Chapter 15

Exotic Invasive Plants

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Ecosystems worldwide are threatened by nonnative plant invasions that can cause undesirable, irreversible changes. They can displace native plants and animals, out-cross with native flora, alter nutrient cycling and other ecosystem functions, and even change an ecosystem's flammability (Walker and Smith 1997). After habitat loss, the spread of exotic species is considered the greatest threat to imperiled species in the United States (Flather et al. 1994; Wilcove et al. 1998; Stein et al. 2000). Many exotic invasives thrive in North America because they were introduced from other continents without natural controls, such as insect predators, plant pathogens, fungi, competing plants, and herbivores (Sheley et al. 1999). In addition, some introduced plants outcompete native species by producing allelopathic compounds that inhibit the growth of other species (Callaway and Aschehoug 2000).

In the Southwest, some exotic species respond particularly well to disturbances such as those created by forest restoration treatments. Forest thinning and its associated activities (e.g., skidding, landing construction, and slash pile burning), as well as fire, induce changes in understory composition and can promote the spread of invasive species. Severe disturbances that open forest canopies and expose mineral soil provide habitat for opportunistic species, some of which are both exotic and invasive (Honnay et al. 2002). These conditions are associated not only with restoration treatments, but also with severe wildfires. In one northern Arizona study, species richness and abundance of exotic forbs were higher within the first five years following disturbance in areas where wildfires killed more than 90 percent of tree crowns than on unburned, thinned-only, and thinned and burned plots (Griffis et al. 2001). Exotic species documented by Crawford et al. (2001) attained their highest cover values where high-intensity fires had killed most trees, intermediate levels on moderately burned areas where most trees survived, and lowest levels (minute amounts) on unburned sites.

Still unclear, though, are the long-term implications of these newcomers, especially in light of plans to extensively apply restoration or fuel-reduction treatments throughout southwestern forests. Managers face a dilemma here. Without thinning, large forest tracts may burn severely, and invasive species may spread. But they may also spread in response to restoration activities. In some cases, especially in urban–wildland interface areas, restoration treatments will effectively break buffer zones that separate areas infested with noxious species from those that are not, creating much larger expansion fronts for invasive plants. Managers who conduct restoration or fuel-reduction treatments without considering the presence and phenology of invasive plants may trade unnaturally high densities of trees and fuels for noxious plant infestations. Therefore, such treatments must be combined with judicious practices to monitor and contain their spread. In this chapter we hope to promote such practices by providing an overview of the exotic and potentially invasive plant species found in southwestern ponderosa pine forests and adjacent piñon-juniper woodlands; providing examples of species designated as "noxious," with widely differing life history strategies, that have the potential to be persistent and ecologically significant in restoration areas; and proposing ways in which this information can be used to plan and apply restoration treatments.

Major Species of Concern

The list of exotic plant species that have appeared in southwestern forests in recent years is extensive (Tables 15.1, 15.2). Many are seral opportunists that readily colonize recently disturbed sites. Some native species also occupy this ecological niche, but studies indicate that species introduced to the Southwest or United States in recent years are increasingly playing this role (Crawford et al. 2001). Given that it is not possible or even desirable to control all nonnative species, managers must focus on "invaders," or species vigorous, persistent, prolific, and widespread enough to cause serious economic and environmental impacts (Vitousek et al. 1996; Novak and Mack 2001). In this chapter, we focus on a subset of invasive exotics in the region, those legally designated as "noxious" by state statutes (sensu Skinner et al. 2000; Table 15.2). Many of these species were introduced accidentally, but a few were intentionally introduced as ornamentals, windbreaks, or forage

TABLE 15.1.

A sampling of plant species not native to the United States that have been found in southwestern ponderosa pine ecosystems.

SCIENTIFIC NAME[1]	COMMON NAME
Agropyron cristatum (L.) Gaertn.	Crested wheatgrass
Agropyron desertorum (Fisch. ex Link) J.A. Schultes	Desert wheatgrass
Alopecurus geniculatus L.	Water foxtail
Bromus inermis Leyss.	Smooth brome[2]
Bromus japonicus Thunb. ex Murr.	Japanese brome
Bromus rubens L.	Red brome
Cerastium fontanum Baumg. ssp. *vulgare* (Hartman) Greuter & Burdet	Big chickweed
Ceratocephala testiculata (Crantz) Bess.	Curveseed butterwort
Chenopodium album L.	Lambsquarters[2]
Chenopodium botrys L.	Jerusalem oak goosefoot
Coronilla varia L.	Purple crownvetch
Dactylis glomerata L.	Orchardgrass
Dianthus armeria L.	Deptford pink
Festuca ovina L.	Sheep fescue
Lactuca serriola L.	Prickly lettuce
Leonurus cardiaca L.	Common motherwort
Lolium perenne L.	Perennial ryegrass
Lotus corniculatus L.	Birdfoot deervetch
Malva neglecta Wallr.	Common mallow
Marrubium vulgare L.	Horehound
Medicago lupulina L.	Black medick
Medicago sativa L.	Alfalfa
Melilotus alba Medikus	White sweetclover
Melilotus officinalis (L.) Lam.	Yellow sweetclover
Mollugo cerviana (L.) Ser.	Threadstem carpetweed
Myosotis stricta Link ex Roemer & J.A. Schultes	Strict forget-me-not
Opuntia ficus-indica (L.) P. Mill.	Tuna cactus
Phleum pratense L.	Timothy
Poa compressa L.	Canada bluegrass
Poa pratensis L.	Kentucky bluegrass[2]
Polygonum aviculare L.	Prostrate knotweed
Polygonum convolvulus L.	Black bindweed
Polygonum nepalense Meisn.	Nepalese smartweed
Rubus discolor Weihe & Nees	Himalayan blackberry
Rumex acetosella L.	Common sheep sorrel
Rumex crispus L.	Curly dock
Sanguisorba minor Scop.	Small burnet
Sisymbrium altissimum L.	Tall tumblemustard

TABLE 15.1. *(continued)*

SCIENTIFIC NAME[1]	COMMON NAME
Sisymbrium irio L.	London rocket
Sonchus asper (L.) Hill	Spiny sowthistle
Taraxacum officinale G.H.Weber ex Wiggers	Common dandelion[2]
Thinopyrum intermedium (Host) Barkworth & D.R. Dewey	Intermediate wheatgrass
Thinopyrum ponticum (Podp.) Z.-W. Liu & R.-C. Wang	Rush wheatgrass
Thlaspi arvense L.	Field pennycress
Tragopogon dubius Scop.	Yellow salsify
Trifolium repens L.	White clover
Triticum aestivum L.	Common wheat

Sources: From Beaulieu 1975, Oswald 1981, Foxx 1996, Crawford et al. 2001, Griffis et al. 2001, Waltz et al. 2001.

Note: Other plants that are classified as nonnative *to the region* by some sources include Cuman ragweed (*Ambrosia psilostachya*), prostrate pigweed (*Amaranthus albus*), fetid goosefoot (*Chenopodium graveolens*), Over's goosefoot (*Chenopodium overi*), Canadian horseweed (*Conyza canadensis*), blunt tansymustard (*Descurainia obtusa*), flatspine stickseed (*Lappula occidentalis*), common pepperweed (*Lepidium densiflorum*), Norwegian cinquefoil (*Potentilla norvegica*), Macoun's cudweed (*Pseudognaphalium macounii*), and bigbract verbena (*Verbena bracteata*).

[1]Scientific nomenclature follows USDA Natural Resource Conservation Service (2002)

[2]Both introduced and native.

crops. In addition to designated noxious species, a number of other species may be invasive too, including intentionally introduced grasses such as Kentucky bluegrass (*Poa pratensis*) and smooth brome (*Bromus inermis*) (Weaver et al. 2001). An understanding of the various life history strategies of successful invaders, whether designated as noxious or not, can help in designing strategies to control other invasive species.

Annuals

Many annual plants readily invade recently disturbed sites in the region, but only a few persist and even expand into undisturbed areas. These include several designated as noxious, including Russian thistle (*Salsola tragus*), medusahead (*Taeniatherum caput-medusae*), yellow starthistle (*Centaurea solstitialis*), and cheatgrass (*Bromus tectorum*). Russian thistle is so ubiquitous throughout the West that many people do not recognize it as an introduced species (Tellman 1997). It occurs on some restoration sites, and was common following high-severity wildfires (Crawford et al. 2001). Medusahead and yellow starthistle are not presently very common in the region but occur along county roads and national forests in northern

TABLE 15.2.

Invasive species of southwestern ponderosa pine communities legally designated as "noxious" in one or more states.

SCIENTIFIC NAME[1]	COMMON NAME	ARIZONA[2]	COLORADO[3]	NEW MEXICO[4]	UTAH[5]	LIFE HISTORY STRATEGY
Acroptilon repens (L.) DC.	Russian knapweed, Hardheads	RES	NW	Class B	NW	long-lived perennial
Aegilops cylindrica Host	Jointed goatgrass	RES	NW	Class C	–	winter annual
Alhagi maurorum Medik.	Camelthorn	RES	NW	Class A	–	long-lived perennial
Bromus tectorum L.	Cheatgrass, Downy brome	–	NW	–	–	annual
Cardaria draba (L.) Desv.	Whitetop, Hoary cress, Globe-podded hoary cress	RES	NW	Class A	NW	perennial
Carduus nutans L.	Musk thistle, Nodding plumeless thistle	–	NW	Class B	NW	biennial
Centaurea biebersteinii DC.	Spotted knapweed	RES	NW	Class A	NW	short-lived perennial
Centaurea diffusa Lam.	Diffuse knapweed	RES	NW	Class A	NW	usually biennial
Centaurea iberica Trev. ex Spreng.	Iberian starthistle	PRH	–	–	–	short-lived perennial
Centaurea solstitialis L.	Yellow starthistle	RES	NW	Class A	NW	annual
Centaurea triumfettii All.	Squarrose knapweed	PRH	NW	–	NW	short-lived perennial
Chorispora tenella (Pallas) DC.	Blue mustard, Crossflower	–	NW	–	–	annual
Cirsium arvense (L.) Scop.	Canada thistle	PRH	NW	Class A	NW	perennial
Cirsium vulgare (Savi) Ten.	Bull thistle	–	NW	Class B	–	biennial
Conium maculatum L.	Poison hemlock	–	NW	Class B	–	biennial

TABLE 15.2. *(continued)*

SCIENTIFIC NAME[1]	COMMON NAME	ARIZONA[2]	COLORADO[3]	NEW MEXICO[4]	UTAH[5]	LIFE HISTORY STRATEGY
Convolvulus arvensis L.	Field bindweed	REG	NW	Class C	NW	perennial
Cynoglossum officinale L.	Houndstongue, Gypsy flower	–	NW	–	–	biennial
Descurainia sophia (L.) Webb ex Prantl	Flixweed, Herb sophia	–	NW	—	–	annual or biennial
Elaeagnus angustifolia L.	Russian olive	–	–	Class C	–	perennial (tree)
Elymus repens (L.) Gould	Quackgrass	RES	NW	–	NW	perennial, rhizomatous
Erodium cicutarium (L.) L'Hér. ex Ait.	Redstem filaree	–	NW	–	–	annual or biennial,
Euphorbia esula L.	Leafy spurge	PRH	NW	Class A	NW	long-lived perennial
Hypericum perforatum L.	Common St. John's wort	–	NW	–	–	perennial, rhizomatous
Isatis tinctoria L.	Dyer's woad	RES	NW	Class A	NW	winter annual, biennial, or short-lived perennial
Lepidium latifolium L.	Perennial pepperweed	–	NW	Class A	NW	perennial
Leucanthemum vulgare Lam.	Ox-eye daisy	–	NW	–	–	perennial
Linaria dalmatica (L.) P. Mill. ssp. *dalmatica*	Dalmatian toadflax	RES	NW	Class A	–	perennial
Lythrum salicaria L.	Purple loosestrife	PRH	NW	Class A	NW	perennial, rhizomatous
Onopordum acanthium L.	Scotch thistle, Scotch cottonthistle	RES	NW	Class A	NW	biennial

Portulaca oleracea L.	Common purslane, Little hogweed	REG	–	–	–	annual
Potentilla recta L.	Sulfur cinquefoil	–	NW	–	–	perennial
Salsola tragus L.	Russian thistle	–	NW	–	–	annual
Salvia aethiopis L.	Mediterranean sage	–	NW	–	–	biennial
Saponaria officinalis L.	Bouncing bet	–	NW	–	–	perennial
Taeniatherum caput-medusae (L.) Nevski	Medusahead	–	–	–	NW	winter annual
Tamarix ramosissima Ledeb.	Tamarisk	–	NW	Class C	–	perennial (tree)
Ulmus pumila L.	Siberian elm	–	–	Class C	–	perennial (tree)
Verbascum thapsus L.	Common mullein	–	NW	–	–	biennial

[1]Scientific nomenclature follows USDA Natural Resource Conservation Service (2002).

[2]Arizona weed classes: REG = regulated noxious weeds, or exotic plant species "well established and generally distributed in the state"; RES = restricted noxious weeds, or exotic plant species that "occur in Arizona in isolated infestations or very low populations"; PRH = prohibited noxious weeds, exotic plant species that "do not occur in Arizona" (Plant Service Division. 1996. Arizona Noxious Weed Regulation. Arizona Department of Agriculture.).

[3]Colorado weed classes: NW = noxious weed, defined as "plant species not indigenous (non-native) to the state of Colorado that meets at least one of several criteria regarding their negative impacts upon crops, native plant communities, livestock, and the management of natural or agricultural systems" (Colorado State Code. 1996. Colorado Noxious Weed Act. State of Colorado.).

[4]New Mexico weed classes: Class A = species are not native to an ecosystem and have a limited distribution within the state; Class B: species not native to the ecosystem and are presently limited to a particular area of the state; Class C: not native to the ecosystem, yet are widespread throughout the state (Office of the Director/Secretary. 1998. New Mexico Noxious Weed List. New Mexico Department of Agriculture.).

[5]Utah defines a noxious weed (NW) to include "any plant the commissioner (of Agriculture and Food, or the commissioner's representative) determines to be especially injurious to public health, crops, livestock, land or other property" (Utah Department of Agriculture. 1994. Utah Noxious Weed Act.).

Arizona (EnviroSystems Management 2001). Cheatgrass is common on restoration treatments in the region, especially in the Uinkaret Mountains in northwestern Arizona (Waltz et al. 2001; Chapter 12). It was present on burned sites in Mesa Verde National Park in southwestern Colorado (Floyd et al. 2001) and in areas burned by wildfires in northern Arizona (Crawford et al. 2001), and may expand as the scale and frequency of restoration treatments increase (Pierson and Mack 1990).

A number of characteristics make cheatgrass an especially successful invader. It is a cool-season species that germinates and extends its roots at lower temperatures than many other species, and once established can decrease both growth and survival of native species, such as bluebunch wheatgrass (*Agropyron spicatum*) (Harris 1967). It can even outcompete many otherwise competitive plants, including Russian thistle (Piemeisel 1951). Given adequate moisture, cheatgrass can produce two generations in one growing season, one early in the spring and one late in the fall (Mosley et al. 1999). In addition, a single cheatgrass plant may produce up to 5,000 seeds (Young et al. 1987) whose long awns attach to animal hair or feathers and to socks, shoes, and vehicles. Seeds usually remain viable only for a year or so, but if stored in dry conditions, as in hay bales, may remain viable for several years (Mosley et al. 1999).

Cheatgrass also has high ecological amplitude. In the Great Basin region, it proliferates on disturbed sites (Hull and Pechanec 1947) but also thrives in areas that have never been cultivated or grazed (West 1991). It lives both in desert communities receiving only 5.5 inches (14 centimeters) of annual precipitation and in high-elevation coniferous forests with over 22 inches (56 centimeters) of precipitation (Mosley et al. 1999).

Finally, cheatgrass can alter invaded sites in several ways. First, mature plants add organic matter to the soil surface, enhancing germination and establishment of young cheatgrass plants (Evans and Young 1970). Second, cheatgrass invasion can alter soil chemistry, leading to decreased nitrogen availability and altered species composition (Evans et al. 2001). Finally, cheatgrass changes site flammability and therefore the frequency of burning. This is particularly well documented in sagebrush steppes, where large stands of cheatgrass now facilitate a burning frequency of every 5 years or so, instead of historic fire intervals ranging from 25 to 100 years (Wright and Bailey 1982; Pellant and Reichert 1984). This increase is especially detrimental to nonsprouting shrubs such as big sagebrush (*Artemisia tridentata*), but can also kill other perennial grasses and forbs (Pellant 1990).

Effective control of annuals such as cheatgrass is possible only if small populations are eradicated, treatments are timed to prevent seed set, and perennial species are well established to slow reinvasion (Mosley et al. 1999). Once large stands of cheatgrass are established, management becomes more complicated and less successful. Treatment options include hand-pulling, mowing, grazing, burning, herbicides, and reseeding. Hand-pulling and mowing are feasible only for small patches, and only if plants are treated before seeds develop. Burning may reduce germination rates (Whisenant and Uresk 1990) but may enhance seed production per plant (Young 1983). Burning in the fall, followed by herbicide application or livestock grazing before seed set the following year, can be used to reduce vigor and seed set rates of remaining cheatgrass plants (Bunting et al. 1987; Rasmussen 1994).

Repeated applications of herbicides such as paraquat, glyphosate, atrazine, and sulfometuron have been used to treat cheatgrass (Mosley et al. 1999). When integrated with reseeding programs that establish a competitive perennial plant community, these herbicides show promise. Introduced perennial grasses have been used in most reseeding programs (Mosley et al. 1999), but interest in finding native species that compete effectively with cheatgrass is increasing. Use of a headsmut pathogen for biocontrol of cheatgrass also shows potential (Meyer et al. 2000).

Biennials

A number of exotic invasive species are biennials, plants that live through two growing seasons. Two are designated as noxious in all four states (Table 15.2) and are of high priority for treatment: diffuse knapweed (*Centaurea diffusa*) and Scotch thistle (*Onopordum acanthium*). Diffuse knapweed is present along county roads in the region (EnviroSystems Management 2001) but has not been reported on forest restoration treatments. Based on its affinity for disturbed habitats and dominance in successional ponderosa pine forests in the Pacific Northwest (Roché and Roché 1999), it will likely spread into restoration treatments in the Southwest. Scotch thistle has been observed on restoration treatments in the Uinkaret Mountains (Waltz et al. 2001). Other biennials that are listed as noxious in at least one state in the region include houndstongue (*Cynoglossum officinale*), Mediterranean sage (*Salvia aethiopis*), common mullein (*Verbascum thapsus*), musk thistle (*Carduus nutans*), and bull thistle (*Cirsium vulgare*); the latter three have occurred on burned and logged sites in the region.

Common mullein has been documented both as a minor and inconsistent component of postfire vegetation (Sackett et al. 1994; Foxx 1996) and as a major species immediately following burning and other restoration treatments (Ffolliott et al. 1977; Waltz et al. 2001). Mullein produces many seeds that remain viable up to thirty-five years (Darlington and Steinbauer 1961); it was the most common seed quantified in seed bank studies in northern Arizona (Springer 1999; Korb 2001b). However, it is often a transient species that decreases successionally as bare ground and light availability decline (Gross 1980). Ffolliott et al. (1977) documented shifting herbage production in ponderosa pine stands burned in the fall, from mostly mullein a year after fire to a mixture of native grasses and forbs eleven years later. Mullein is not currently considered as high a priority for treatment as other, more persistent biennials.

Bull thistle occupies severely disturbed sites in southwestern forests, such as log landings where the soil has been mixed or logging slash has been burned (Korb 2001b). Musk thistle only recently invaded Colorado, where populations expanded dramatically in severely burned piñon-juniper woodlands after a large 1996 wildfire in Mesa Verde National Park (Floyd et al. 2001). Scotch thistle has been observed in moist sites on restoration treatments in the Uinkaret Mountains (Waltz et al. 2001).

All three of these biennial thistles reproduce exclusively from seeds. First-year plants are rosettes; these develop into flowering plants the second year. A single bull or Scotch thistle can produce up to 50,000 seeds, an individual musk thistle twice that many (Beck 1999). Few bull thistle seeds survive as long as five years in the soil (Roberts and Chancellor 1979), but musk thistle seeds may remain viable for a decade or more (Burnside et al. 1981). The leaves of all three thistles are armed with spines, thus making them unlikely forage for livestock or wildlife.

The presence of these thistles raises at least three concerns. First, they may inhibit other species. Musk thistle releases soluble compounds that inhibit other plants and apparently stimulate recruitment of its own seedlings (Wardle et al. 1993). Bull thistle produces copious litter that leads to short-term nitrogen demand from decomposing microorganisms and deprives other plants of nitrogen (de Jong and Klinkhammer 1985). In one study, ponderosa pine seedlings within 6.6 feet (2 meters) of bull thistles had 25 to 33 percent lower growth rates than seedlings without nearby thistles (Randall and Rejmanek 1993). Second, because these biennial thistles have habitat affinities similar to those of some native plants, they have the potential to outcompete species such as Wheeler's thistle (*Cirsium* *wheeleri*) (Epple 1995), as well as rare species such as Mogollon thistle (*Cirsium parryi* ssp. *mogollonicum*) or Arizona bugbane (*Cimicifuga arizonica*) (DeWald and Phillips 1996). Mogollon thistle consists of only a few scattered populations and therefore is particularly vulnerable (Schaack and Goodwin 1990). Finally, bull thistle has the potential to hybridize with native *Cirsium* thistles, condemning them to what Simberloff (2001) calls "genetic extinction."

Control of biennial thistles must focus on preventing seed set and providing a competitive ground cover to discourage reinvasion (Beck 1999). Severing the root below the soil surface will kill these thistles. Therefore, hand-pulling or -hoeing can be used to control small patches, provided that plants are treated before they bolt. The presence of competitive vegetation can also increase the effectiveness of hand-hoeing musk thistles (Floyd et al. 2001). Mowing may reduce seed set, but some plants recover and produce seeds regardless of when they are mowed (McCarty and Hatting 1975). Therefore, multiple mowings during the growing season may be required to eliminate seed set. A number of herbicides have also proven effective, including clopyralid, dicamba, picloram, chlorsulfuron, and metsulforon (Beck 1999). The choice of herbicide and appropriate use rate depends on the growth stage and stand density of the thistles, associated plant species, and other environmental factors. A few insects have been approved as biological control agents for biennial thistles; most of these prefer musk thistle (Beck 1999). Unfortunately, one seedhead weevil (*Rhinocyllus conicus*) introduced before current screening methods were in place also attacks native thistles, including some that are endangered (Louda et al. 1997).

Perennials

Invasive herbaceous perennials are a serious concern in forest restoration. A number of woody perennials have been introduced to the region. Tamarisk (*Tamarix ramosissima*), Russian olive (*Elaeagnus angustifolia*), and Siberian elm (*Ulmus pumila*) are extremely successful invaders of riparian areas and road ditches, but are rare in drier forests. In contrast, Canada thistle (*Cirsium arvense*), leafy spurge (*Euphorbia esula*), Dalmatian toadflax (*Linaria dalmatica*), spotted knapweed (*Centaurea biebersteinii*), and Russian knapweed (*Acroptilon repens*) are considered high-priority species for control, due to their ability to persist and expand into undisturbed forests. Of these species, Dalmatian toadflax and Canada thistle have been observed on burned areas in the region (Sackett et al. 1994; Crawford et al. 2001; Floyd et al. 2001). Dalmatian toadflax was present both in pretreatment seed banks and in posttreatment vegetation following slash burning in northern Arizona (Korb 2001b). The remaining species have not been documented on restoration treatments but are present throughout the region and may expand into burned or logged forest sites. Field bindweed (*Convolvulus arvensis*) has been documented in areas burned by wildfires (Crawford et al. 2001) and on restoration treatments in the Uinkaret Mountains (Waltz et al. 2001). However, it is usually associated with continual severe soil disturbances, such as those in agricultural fields (Whitson et al. 1991), and therefore is not considered of high priority for control on restoration sites.

Several of these species can spread quickly because of their extensive lateral roots. Russian knapweed, Dalmatian toadflax, Canada thistle, and leafy spurge all have rapidly spreading root systems, and new shoots or roots can quickly arise from root buds in response to mowing, burning, or hand-pulling. Canada thistle roots can extend outward as much as 20 feet (6 meters) in one growing season (Morishita 1999). In addition, many of these perennials produce large numbers of seeds that can persist in the seed bank for several years. A single Dalmatian toadflax plant can produce up to 500,000 seeds, some of which can remain dormant for at least ten years (Robocker 1970).

Many perennial noxious species have a wide habitat range and can persist in relatively undisturbed communities. Canada thistle readily invades road ditches and other disturbed sites, but is also found in undisturbed riparian habitats (Stachon and Zimdahl 1980). Leafy spurge invades disturbed sites but is also found in areas with little or no human disturbance, such as national parks and Nature Conservancy preserves in Montana, the Dakotas, and Minnesota (Randall 1997). Even small natural disturbances can provide habitat for perennial invaders such as Dalmatian toadflax, and these small source patches then promote continued spread. Heavy grazing, or other disturbances that remove residual perennial plants, can also aid their spread (Whitson 1999).

The superior competitiveness of these species stems from several adaptations. First, extensive root systems allow them to compete effectively for water and nutrients. Second, some have attributes that protect them from grazing. Leafy spurge contains a milky juice that causes irritation or, very rarely, even death to cattle (Kingsbury 1964). Russian knapweed is toxic to horses, and other livestock and wild herbivores will not eat it because of its extremely bitter taste (Young et al. 1970). The leaf spines of Canada thistle make it unlikely forage (Morishita 1999). Third, some species, such as Russian knapweed, contain allelopathic substances that hinder the growth of some other species (Stevens and Merrill 1985). These attributes assist in the formation of dense, monotypic stands (Belcher and Wilson 1989). These species also may hybridize with native species. For example, Canada thistle has the potential to out-cross with native *Cirsium* species, and leafy spurge may out-cross with native spurges. Arizona alone supports more than three dozen species of native *Euphorbia* (Epple 1995); out-crossing with leafy spurge could eliminate some.

A successful strategy for controlling such perennials must include two parts: first, killing or removing the aboveground portion of the plants for several consecutive years before flowering to prevent seed set; second, establishing competitive plants to prevent reinfestations. Because these perennials sprout vigorously from the roots following damage, any onetime application will not control them. Several consecutive years of herbicide application, mowing, or pulling may be required to kill the plants. Even cultivation may not control some perennial invaders unless the roots are removed from the site; root fragments of Canada thistle as small as 0.5 inches (1.25 centimeters) long can resprout (Hamdoun 1972). Because of their ability to spread dramatically once established and because they are so difficult to kill, it is critical to treat small infestations of these species immediately and aggressively. Control is very difficult once they are allowed to spread.

Hand-pulling, mowing, and sheep or goat grazing, when carried out thoroughly and repeatedly, may not eradicate existing plants but can be useful in preventing seed set and minimizing their spread. Canada thistle can survive grubbing once or twice a month for many years before its root reserves are depleted (Sheley et al. 1995). Pulling Dalmatian toadflax annually for five or six years may be effective in containing its spread, especially if plants are pulled before seeds develop (Lajeunesse 1999). Mowing three or four times a year for three years reduced densities of Canada thistle in one study (Welton et al. 1929). Mowing two or three times before herbicide application can also enhance control of Canada thistle (Beck and Sebastian 1993). Sheep grazing has been used in Montana to suppress stands of Dalmatian toadflax; scorching floral stalks with propane burners can help prevent seed production in this species (Lajeunesse 1999). By reducing litter cover, prescribed fire enhances establishment of the flea beetle *Apthona nigriscutis*; which attacks leafy spurge (Fellows and Newton 1999). Angora goats can suppress leafy spurge but may also consume shrubs and trees (Kirby et al. 1997). Regardless of the technique used, treating plants before they flower will prevent, or reduce, the production of viable seeds.

A number of herbicides are approved for the treatment of noxious perennials. The choice depends on several criteria, including the target species, associated plant species, topographic setting, and infestation size. Herbicide effectiveness can be quite variable. For example, the high genetic variability of toadflax species, and the waxy leaf coating of Dalmatian toadflax, can hinder herbicide effectiveness (Lajeunesse 1999). Still, both picloram and dicamba have successfully decreased Dalmatian toadflax density, especially when infestations were treated for several years (Sebastian and Beck 1989; Sebastian et al. 1990). Picloram and dicamba are also used to treat Canada thistle; in addition, clopyralid, 2,4-D, and glyphosate have reduced thistle densities (Morishita 1999). Floyd et al. (2001) used a mixture of clopyralid plus 2,4-D to treat Canada thistle on burned areas in Mesa Verde National Park, and reported an average kill of 80 percent the following year on three-quarters of the plots treated. Regardless of herbicide choice, it is critical to follow recommended application rates and timing, and to re-treat in subsequent years. Maintaining a competitive plant community may also slow the spread of some perennial invaders (Lajeunesse 1999; Morishita 1999).

A number of potential biological control organisms are being studied for control of perennial noxious species, including insects, nematodes, and fungi for Canada thistle (Morishita 1999) and several species of insects for both Dalmatian toadflax (Lajeunesse 1999) and leafy spurge (Wilson and McCaffrey 1999). One nematode has been released for control of Russian knapweed, and insects and mites are being tested (Whitson 1999). Whether these natural enemies can control target species remains to be seen, but they may prove useful in an integrated approach for large infestations.

Application to Restoration Treatments

One of the most important steps in planning a restoration project is an inventory, in and adjacent to the project area, of noxious plants and other invasive species. This inventory should be used to prioritize treatment areas. Restoration efforts in areas with high levels of noxious species should be postponed until control efforts have been in place long enough to significantly reduce their populations and seed banks. Pretreatment inventories can also help identify noxious plant concentrations so that they can be avoided when planning placement of roads and landings. It must be emphasized, though, that pretreatment absence of exotic invasive plants does not guarantee that these species will not appear after treatment—from the seed bank, nearby infestations, or contaminated equipment and materials. Areas adjacent to roads with a past history of timber harvesting and livestock grazing are especially likely to have a rich assortment of invasive species hidden in seed banks (Korb 2001b).

A second important principle is to reduce severe soil disturbance as much as possible or, before severe disturbances occur, remove topsoil and litter so that it can be returned following disturbance (Korb 2001b). During logging operations, options include using skyline or helicopter logging instead of traditional skidding, running heavy equipment when soils are frozen, and limiting the amount of time that the timber operation is active. Broadcast burning of logging slash, instead of piling and burning, will reduce fire residence time and the amount of exposed mineral soil (Korb 2001b). If piling is necessary, hand-piling is preferable to skidding; small, tall piles result in less area disturbed than large, spread-out piles. Managers should also consider burning slash on roads designated to be obliterated, and adding soil or duff amendments to reduce the expression of exotic plant species (Korb 2001b). Fire managers should take advantage of natural barriers and existing roads as much as possible in order to limit construction of new firelines. Areas free of noxious species should be used for staging logging and burning operations. Workers should clean equipment before entering the project area and avoid traveling through areas with known infestations.

When severe soil disturbances do occur, whether from road building, logging, pile burning, or wildfires, seeding and addition of soil amendments may be appropriate, depending on the intensity and size of the disturbance, proximity to infested areas, availability of local native seed sources, slope, and erosive potential. Seeding alone or in combination with soil amendments after wildfires (Floyd et al. 2001) or after slash pile burning (Korb 2001b) may reduce densities of exotic invaders. Areas seeded with a native grass mixture in Mesa Verde National Park following a large wildfire in 1996 had lower densities of musk and Canada thistle in subsequent years than unseeded areas (Floyd et al. 2001). Of greatest importance in seeding is selecting only native species and using seed and straw mulch that are certified free of noxious species. Use of native seeds collected from the project area is preferable to avoid diluting or swamping the locally adapted genetic strain of the same native species (Chapter 14; Fenster and Dudash 1994). The only potential exception to the "natives only" rule is the use of annual cereal grains such as barley (*Hordeum vulgare*), cereal rye (*Secale cereale*), common oat (*Avena sativa*), and winter wheat (*Triticum aestivum*) for establishing temporary cover on severely disturbed sites until native species can become established (Robichaud et al. 2000).

Another important principle to follow either after restoration treatments or wildfires is to monitor and immediately treat designated noxious species and other plants known to be invasive elsewhere (Randall 1997). Populations of many invasive species often remain small and localized for long periods before suddenly expanding explosively (Hobbs and Humphries 1994). Eradication of species just beginning to colonize an area may be possible; delaying treatment can greatly increase the cost of control efforts and the threat to native plant diversity (Higgins et al. 2000).

Finally, we echo Hobbs and Humphries' (1994) plea for the need to focus on the invaded ecosystem and its management, rather than just on individual invaders. Failure to recognize and treat the ultimate causes of exotic species invasion will allow the process of reinvasion to continue. The presence of highways, nearby developments, and numerous forest roads (Schmidt 1989), combined with heavy livestock and wildlife grazing (Safford and Harrison 2001) and large numbers of human visitors (Lonsdale 1999) provides ample avenues and pathways for the introduction and spread of introduced species. Failure to deal with these contributing causes will make control of these plants highly unlikely.

Conclusion

An amazing variety of nonnative plant species has been introduced to the Southwest. The presence of these newcomers makes it highly unlikely that the vegetative composition characteristic of presettlement forests can ever be restored. We cannot eradicate them all. The ecological impacts of many exotic species are seemingly inconsequential, but there are a number of exotics that we cannot afford to ignore—those with a high potential for expanding explosively and altering the long-term structure and function of forested ecosystems.

Troubling knowledge gaps remain regarding invasive plant colonization of treated sites, including assessments of what species, in addition to those listed as "noxious," have the capacity, through their tenacity and invasiveness, to greatly alter posttreatment plant communities (Hiebert 1997). For example, can we use factors such as a species' history of invasion or ability to reproduce vegetatively to predict probabilities of invasiveness (Kolar and Lodge 2001)? We lack basic ecological knowledge on the underlying demographic mechanisms of successful biological invasions (Lavorel et al. 1999; Grigulis et al. 2001) and how they are influenced by the timing and severity of disturbances such as fires in various soil types. For example, are there times of the year when prescribed burns might not spread invasive species? We need better documentation of the relative contributions of roads, recreation, and grazing to invasive species spread, and we need to develop new techniques to curtail their spread into forests from towns or roadsides. In particular, what native grass species can be used to seed road ditches and other recently disturbed lands to deter nonnative invaders? Finally, more information is needed on cost-effective, integrated options for treating established concentrations of noxious weeds and on the unintended consequences of management strategies. In our zeal to control unwanted noxious plant species, for example, how can we avoid hampering the expression of fire-adapted native species?

As the scale of restoration projects expands and focuses on urban–wildland interface areas, we will increasingly need to integrate invasive species data from urban areas with knowledge of infestations in forests. New, seamless mapping technologies will need to be developed to allow us to plan projects at bigger scales and in concert with other objectives of restoration efforts, such as maintaining or enhancing rare species. Models that operate at landscape levels and incorporate multiple datasets will be needed to accomplish these ambitious restoration objectives and to predict potential responses of invasive species. Such models require knowledge of the important ecological processes involved in biotic invasions, as well as their spatial scales (Higgins et al. 1996).

Restoration treatments can be modified to decrease the probability of exotic plant invasion by limiting the amount and intensity of soil disturbances and by using local soil and seed amendments. No standardized approach exists for containing invasive species once established; each treatment has its trade-offs. Control efforts will be most successful if infestations are aggressively treated when they are small and eradication is possible. Waiting until species are widespread decreases the odds of successfully containing them and greatly increases environmental and economic costs. Control efforts will be most successful when they use integrated approaches that focus on the ultimate sources of introductions and on restoring competitive vegetative cover.

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