate impacts of recently reintroduced fire on plant communities. Extrapolation of short-term species response from one or two burns to predict long-term trends should be done with caution (Dunwiddie 2002).

A century of fire exclusion has greatly increased woody fuels in Oregon oak woodland communities. Prescribed fire may have limited usefulness to control woody vegetation, and short-term applications of fire are unlikely to reduce accumulations of woody fuels or to control species that sprout after aboveground disturbance (Pendergrass and others 1998). On some sites, repeated burning can reduce fuel loads without harming mature oaks. Alternatively, mechanical destruction and removal of woody fuels prior to burning may be necessary to reduce the risk of damaging overstory oaks (Thysell and Carey 2001b).

Careful evaluation of species composition and life history traits may help managers select between different management options for restoring some of the ecological functions of lowland prairies. The timing, frequency, and season of burning must be selected carefully to avoid damaging native species, particularly sensitive or endangered species, or promoting establishment and spread of nonnative species.

Alaska

The following three ecoregions will be discussed in this subsection: coastal hemlock-spruce forest, interior boreal forest, and tundra (classification after Küchler 1967). Currently, fire does little to contribute to the invasion of nonnative species into these plant communities. However, rapidly warming temperatures in the northern latitudes associated with global climate change, may increase fire frequency in these plant communities while simultaneously disrupting environmental barriers that currently limit nonnative plant species invasion.

Coastal Hemlock-Spruce Forests

Three distinct vegetation types occur in the coastal hemlock-spruce region of Alaska: western hemlock-Sitka spruce forests, deciduous brush thickets, and muskeg bogs. Fire regimes in this region may be characterized by major stand-replacing fires occurring every 200 years or more (Arno 2000, table 5-1, pg. 98).

Role of Fire in Promoting Nonnative Plant Invasions in Coastal Hemlock-Spruce Forests

Due to the moist conditions in the coastal hemlock-spruce region of Alaska, fires are typically rare, though more frequent in forest stands of the Kenai Peninsula which include black spruce (*Picea mariana*), white spruce (*Picea glauca*), and paper birch (*Betula papyrifera*). When fires in the coastal hemlock-spruce region do occur, they are relatively small (less than 10 acres) and tend to be located along road systems and near populated areas (Noste 1969). Nonnative plant species are largely restricted to locations that have been recently or frequently disturbed by humans (Densmore and others 2001; DeVelice, no date; Duffy 2003). These locations are also associated, coincidentally, with fire occurrence; however, fires have not been observed to promote the invasion of nonnative species into coastal hemlock-spruce communities of Alaska. Unfortunately, few fire effects studies have been conducted within coastal hemlock-spruce forest and none have been carried out in deciduous brush thickets or muskeg bogs.

In coastal hemlock-spruce forests on the Kenai Peninsula, spruce bark beetle outbreaks have killed mature spruce trees over large areas. Despite moist conditions, wildfires have burned after these outbreaks, further opening the forest canopy. As part of an inventory of nonnative plant species on the Chugach National Forest, Kenai Peninsula, Duffy (2003) examined two burns located within beetle outbreak areas and found that native herbaceous species dominated both sites. Nevertheless, two nonnative plant species that were present in unburned areas, field foxtail (*Alopecurus pratensis*) and timothy, a widespread nonnative species in the coastal hemlock-spruce region (Heutte and Bella 2003), also established within burned areas. Fall fires may encourage the spread of established populations of timothy in beetle outbreak areas, as late season fire stimulates growth, production of reproductive tillers, and increased seed production in this plant species in other geographical regions (Esser 1993a). Fire response information is unavailable for field foxtail.

Twentieth-century wildfires created favorable moose habitat in coastal hemlock-spruce forests on the Kenai Peninsula by eliminating conifer overstories and stimulating shoot production of willow, aspen, and birch (Miner 2000). Boucher (2001) evaluated the relative effectiveness of prescribed burns for creating moose habitat by examining 17 prescribed burns conducted in coastal hemlock-spruce stands on the Chugach National Forest, Kenai Peninsula. The "probably introduced" (Hultén 1968) Dewey's sedge (*Carex deweyana*) developed minor cover in 3 of the burns but was absent from paired transects located in adjacent unburned forest. However, the reverse was true in a fourth set of paired transects. This study found no statistically or ecologically significant relationship between prescribed fire and invasion by Dewey's sedge. Common dandelion, a species noted to increase in frequency after fire in the lower 48 states due to its abundant production of wind-dispersed seed (Esser 1993b), was also observed in this study but was no more abundant in prescribed burns than in adjacent unburned forest.

Nonnative plant inventories conducted within the coastal hemlock-spruce region of Alaska indicate that populations of nonnative species that may have the ecological potential to invade after fire are present in the region. Though these species have been observed to invade after fire in lower latitudes, there is no evidence to indicate whether these species will invade or increase after fire in the coastal hemlock-spruce region of Alaska. Currently, these nonnative plants are largely restricted to roadsides and populated areas within this region. For example, in an observational study of roadside vegetation along the coastal slope of the Haines Road in southeast Alaska (Lausi and Nimis 1985), four nonnative species that have been observed to tolerate or spread after fire in lower latitudes were found: yellow toadflax (*Linaria vulgaris*), common dandelion, and orchard grass (*Dactylis glomerata*) in all roadside locations regardless of vegetation type, and western wheatgrass (*Pascopyron smithii*) in deciduous thickets and muskeg bogs. Both yellow toadflax and common dandelion are established in coastal hemlock-spruce forest and boreal forest regions of Alaska (Lapina and Carlson 2005). Though yellow toadflax is more common in southcentral Alaska, it has also been identified in Juneau and Skagway (Heutte and Bella 2003). Due to its deep taproot, yellow toadflax typically survives even severe fires, and postfire environments are favorable to seedling establishment (Zouhar 2003b, FEIS review). Yellow toadflax may require an initial disturbance, such as fire, for establishment, but once a population is established yellow toadflax can spread into adjacent undisturbed locations within the coastal hemlock-spruce region (Lapina and Carlson 2005). Orchard grass has been observed throughout the coastal hemlock-spruce region of Alaska (Heutte and Bella 2003) and is reported to be somewhat tolerant of fire disturbance, perhaps even facilitating the spread of low-intensity fires when dormant (Sullivan 1992a, FEIS review). Western wheatgrass is noted to maintain or slightly increase in response to fire (Tirmenstein 1999, FEIS review). In conclusion, on some sites roadside fires within the coastal hemlock-spruce region may contribute to the invasion and establishment of nonnative species within natural areas located adjacent to roads.

In a compilation of inventory and field guide data, nonnative species observed in the coastal hemlock-spruce region of Alaska were identified and described (Heutte and Bella 2003), including several species that invade or increase after fire. Several species observed in the coastal hemlock-spruce region invade broadcast-burned clearcuts of the coastal Douglas-fir region of Washington and Oregon. For example, Canada thistle is located around human settlements in the region; this species is known to survive fire and establish in postfire environments (Zouhar 2001d). Bull thistle has been observed in Ketchikan, Haines, Gustavus, Juneau, and Prince of Wales Islands and, similar to Canada thistle, postfire environments are favorable to its establishment (Zouhar 2002b). Tansy ragwort, a species that responds favorably to a variety of disturbances including slash fires (Stein 1995), has been observed in Ketchikan and Juneau. Spotted knapweed (*Centaurea biebersteinii*), a species observed invading burned and logged land in British Columbia (Zouhar 2001c, FEIS review), has been located in Skagway, Valdez, and Prince of Wales Island. This species produces a taproot capable of surviving low-severity fire and large amounts of fire-tolerant seed (Zouhar 2001c). Scotch broom has also made inroads into the coastal hemlock-spruce region where it is found in Ketchikan, in private yards in Sitka, Hoonah, and Petersburg (Heutte and Bella 2003) and Prince of Wales Island (Lapina and Carlson 2005). A similar shrub species, gorse, has also made its way to the Queen Charlotte Islands (Heutte and Bella 2003).

Two species that increase after prescribed fire in Oregon oak woodlands of Washington and Oregon have also been identified in the coastal hemlock-spruce region. Hairy catsear has been observed along logging roads on northern Zarembo Island, in the upper Lynn Canal on Queen Charlotte Island, and in Juneau. St. Johnswort has been observed in Hoonah and Sitka (Heutte and Bella 2003).

In addition, white sweetclover is found in both the boreal forest and coastal hemlock-spruce regions of Alaska (Lapina and Carlson 2005). In the coastal hemlock-spruce region, white sweetclover has infested gravel bars and sand dunes along the Stikine River in Tongass National Forest (Heutte and Bella 2003).

In an extensive survey of nonnative plant species in Alaskan National Parks (Densmore and others 2001), three nonnative species, which literature reviews indicate are fire-adapted, were observed in both the Kenai Fjords National Park (KFNP) and Sitka National Historical Park (SNHP): common dandelion, yellow toadflax, and yellow sweetclover (*Melilotus officinalis*). As with the first two species described previously, yellow sweetclover invades disturbed areas in Alaska (Lapina and Carlson 2005). In a review published by the Alaska Natural Heritage Program (2004), the author suggests that yellow sweetclover easily invades open habitats such as those created by fire, although primary literature was not cited to support this observation. All three species are currently limited to roadsides and trails, but there is concern that future construction projects may encourage their spread (Densmore and others 2001).

Two inventories of nonnative species were conducted in Chugach National Forest, one in the mountains of the Copper River area and Kenai Peninsula (Duffy 2003) and the other restricted to trails of the Kenai Peninsula (DeVelice, no date). Nonnative plant species were confined to areas subject to human disturbance such as roads, boat ramps, trailheads (Duffy 2003), and trails (DeVelice, no date) and were rare in densely forested and alpine areas (DeVelice, no date; Duffy 2003).

FEIS literature reviews were consulted for fire effects information on nonnative species identified in the Chugach National Forest inventories. Six species are noted to either establish or increase after fire in other geographical regions (table 10-4). The remaining species in the inventories either do not respond favorably to fire, or fire effects information is unavailable. Fire response information has previously been described for three of the six species: common dandelion, timothy, and yellow sweetclover. In addition, fire stimulates the production of reproductive tillers in perennial ryegrass (Sullivan 1992b) and encourages the establishment or increase of common sheep sorrel in regions outside of Alaska (Dunwiddie 2002; Esser 1995; Pickering and others 2000; Tveten 1997). Kentucky bluegrass was among the most commonly encountered nonnative plant species in the Chugach National Forest inventories (DeVelice, no date; Duffy 2003). Kentucky bluegrass is a rhizomatous, mat-forming perennial that has been used for soil stabilization along Alaskan highways. Though established populations have been observed to displace native species and alter succession in plant communities located in other regions (Uchytil 1993), a review published by the Alaska Natural Heritage Program states that Kentucky bluegrass does not seriously alter successional development in Alaska (Alaska Natural Heritage Program 2004). The source of this information was not given.

Effects of Nonnative Plant Invasions on Fuels and Fire Regimes in Coastal Hemlock-Spruce Forests

There is no indication that fire regimes in the coastal spruce-hemlock region of Alaska have been altered by nonnative plant invasions. The climate is wet, and fire frequency and severity are probably more closely associated with rare periods of dry weather than with fuel conditions. Nevertheless, there are a few nonnative species that may have the potential to influence fire regimes in this region.

- Japanese, giant, and bohemian knotweed (*Polygonum* x *bohemicum*) are highly invasive nonnative plant species that are becoming increasingly common along streams and rivers, utility rights-of-way, and gardens in Alaska and the coastal northwest. Populations of Japanese knotweed are established in the Tongass National Forest (Alaska Natural Heritage Program 2004) and in Juneau, Sitka, and Port Alexander (Lapina and Carlson 2005). Knotweeds are well established in the Anchorage area, and there is concern that they could spread into adjacent forestland (Duffy 2003). In a review published by the Alaska Natural Heritage Program (Lapina and Carlson 2005), it is noted that Japanese knotweed may pose a fire hazard during the dormant season due to an abundance of dried leaves and stems, suggesting that dense populations of dormant plants may encourage fire spread during rare periods of abnormally dry winter weather.
- Orchard grass, a roadside weed in southeastern Alaska (Lausi and Nimis 1985), develops a dense thatch (Sullivan 1992a) that may aid fire spread.

Table 10-4—Nonnative plant species that were observed in Chugach National Forest inventories and are reported to establish or increase in response to fire in other locations.

Scientific name	Common name	Kenai Peninsula-Trails (DeVelice n.d.)	Kenai Peninsula-Mountains (Duffy 2003)	Seward area-Fjordland (Duffy 2003)	Cordova area-Foreland/Fjordland (Duffy 2003)
Lolium perenne	Perennial ryegrass	X		X	
Melilotus officinalis	Yellow sweetclover	X		X	
Phleum pratensis	Timothy	X	X	X	
Poa pratensis	Kentucky bluegrass	X	X		X
Rumex acetosella	Common sheep sorrel	X			
Taraxacum officinale	Common dandelion	X	X	X	X

- Bird vetch (*Vicia cracca*) populations occur in the Seward area of coastal Alaska (Duffy 2003) as well as Ketchikan and Unalaska (Lapina and Carlson 2005). Its potential impact on fire regimes is discussed in the boreal forest subsection that follows.

Use of Fire to Manage Invasive Plants in Coastal Hemlock-Spruce Forests

There is no indication that prescribed fire is currently being used to manage invasive nonnative plants in the coastal hemlock-spruce region of southeast Alaska. This may be due to the wet climate or the infeasibility of controlling the widely-dispersed, relatively small populations of nonnative plants found in this region. However, Heutte and Bella (2003) mention that wetland invasions of reed canarygrass, a plant of questionable nativity, may be effectively controlled with fire. Whether the authors are referring to control efforts within the coastal hemlock-spruce region is not clear, though fire is used to control this species in western Washington and Oregon.

Boreal Forests

Two general boreal forest types are represented in Alaska: black spruce (*Picea mariana*) and spruce (*P. mariana, P. glauca*)-birch (*Betula papyrifera*) forests. The most widespread boreal forest type in Alaska, black spruce forests are dense to open lowland forests composed of mixed hardwoods and black spruce or pure stands of black spruce. The fire regime of black spruce forests is characterized by major stand-replacing fires occurring approximately every 35 to 200 years (Duchesne and Hawkes 2000, table 3-1, pg. 36), and most of the plant species that occupy black spruce forest communities are adapted to fire disturbance. Interior black spruce forests of Alaska burn relatively frequently for several reasons: climatic conditions, fire-prone lichen-covered trees, and flammable organic ground cover (Lutz 1956; Viereck 1983).

Permafrost is common in black spruce forests. One of the most important ecological impacts of fire in black spruce forests is the long-term effect that fire has on soil temperatures (Dyrness and others 1986; Swanson 1996; Viereck 1973; Viereck and Dyrness 1979). Fire leads to considerable soil warming and a deepening of the biologically active layer in the soil profile that persists for several years subsequent to fire disturbance. These changes, along with the accelerated processes of decomposition and mineralization associated with fire, lead to enhanced productive capacity in postfire plant communities. In general, early-seral native species in black spruce forests have high nutrient requirements and fast growth rates that allow them to dominate the early stages of succession (Dyrness and others 1986).

In contrast, spruce-birch forests are dense interior forests composed of white spruce, paper birch, quaking aspen (*Populus tremuloides*), and poplar (*P. balsamifera*). Fire regimes in this forest type are characterized by minor mixed-severity and major stand-replacing fires with fire frequencies between <35 and 200 years (Duchesne and Hawkes 2000, table 3-1, pg. 36). Permafrost is rare in this forest type (Foote 1983).

Role of Fire in Promoting Nonnative Plant Invasions in Boreal Forests

Black spruce—Ecological studies conducted in black spruce forests do not mention the presence of nonnative plant species in postfire plant communities (Cater and Chapin 2000; Dyrness and Norum 1983; Swanson 1996; Van Cleve and others 1987; Viereck and Dyrness 1979; Viereck and others 1979). Instead, severely burned areas are quickly colonized by black spruce and several other native species with wind-borne propagules, such as fireweed (*Epilobium angustifolium*), bluejoint reedgrass (*Calamagrostis canadensis*), willow (*Salix* spp.), fire-adapted bryophytes, fire moss (*Ceratodon purpureus*) and liverwort (*Marchantia polymorpha*). Areas that burn at a lower severity are rapidly reoccupied by sprouts originating from underground parts of surviving vegetation (Dyrness and Norum 1983).

There has been a recent increase in nonnative plant species along road systems in Interior Alaska (Burned Area Response National-Interagency Team 2004; Lapina and Carlson 2005). Interior Alaskan road systems traverse a landscape of boreal forest composed of both black spruce and spruce-birch forest. Since over 100 miles (160 km) of road corridor were burned in the 2004 fire season alone, there is concern that fire and the use of bulldozers to create firelines may promote the invasion and spread of roadside nonnative plant species in black spruce and spruce-birch forest (Burned Area Response National-Interagency Team 2004). Invasive species can be transported into fire areas when bulldozers and other suppression equipment are not cleaned of soil and plant material prior to being moved, or when equipment is driven through populations of invasive species located adjacent to fire areas. In the Burned Area Emergency Stabilization and Rehabilitation Plan for the 2004 Alaska fire season, a list of priority nonnative plant species that occur either within or adjacent to burned areas was provided (table 10-5) to assist with postfire monitoring, assessment, and control (Burned Area Response National-Interagency Team 2004). Several

Table 10-5—Nonnative plant species of Interior Alaska that occurred within or adjacent to areas burned during the 2004 fire season (Burned Area Response National-Interagency Team 2004).

Scientific name	Common name
Avena fatua	Wild oats
Bromus inermis	Smooth brome
Capsella bursa-pastoris	Shepherd's purse
Centaurea cyanus	Garden cornflower
Cirsium arvense	Canada thistle
Convolvulus arvensis	Field bindweed
Crepis tectorum	Narrowleaf hawksbeard
Descurainia sophia	Flixweed tansymustard
Galeopsis tetrahit	Brittlestem hempnettle
Hieracium caespitosum	Yellow hawkweed
Lappula squarrosa	Bristly sheepburr
Lepidium densiflorum	Common pepperweed
Leucanthemum vulgare	Oxeye daisy
Linaria vulgaris	Yellow toadflax
Matricaria discoidea	Pineapple weed
Melilotus alba	White sweetclover
Melilotus officinalis	Yellow sweetclover
Phalaris arundinacea	Reed canarygrass
Plantago major	Common plantain
Rorippa sylvestris	Creeping yellowcress
Sisymbrium altissimum	Tumble mustard
Sonchus arvensis	Perennial sowthistle
Tanacetum vulgare	Common tansy
Taraxacum officinale	Common dandelion
Trifolium hybridum	Alsike clover
Vicia cracca	Bird vetch

of these species are observed to increase after fire in lower latitudes. Species previously mentioned in this chapter include Canada thistle, yellow toadflax, white and yellow sweetclovers, and common dandelion. In addition, flixweed tansy mustard (*Descurainia sophia*) can form dense populations from soil seed banks in the early stages of secondary succession after fire (Howard 2003a, FEIS review).

Fireline creation and revegetation impact both soils and native plant communities. When bulldozed firelines are constructed, organic matter is removed, leaving mineral soil exposed. Deep thawing of exposed mineral soil in firelines has resulted in extensive soil erosion in black spruce forests (DeLeonardis 1971; Dyrness and others 1986; Viereck 1973; Viereck and Dyrness 1979). While native grasses and forbs establish rapidly in burned areas without soil disturbance, vegetation does not establish readily on previously frozen, severely disturbed soils (DeLeonardis 1971). In the past, firelines have been fertilized and seeded with nonnative grasses, such as 'Manchar' smooth brome (*Bromus inermis*), creeping red fescue (*Festuca rubra*), and 'Rodney' oats (*Avena fatua*) (Bolstad 1971; Viereck and Dyrness 1979). Experimental plantings of nonnative grasses and legumes in boreal forests resulted in rapid initial growth followed by decreased cover in following years (Johnson and Van Cleve 1976).

The use of nonnative species in boreal forest and tundra revegetation projects has been controversial (Johnson and Van Cleve 1976). So far, the intentional introduction of nonnative species for fireline revegetation has not led to their long-term establishment or spread into burned areas. However, since native species such as bluejoint grass and fireweed are widely adapted and competitive in early seral environments, it may be preferable to revegetate firelines with these species. The Burned Area Emergency Stabilization and Rehabilitation Plan for the 2004 Alaska fire season specifies that only certified weed free native seed mixes will be used during postfire revegetation projects in boreal forests and the use of straw mulch for soil stabilization will be discouraged (Burned Area Response National-Interagency Team 2004).

Fire effects studies conducted in the boreal forest region of Alaska have not reported nonnative plant species establishment in postfire communities. However, plant inventories conducted in the boreal forest region record the presence of nonnative plant species observed to increase after fire disturbance in other regions. For example, in an extensive study of nonnative plant species in Alaska National Parks (Densmore and others 2001), common dandelion and white sweetclover were observed in boreal environments in Denali and Wrangell-St. Elias National Parks.

White sweetclover has invaded roadsides and river bars in the boreal forest region of Alaska. A cold-hardy cultivar of white sweetclover was seeded on highway cutbanks outside of Denali National Park and has repeatedly established in the park. It is probably transported on vehicle tires from highway plantings (Densmore and others 2001). White sweetclover has also invaded early-successional river bars in interior and south-central Alaska. In particular, it has invaded extensive acreage along the Nenena and Matanuska rivers. There is concern that white sweetclover may invade native boreal forest communities; it has been observed spreading into open areas and forest clearings in other regions (Lapina and Carlson 2005).

In a survey of roadside vegetation in the Susitna, Matanuska, and Copper River drainages, several nonnative species were identified that respond favorably to fire disturbance in other regions. Among the most frequent species observed were common dandelion and timothy (73 percent and 53 percent, respectively). In addition, white sweetclover was observed in 29 percent of survey sites. White sweetclover, timothy, and bird vetch were among the five species noted for the worst infestations observed. All three species form large, dense populations and have been observed invading native plant communities within the boreal forest region (Lapina and Carlson 2005).

In addition, several species recently introduced to the boreal forest region of Alaska were noted in this survey, including two species that invade after fire in other regions. Canada thistle was located at only one site but was recommended for immediate control by the survey's authors. Two small populations of cheatgrass (*Bromus tectorum*), a notorious fire-adapted grass that invades fire-disturbed areas and has altered fire regimes in arid habitats of the western United States (Zouhar 2003a, FEIS review), were located in southern Wasilla and Houston. Though cheatgrass often responds favorably to fire in arid environments, how this species will respond to fire in the boreal forest region is unknown.

Spruce-birch—While wildfire is less common in spruce-birch forest than in black spruce forest (Foote 1983), timber harvesting in the Alaskan interior has concentrated on the spruce-birch forest community. In an observational study that compared sites burned by wildfire with logged sites in the Tanana and Yukon River drainages of central Alaska (Rees and Juday 2002), 17 plant species were found only in burned sites in the early stages of postfire succession. One of these species was nonnative narrowleaf hawksbeard (*Crepis tectorum*), a roadside plant that produces abundant wind-dispersed seed (Lapina and Carlson 2005). However, this study neither demonstrates a statistically significant association between burned sites and the presence of narrowleaf hawksbeard nor provides evidence that fire is contributing to the invasion of this species. No other nonnative species were observed in burned plots of this study.

Effects of Nonnative Plant Invasions on Fuels and Fire Regimes in Boreal Forests

The invasion of bird vetch in both the boreal forest and coastal hemlock-spruce regions of Alaska may have the potential to alter fire regimes. A noxious weed in Alaska, bird vetch has rapidly invaded Alaska's right-of-ways since its initial establishment in the Matanuska Valley and Fairbanks area more than 20 years ago (Klebesadel 1980). It is most common in these areas but has also established in the Anchorage area, and there is concern that it could spread to adjacent forestland (Duffy 2003). It has also been observed near Denali National Park and in Seward, Kenai Peninsula (Nolen 2002). Bird vetch, a nitrogen-fixing perennial forage crop, thrives in areas of soil disturbance and is now abundant along roadsides, railroads, field edges, and abandoned fields where it climbs over bushes and small trees, such as alder and willow, and up fences to a height of 4 to 6 feet (1.2 to 1.8 meters). Bird vetch produces abundant seed that may be carried in tangled vegetation by maintenance and suppression equipment (Alaska Natural Heritage Program 2004). Observers note that fences overgrown with bird vetch alter winter wind flow, causing snowdrift accumulations (Klebesadel 1980). Though unsupported by citations from primary literature, a literature review published by the Alaska Natural Heritage Program states that bird vetch is fire tolerant (Alaska Natural Heritage Program 2004). The density and continuity of bird vetch in high use areas, coupled with its climbing habit, suggest that it might carry fire both along the ground and into shrub and tree canopies. There is also concern that forest fires could allow the movement of bird vetch into new areas (Burned Area Response National-Interagency Team 2004) where it may suppress the growth of native species (Nolen 2002).

Use of Fire to Manage Invasive Plants in Boreal Forests

Thus far, prescribed fire has only been used in control trials to manage invasive nonnative plants in the boreal forest region of Alaska (Conn, personal communication, 2005).

Tundra

Lightning-ignited fires are common in tundra vegetation, but they occur irregularly. Even though there is usually little standing fuel and organic soils tend to be moist year around, cottongrass (*Eriophorum vaginatum*) tussock communities are fire-prone (Racine 1979). Tundra communities experience major stand-replacement fires occurring at frequencies of about 35 to 200 years (Duchesne and Hawkes 2000, table 3-1, pg. 36). Tundra fires tend to be low-severity, with no vascular plant species completely eliminated by fire (Wein 1971).

Role of Fire in Promoting Nonnative Plant Invasions in Tundra

Though recovery of total primary production is usually quite rapid after fire in tundra communities (Wein and Bliss 1973), changes in relative species abundance can be long lasting (Fetcher and others 1984). Nonnative species have not been reported in postfire tundra communities (Landhausser and Wein 1993; Racine 1979, 1981; Racine and others 1987; Vavrek and others 1999; Wein and Bliss 1973).

Similar to black spruce forests, an ecologically important impact of fire in tundra communities is increased soil thaw depth, a condition that can last for more than 23 years after fire (Vavrek and others 1999). Extreme thawing results in exposure of mineral soil, which is then available to establishing species, especially those with wind-dispersed propagules. Firelines are of concern in tundra ecosystems, as they increase soil thaw depths. One year after firelines were constructed to help suppress lightning-caused fires in tundra communities on the Seward Peninsula, firelines had thaw depths of more than 20 inches (50 cm) on the burned side and more than 14 inches (35 cm) on the unburned side (Racine 1981).

The revegetation of disturbed tundra communities after fire and fireline creation has usually been accomplished by seeding northern varieties of commercially available nonnative grasses. While these nonnative grasses may reduce erosion, they do not establish permanent cover in tundra ecosystems without frequent fertilization (Chapin and Chapin 1980). In study plots that examined the effects of bulldozing in the Eagle Creek area of interior Alaska, nonnative grasses commonly used in tundra restoration (table 10-6) were seeded with and without fertilizer. Three years after sowing, the grasses did not interfere with the establishment of native sedges. After 10 years, plot vegetation was composed entirely of native species (Chapin and Chapin 1980).

While these results indicate that native graminoids are superior competitors in tundra ecosystems, environmental changes anticipated with global warming (Chapin and others 1995) may disrupt this advantage. Therefore, it is possible that "…the use of exotic species which have been selected for their performance under arctic conditions maximizes the possibility that an introduced grass or weed will establish in the community…" (Chapin and Chapin 1980, p. 454). Revegetation of disturbed sites, such as firelines, may be accomplished effectively with native sedges and forbs while simultaneously avoiding the introduction of potentially invasive nonnative species into the tundra ecosystem.

Effects of Nonnative Plant Invasions on Fuels and Fire Regimes in Tundra

While there is no indication that nonnative species are changing tundra fire regimes, fire regime and climatic changes initiated by global climate warming could influence the susceptibility of tundra plant communities to invasion by nonnative species. The effects of global warming are expected to be particularly pronounced in northern latitudes (Chapin and others 1995). In a spatially explicit model of vegetation response to warming climate on Seward Peninsula (Rupp and others 2000), a 3.6 °F (2 °C) temperature increase was associated with increased flammability of tundra vegetation, increased fire frequency, fires of greater spatial extent, and gradual expansion of spruce-birch forest into previously treeless tundra communities and/or conversion of tundra vegetation to grassland steppe. In an 11-year manipulated experiment conducted in a moist tundra community near Toolik Lake, located in the northern foothills of the Brooks Range of Alaska, environmental manipulations simulating global warming (increased air and soil temperatures, decreased light availability, increased nutrient mineralization and decomposition) resulted in decreases in species richness and shifts in community composition, including increased dominance of birch (*Betula* spp.) and decreased dominance of evergreen shrubs and understory forbs (Chapin and others 1995). Since tundra communities have few species to begin with, species loss coupled with increasing temperatures, soil fertility, and fire frequency may have dramatic ecosystem consequences, including increased vulnerability to invasion by nonnative species. It would be prudent under such uncertain climatic conditions to take steps to prevent the introduction of nonnative plant species to this region.

Table 10-6—Nonnative grasses used in tundra restoration and examined by Chapin and Chapin (1980).

Scientific name	Common name
Alopecurus pratensis	Meadow foxtail
Festuca rubra	Red fescue
Lolium perenne ssp. *perenne*	Perennial ryegrass
Phalaris arundinacea	Reed canarygrass
Phleum pratense	Timothy
Poa pratensis	Kentucky bluegrass

Use of Fire to Manage Invasive Plants in Tundra

Nonnative species are not well established in the tundra region; therefore, there is no need for control efforts such as prescribed fire at this time.

Summary of Fire-Invasive Plant Relationships in Alaska

In Alaska today, nonnative plants are largely restricted to areas heavily impacted by human activities. With the construction of roads, trails, and firelines in pristine native plant communities, the threat of nonnative plant species establishment and spread increases. Ecological barriers to nonnative species establishment may weaken with future climatic changes and should not be relied upon to slow the invasion of nonnative plants into areas disturbed by fire or human activities. To improve the knowledge base about which species are likely to invade after fires in Alaska, fires that intersect anthropogenic disturbances such as roads and firelines should be monitored for several years after burning. In order to ensure that nonnative species do not establish in Alaskan plant communities, development and fire suppression activities must be conducted in a careful manner that precludes the inadvertent introduction of nonnative plant propagules. Because of the frequency and size of fires in Alaska, existing infestations of fire-adapted species should be controlled to decrease the dispersal opportunities along fire perimeters. Revegetation with competitive, early-seral native species after fire and soil disturbance is also important, both for reducing soil thaw and erosion and for the prevention of nonnative species establishment.

Conclusions

Throughout much of the Northwest Coastal bioregion, fires are more closely tied to varying weather conditions than to fuel conditions (Agee 1997). Therefore, changes to local climate are likely to modify regional fire regimes. Paleoecological data, as reviewed by McKenzie and others (2004), indicate that periods of warmer temperatures and decreased precipitation have been associated with increased fire frequency and decreased fire severity in this region. Though future climatic conditions are difficult, if not impossible, to predict, global circulation models indicate that fire seasons (measured by degree-days, temperature, and drought indices) may lengthen throughout the Pacific Northwest over the next century. Fires may burn earlier and later in the year than they do now and total area burned may increase. In addition, carbon dioxide fertilization may contribute to fuel production, increasing potential fire severity. Warmer temperatures may reduce winter snow pack in the mountains, leading to increased moisture stress, insect and disease outbreaks, and fuel loading in montane forests, which could result in more frequent and severe fires in Northwest coastal forested communities (as reviewed by McKenzie and others 2004).

Given the uncertainties regarding future climatic conditions and fire regimes, fire management techniques should be developed that avoid transporting or facilitating the movement of nonnative plant propagules between different environments. Nonnative plants that are currently not invasive in particular local plant communities may become so in the future. Regionally organized programs of native seed collection, propagation, and storage for postfire restoration projects will help discourage the seeding of nonnative plants.

Montane communities of Washington and Oregon and coastal hemlock-spruce forests, boreal forests, and tundra communities of Alaska have relatively few established populations of nonnative plants, providing land managers with an opportunity to prevent the spread of these species into intact natural communities. Of these communities, boreal forests are the most vulnerable to the spread of fire-adapted nonnative species due to the frequency and scale of fire in this region. The best way to prevent future expansion of these species is through early detection and rapid control response. Whether nonnative species are absent from these ecosystems due to ecological barriers such as climate or to a lack of a local seed source is unclear, though the presence of increasing numbers of nonnative species in Alaska suggests the latter. If environmental conditions are indeed preventing nonnative plant invasions into high elevation and high latitude environments, global climate change could remove existing ecological barriers to species establishment. Therefore, preventing the introduction of nonnative plant propagules into these communities is also critically important. Human activities, such as fire suppression, that inadvertently or intentionally introduce nonnative plants into these communities may cause irreparable harm.

Few studies have examined the effects of natural fire on upland and riparian forests of the coastal Douglas-fir region of Washington and Oregon. Examination of short- and long-term changes in plant community composition and structure that follow natural fires should be a research priority, particularly with regard to invasion and establishment of nonnative plant species. In remote wildlands, the ecological impacts of natural fire need to be weighed against those of fire exclusion, particularly fire management activities that promote the invasion of nonnative species, such as fireline construction and postfire seeding.

Fire suppression activities may encourage the invasion of riparian communities by nonnative species. The building of firelines can increase runoff and soil erosion, facilitating the invasion of riparian communities through the delivery of soil and seed from upland communities. Firelines built through riparian forests and down the fall lines of steep slopes are especially damaging (Beschta and others 2004).

In the dense coastal Douglas-fir forests of the Northwest Coastal bioregion, the ecological impact of nonnative plant populations is currently restricted to the earliest stages of forest succession that follow logging and slash fires. Ruderal nonnative forbs, such as Canada thistle, are displacing native early seral vegetation in some locations and reducing tree regeneration in others. Though nonnative plants are typically eliminated from the plant community after a few years of forest stand development, nonnative shade-tolerant plant species are capable of persisting and/or invading forest understories if relatively open stand conditions are maintained through clearcutting, silvicultural thinning, or prescribed underburning. In particular, false brome poses a serious threat to forest understory communities and may affect fire behavior and spread. Research is needed to determine how invasive this species is in shaded forest understories, how it responds to natural and prescribed fire, and how its foliage and population characteristics might influence fuel characteristics and fire behavior.

In contrast, Oregon oak woodland communities of western Washington and Oregon are already extensively invaded by nonnative plant species. However, the populations of individual plant species, both native and nonnative, are distributed heterogeneously across the landscape. Due to the diversity in plant community composition, responses to prescribed fire are highly variable and site-specific. Disturbance regimes that effectively achieve management goals will need to be developed from long-term localized research and observation. In most situations, fire used alone will not be as effective as fire used in conjunction with other management techniques such as the seeding of native plants.

Notes