



CHAPTER

# 7

## Invasive Plant Management on Anticipated Conservation Benefits: A Scientific Assessment

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## INTRODUCTION

Invasive plant species have many negative impacts on rangelands throughout the world. Invasive plants can displace desirable species, alter ecological processes, reduce wildlife habitat, degrade riparian systems, and decrease productivity (DiTomaso 2000; Masters and Sheley 2001). Invasive plants are estimated to infest about 100 million ha in the United States (National Invasive Species Council 2001). Experts recognize invasive species are the second most important threat to biodiversity after habitat destruction (Pimm and Gilpin 1989; Randall 1996; Wittenberg and Cock 2001). Furthermore, Wilcove et al. (1998) estimate invasive species have contributed to the placement of 35% to 46% of the plants and animals on the federal endangered species list. In 1994, the impacts of invasive plant species in United States were estimated to be \$13 billion per year (Westbrooks 1998). The amount of land infested by invasive plants is rapidly increasing (Westbrooks 1998) and subsequently the negative impacts of invasive plants are escalating. To address this issue, federal agencies and private land managers have developed and implemented integrated pest management (IPM) programs on rangeland.

### Description of IPM on Rangeland

IPM is a long-standing, science-based, decision-making process that identifies and reduces risks from pests and pest management-related strategies (USDA Regional IPM Centers 2004). It was initially used to describe agricultural systems, but has since expanded to include wildlands and rangelands. IPM processes involve the coordinated use of pest biology, environmental information, and management technologies to prevent significant pest damage through economical means, while

at the same time posing minimal risk to people, property, resources, and the environment.

In recent years, invasive plant management has evolved to more frequently incorporate an IPM philosophy, as opposed to focusing on a single control option with little consideration of the ecosystem or the side effects of particular control methods. Although IPM approaches are not currently used in many regions, research has shown that integrating various combinations of control options can provide more effective control compared to a single option.

IPM strategies have been used in rangelands for at least 20 yr, with interest in this approach greatly increasing over the last decade. Examples of more current IPM approaches include the combination of biological control agents (Lym and Nelson 2002; Nelson and Lym 2003; Wilson et al. 2004; DiTomaso 2008; Joshi 2008), prescribed burning (DiTomaso et al. 2006a), grazing (Sheley et al. 2004), mowing (Sheley et al. 2003; Renz and DiTomaso 2006), and revegetation (Enloe et al. 2005). Many of these integrated approaches combine nonchemical strategies with judicious use of herbicides. In most of these cases, the goal of the IPM approach was to establish a more desirable plant community that not only provides necessary ecosystem functions, but also provides some resistance to reinvasion, and thus, more effective long-term management of invasive plant species.

### Description of Assumed Conservation Benefits of IPM on Rangeland

Specific plant species have been perceived as weeds since agriculture began about 10 000 yr ago. Early agriculturalists used hoes and grubbing implements to control weeds for



Invasive species have the capacity to create complete monocultures like the field of medusahead near Boise, Idaho pictured above. (Photo: Alex Boehm)

the specific benefit of increased commodity production (Radosevich et al. 1997). During the 20th century, synthetic organic chemicals were extensively used to control invasive weeds in rangeland production systems, and natural enemies of invasive weeds have been used to reduce some undesirable plant populations below an economic threshold. The primary objective was to enhance grass production, while minimizing adverse ecological and human impacts of the control effort.

Currently, ecologists and land managers recognize substantial adverse ecological, environmental, and economic impacts associated with invasive plants (Pimentel et al. 1999; DiTomaso 2000; Levine et al. 2003). In response, they have designed more-comprehensive invasive plant management approaches in an attempt to achieve an array of benefits in addition to enhanced control and grass production. Invasive weed prevention strategies and programs are aimed at protecting noninfested rangeland. A major focus has been to manage invasive plants to establish and/or maintain a desired plant community, especially to promote restoration of natural plant communities. The assumed benefit of restoring desired vegetative cover is to create and maintain healthy functioning ecosystems that reduce reinvasion, protect soils, control erosion, reduce sediment, improve water quality and quantity, and enhance stream flow. Invasive plant management also aims to benefit biological diversity and wildlife through habitat improvement.

## OBJECTIVES AND APPROACH

The objective of this chapter is to provide a comprehensive assessment of the degree to which invasive plant management is achieving several commonly anticipated and desired benefits. We used a comprehensive review of peer-reviewed literature to assess the efficacy of various invasive plant management practices for each of nine conservation purposes developed for the Natural Resources Conservation Service (NRCS) conservation practice standard of herbaceous weed control. In contrast to the other conservation practice standards, this one was developed simultaneously with the Conservation Effects Assessment Program, so the stated purposes do not directly match those

in the new standard. This new conservation practice standard is defined as the removal or control of herbaceous weeds including invasive, noxious, and prohibited plants. The writing team developed the following conservation purposes at the request of the NRCS for this chapter: 1) protect noninfested rangeland; 2) enhance quantity and quality of commodities; improve forage accessibility, quality and quantity for livestock; 3) control undesirable vegetation; 4) create a desired plant community; 5) change underlying causes of weed invasion; 6) restore desired vegetative cover to protect soils, control erosion, reduce sediment, improve water quality and quantity, and enhance stream flow; 7) maintain or enhance wildlife habitat including that associated with threatened and endangered species; 8) protect life and property from wildfire hazards; and 9) minimize negative impacts of pest control on soil resources, water resources, air resources, plant resources, and animal resources. The chapter also contains a section detailing recommendation and knowledge gaps, and conclusions addressing this conservation practice.

## ASSESSMENT OF INTEGRATED PEST MANAGEMENT CONSERVATION PRACTICES

### Protecting Noninfested Rangeland

Invasive plant management has traditionally focused on controlling invasive plants on already-infested rangelands, with less emphasis placed on protecting noninfested rangeland by preventing invasions (Zalvaleta 2000; Peterson and Viegslis 2001; Simberloff 2003). A proactive approach focused on systematic prevention and early control provides solid economic returns where, on average, every dollar spent on early intervention prevented \$17 in later expenses (OTA 1993). The major components of invasive plant prevention programs include minimizing invasive plant introduction into noninfested areas (often through vector management), early detection and eradication of satellite patches, and increasing the resistance of desirable plant communities and soil systems to invasion (Davies and Sheley 2007).

*Minimizing Invasive Plant Introductions.* A substantial amount of literature documents the

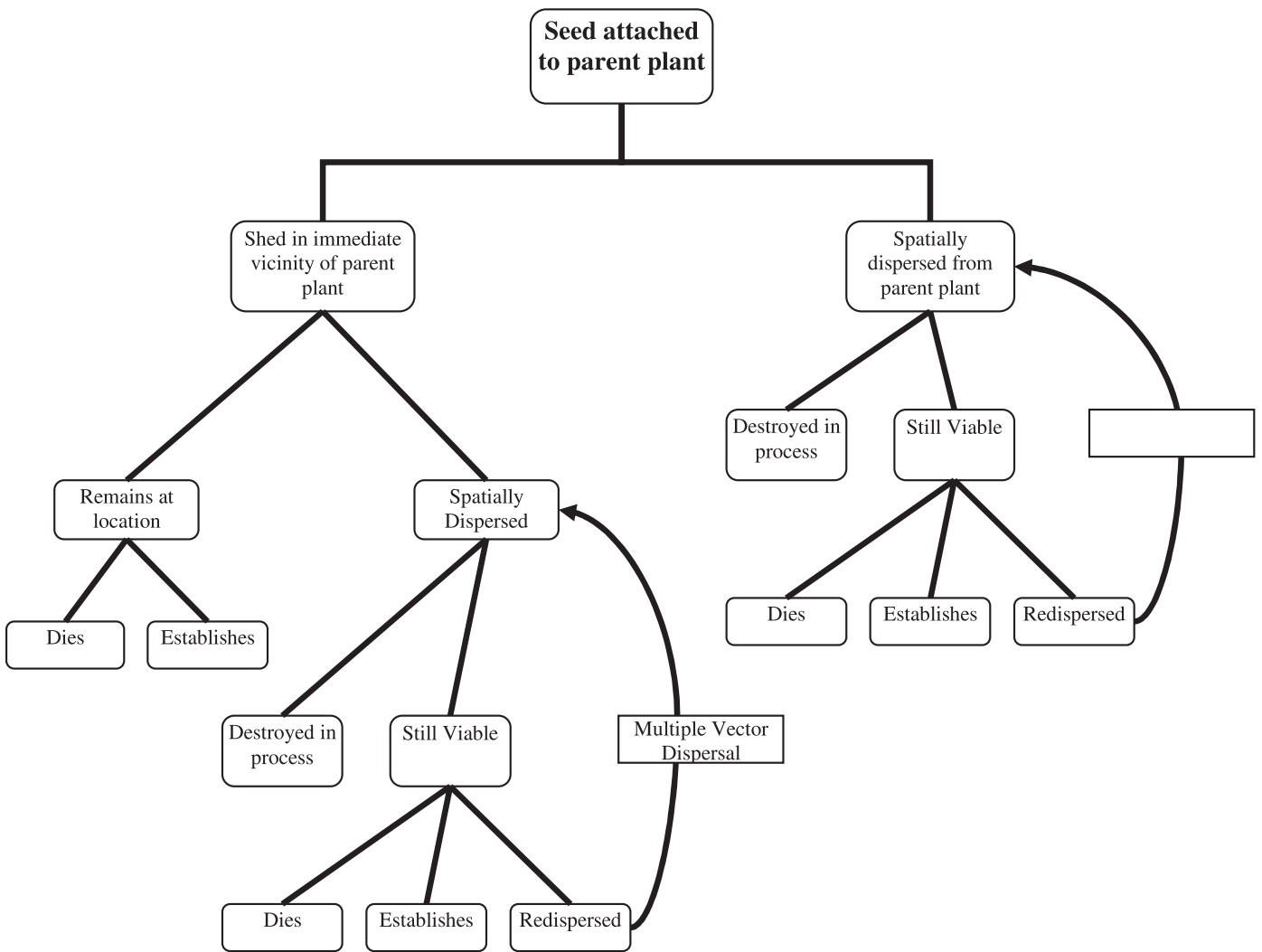


FIGURE 1. The potential fates and pathways of seed.

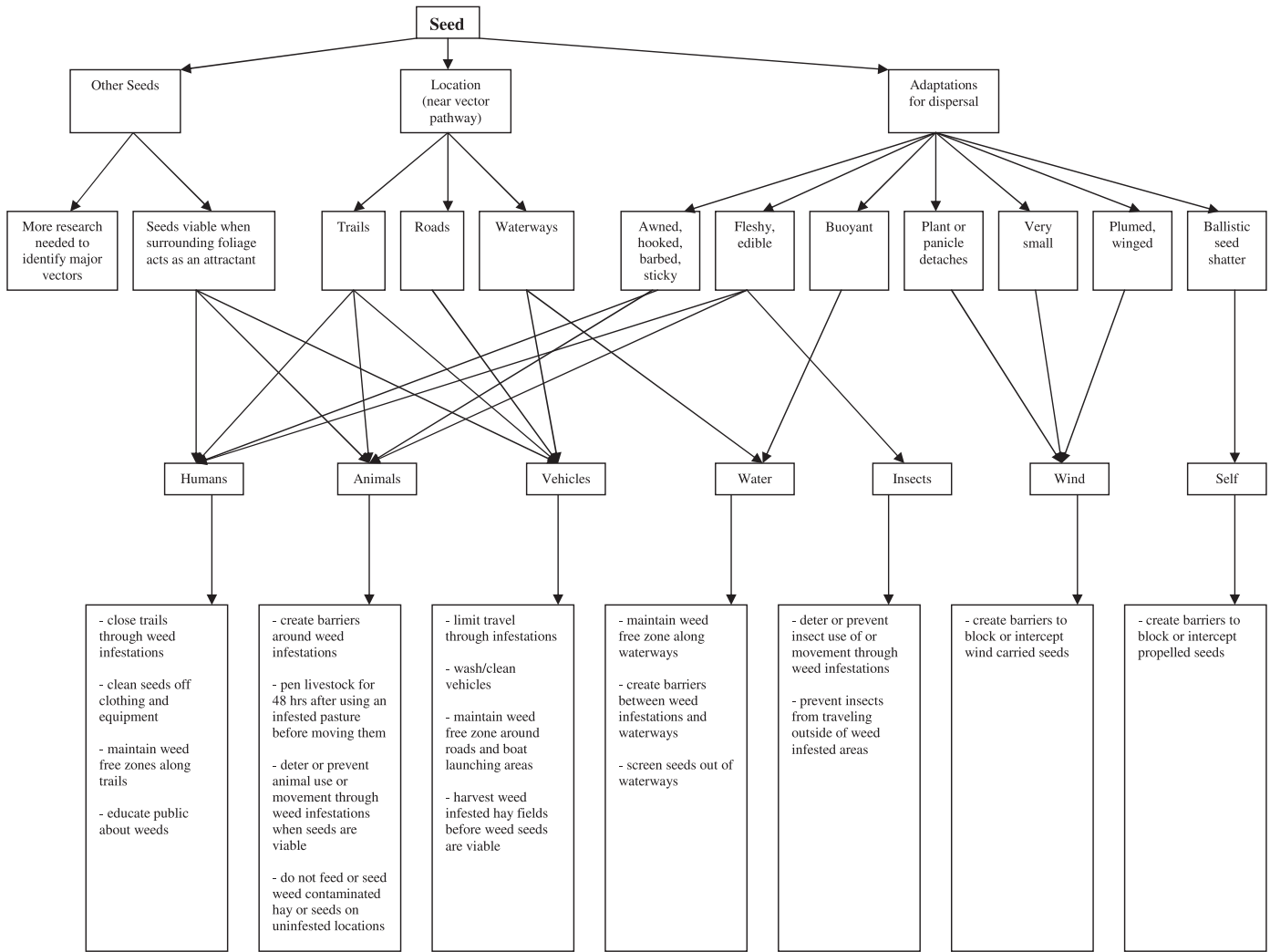
modes of plant dispersal throughout the world (Riley 1930; Janzen 1982). National, regional, and local introductions of invasive plants can occur many different ways (Plummer and Keever 1963). The most successful methods for reducing introductions of invasive plants are to create a break or diversion, especially in short- and long-distance dispersal (Fig. 1; Davies and Sheley 2007). Identifying vectors that are major dispersers of an invasive plant species provides vital information necessary for interrupting dispersal to new areas (Wittenberg and Cock 2001; Ruiz and Carlton 2003).

Dispersal vectors for some invasive species are known (Selleck et al. 1962; Brown and Archer 1987; Miller 1996; Kindschy 1998), but there is an obvious paucity of information about

dispersal vectors. Regardless, Davies and Sheley (2007) provide a conceptual framework for preventing spatial dispersal of invasive plants. The framework identifies major potential vectors by incorporating invasive plant seed adaptations for dispersal through space and infestation locations relative to vector pathways (Fig. 2). Land managers can use the framework to guide efforts to limit dispersal of invasive plant seeds where it is possible.

A major weakness in invasive plant management is our lack of knowledge about the efficacy of various prevention strategies. Tests of methods of preventing dispersal are extremely rare in the literature; however, most studies identifying dispersal vectors intuitively suggest a method to minimize these vectors.





**FIGURE 2.** Conceptual framework that identifies the major dispersal vectors of invasive plant species and the corresponding dispersal management strategies.

For example, avoiding human or livestock contact with invasive species possessing hooks, barbs, and awns during seed production will likely help minimize dispersal (Agnew and Flux 1970; Sorensen 1986). We could only find a single study that directly tested a potential prevention strategy. In that study, wind dispersal was limited by increasing neighboring vegetation height for species having large plumes (Davies and Sheley 2007).

**Early Detection and Rapid Control Response.** One key to preventing new infestations is early detection of small patches that have a high probability of expanding into large infestations (Moody and Mack 1988). Early detection occurs at multiple levels of organization. The United States implements a national pest survey and detection program through the US Department of Agriculture–

Animal and Plant Health Inspection Service. On the local level, early detection is difficult and requires educated and well-informed land managers, pest management specialists, and private land owners (Navaratnam and Catley 1986). Systematic weeds surveys (Johnson 1999), mapping based on sampling (Roberts et al. 2004), global positioning systems (Lass and Callihan 1993), and remote sensing (Steven 1993) have all been used to detect new infestations of invasive plants. In spite of the importance of detecting small infestations of invasive weeds, the cost, difficulty of implementation, and lack of reliable technology limit effective local early detection programs.

It is critical that small patches be effectively eradicated quickly after they have been located (Zamora et al. 1989; Simberloff 2003). Eradication involves the destruction of every

individual from an area (Newsome and Noble 1986). Most eradication strategies include aggressively repeated monitoring and control procedures (Weiss 1999a). There is a paucity of successful plant eradications found in the scientific literature and most descriptions of eradication programs are published in non-peer-reviewed formats (Simberloff 2003). Very few successful examples exist in the literature. The first international conference on eradication reported about 15 plant species that were eradicated from various areas around the world (Simberloff 2001). Pokorny and Krueger-Mangold (2007) provide evidence that small-scale eradication is achievable by documenting the successful removal of dyer's woad (*Isatis tinctoria* L.) from various counties in Montana. One biological tenet of successful eradication is that the infestation must be in the initial phases of invasion and only dominate a small area. Removing an invasive species is possible, especially for small infestations, but only under some circumstances and with potentially unpredictable results (Myers et al. 2000).

***Invasion-Resistant Plant Communities and Soil Systems.*** Promoting desired species is a critical component of invasive plant management, especially in an attempt to prevent invasions (Sheley et al. 1996). Researchers have shown that functionally diverse plant assemblages resist invasion better than less-diverse assemblages (Burke and Grime 1996; Levine and D'Antonio 1999; Kennedy et al. 2002; Pokorny et al. 2005). Invasion-resistant plant communities can be achieved by maximizing niche complementarity among desired species (Tilman et al. 1997; Brown 1998; Carpinelli 2001; Fargione and Tilman 2005; Funk et al. 2008). Furthermore, those plant communities that maximize biomass production also minimize invasion (Hooper and Vitousek 1997; Anderson and Inouye 2001).

Strategies aimed at maintaining desired plant communities help protect noninfested rangeland. For example, prescribed fall burning of late-seral big sagebrush–bunchgrass plant communities stimulated the herbaceous component and increased the resistance of the communities to cheatgrass invasion 4 yr postburn (Davies et al. 2008). In another example, clipping to simulate grazing greatly

reduced medusahead (*Taeniatherum caput-medusae* L.) by removing decadent material for desired plant species, which stimulated regrowth and enhanced competitive ability (Sheley et al. 2008).

It has been proposed that invasion-resistant soils can be created by lowering plant-available nitrogen (Vasquez et al. 2008). Managing soil-available nitrogen can be achieved by light to moderate levels of grazing. Grazing animals can remove nitrogen in plant material, making it unavailable to plants (Neff et al. 2005; Steffens et al. 2008). Mowing can remove nitrogen if the plant material is removed from the site (Oomes 1990; Moog et al. 2002).

### ***Conclusions and Management Implications.***

Protection of noninfested rangeland is central to the successful implementation of any integrated weed management program. Achieving the actual protection benefit is possible, but difficult, primarily because of a lack of effective techniques to interrupt dispersal vectors and our inability to detect new infestations before they become large infestations. Once found, small patches of invasive plants can be eradicated, but a comprehensive and intensive long-term eradication program must be employed. Many social, technological, and economic barriers exist that minimize the success of eradicating large infestations. The most scientifically developed strategy for protecting noninfested rangeland from invasion are those that convey some degree of invasion resistance to the plant community and possibly soils. Managing desired plant communities to enhance the success of late-seral species, enhance diversity, and maximize productivity should help to minimize invasion and protect noninfested rangeland.

### **Enhance Quantity and Quality of Commodities; Improve Forage Accessibility, Quality, and Quantity for Livestock**

The primary marketable commodities garnered from rangeland ecosystems are cattle and sheep, and, to a lesser extent, goats. Invasive plant management can influence quality and quantity of forage, as well as its accessibility. Consequently, the quantity and quality of livestock products can be impacted, but the





Annual grasses significantly decrease forage capacity of rangeland because cattle typically avoid them once they begin to develop seedheads, which is usually late spring to early summer for cheatgrass. (Photo: Ryan Steineckert)

direction and degree of impact varies for the three classes of livestock based on forage preferences and the plant functional groups being managed. In addition, the longevity of control impacts varies dramatically among invasive weed management strategies and their efficacy.

**Cattle.** Most invasive weeds decrease forage production for cattle (Olson 1999). A substantial amount of literature shows an increase in forage as a response to invasive plant management over an untreated control, but experimental evidence showing a reduction in forage production with weed invasion is limited (except see Maron and Marler 2008). In addition to loss of forage, cattle tend to avoid areas with heavy infestations of weeds (Lym and Kirby 1987; Hein and Miller 1992). For example, leafy spurge (*Euphorbia esula*

L.) reduces the carrying capacity of infested rangeland to near zero because cattle will not graze in areas with 10% to 20% cover of this weed. Few examples of overall economic costs of invasion have been published, but losses of forage for cattle on private land in California alone are estimated to be \$7.65 million per year because of yellow starthistle (*Centaurea solstitialis* L.; Alison et al. 2007).

Among 60 articles addressing invasive weed management and forage, 17 indicated an increase in forage quantity, quality, or accessibility. Increases occur where desired species are sufficiently abundant to respond to control procedures (Kedzie-Webb et al. 2002). Increases in perennial grass biomass ranged from 10% (Lym and Messersmith 1990) to 1 935% (Masters et al. 1996) in response to weed control. Most commonly, weed control



using herbicides increased forage for cattle about two- to threefold after 3 yr (Sheley et al. 2000). For example, picloram used to control spotted knapweed (*Centaurea stoebe* L.) increased grass yield by 1 513 kg · ha<sup>-1</sup> for 2 yr. Nearly all studies were 1 yr to 3 yr in duration, with the period of invasive plant control being about 2 yr or 3 yr. Very little is known about the long-term forage production after a single herbicide application or a sustained control program. However, Rinella et al. (2009) found that leafy spurge had increased and grasses decreased in comparison to nontreated areas 17 yr after picloram treatment.

Effective biological control only exists for a small portion of the total invasive weed species. However, biological controls have increased forage quality and quantity as well as accessibility for cattle where they do exist. Huffaker and Kennett (1959) reported large increases in availability of grasses and forbs as cattle forage 10 yr after the release of natural enemies of St. Johnswort (*Hypericum perforatum* L.). Increased grass production has been reported after release of biological control agents (Rees et al. 1996). *Longitarsus jacobaeae* (Coleoptera: Chrysomelidae) has reduced the density of tansy ragwort (*Senecio jacobaea* L.) and reduced cattle losses to pyrrolizidine poisoning to near zero (McEvoy et al. 1993; Coombs et al. 1996).

Sheep prefer grazing broadleaved plants and can be used to shift plant communities toward grasses that are preferred by cattle. Johnston and Peake (1960) used sheep to reduce leafy spurge basal area and increase the basal area of crested wheatgrass (*Agropyron desertorum* [Fischer ex Link] Shultes). Similarly, sheep grazing increased Idaho fescue (*Festuca idahoensis* Elmer) density and the frequency of Kentucky bluegrass (*Poa pratensis* L.), while reducing spotted knapweed (Olson et al. 1997). Livestock grazing has also been successfully used to reduce annual grasses growing among perennial grasses (Havstad 1994).

Science and technology have not advanced to the point that reseeding desired species is consistently successful, but seeding desirable plants into invasive plant-infested rangeland can increase the quantity and quality of forage for cattle (Enloe et al. 2005; Sheley et al. 2005).

In one successful case, applying clopyralid plus 2,4-D in combination with streambank wheatgrass (*Elymus lanceolatus* Scribn & Sm.) was used to reclaim a rangeland heavily infested by Russian knapweed (*Acroptilon repens* [L.] DC.) to a stand dominated by the sod-forming grass (Benz et al. 1999). More recently, methods for repairing damaged ecological processes have increased the success of revegetation across highly variable landscapes (Sheley et al. 2006, 2009). In these studies, specific processes in need of repair were identified and modified to foster vegetation dynamics toward favorable species. Integrated invasive plant management strategies also have the potential to improve the quantity and quality of rangeland for cattle. Among 100 randomly selected studies investigating integrated invasive weed management, 65 indicated short-term positive responses in the quality and quantity of forage for cattle. In a few cases, the response was synergistic in favoring desired vegetation composition (Sheley et al. 2004; Jacobs et al. 2006). Additive and single main treatment effects were the dominant response, and in most cases nonnative grasses increased over nonnative target invasive weeds (Lym 1998; Endress et al. 2008).

**Sheep and Goats.** Because sheep and goats consume comparatively more forbs than grasses in their diets, invasive plant management does not benefit these small ruminants (Lym and Kirby 1987; Kronberg and Walker 1993). Most broadleaved weeds contribute to the forage quantity and quality of sheep and goats (Olson and Lacey 1994). Although the nutrient content of broadleaved invasive weeds varies with phenology, most are highly nutritious (Bosworth et al. 1980, 1985).

### Conclusions and Management Implications.

In general, the quantity and quality of cattle forage, and thus, cattle, are favored by weed management for a short period. Because many invasive weed management procedures increase forage yield for 2–4 yr, the benefits decrease with time following treatment. Sheep and goats prefer forbs as a major dietary component, and consume many weeds as quality forage. Broadleaved weed management has few positive benefits for sheep and goats. Invasive plant managers may maximize commodity production using multispecies grazing.

## Control Undesirable Vegetation

Controlling undesirable vegetation on rangelands is difficult and rarely cost-effective. Compared to other land types, rangelands generate relatively low revenues per unit area. Typically, rangeland managers face expansive invasive plant infestations with few dollars for management. Additionally, invasive weeds tend to have high intrinsic growth rates and abundant seed production (Hobbs 1991; Rejmanek and Richardson 1996), which allow for rapid reinvasion of sites following use of herbicides, prescribed fire, and other invasive plant control practices (Lym and Messersmith 1985b; DiTomaso et al. 2006b). Therefore, when invasive plants are successfully controlled, they often reoccupy the area very quickly. The keys to sustainably controlling large rangeland weed infestations are frequent use of strategies that provide inexpensive, short-term control, such as prescribed grazing or infrequent use of more expensive strategies that provide longer-term control, such as restoration. In this section, we review widely used weed control strategies with an emphasis on their short- and long-term effectiveness, as well as their costs.

Prescribed fire in areas of invasion has varying effects on different species and can often lead to an increase in production of invasive species. (Photo: Clare Poulsen)

**Prescribed Fire.** Fire consumes weed standing crop, and in this sense fire consistently reduces undesirable vegetation. However, to have any lasting effect, prescribed fire must reduce the

production of biomass in subsequent growing seasons, reduce the existing year's standing crop, and have a neutral or positive effect on desirable species. Some studies report increases in invasive weed biomass production due to fire (Young et al. 1972; Jacobs and Sheley 2003; Travnicek et al. 2005; Thacker et al. 2008), whereas others indicate decreases in biomass (Whisenant et al. 1984; DiTomaso et al. 1999). Whisenant et al. (1984) reported an extreme reduction of Japanese brome (*Bromus arvensis* L.) due to fire, which temporarily reduced this invasive annual grass by 85%. Conversely, Jacobs and Sheley (2003) found that fire more than doubled production of Dalmatian toadflax (*Linaria dalmatica* [L.] Mill. subsp. *dalmatica*). DiTomaso et al. (2006) concluded that the effects of fire depend, in part, on the weed's life history strategy (i.e., annual, biennial, perennial) and characteristics of the fire regime. Fire-based invasive plant management is complex and often not predictable, and detailed studies are needed to identify effective fire regimes for particular species or similar groups of species.

Applying prescribed fire is costly, so it is important for managers to carefully consider the longevity of control. Unfortunately, effects of fire on invasive plants are usually measured for only a year or two postburn. Cheatgrass (*Bromus tectorum* L.) has been measured for longer periods, but this weed does not appear amenable to fire-based control. On Western rangeland, cheatgrass tends to increase with fire because fire suppresses the less fire-tolerant competitors and available resources are rapidly acquired by cheatgrass (Young and Evans 1978; Vasquez et al. 2008). Cheatgrass invasion often increases fire frequency by increasing fine fuel loads, so burning cheatgrass can trigger a frequently repeated cycle whereby cheatgrass increases fire and fire increases cheatgrass (Knapp 1996). Conversely, prescribed fire that reduces invasive plants mainly destroys propagules, rather than by altering the environment in a manner that disfavors invasive weeds (DiTomaso et al. 2006a). When this is the case, weeds have only to replenish their propagule supplies to regain preburn abundances. Invasive plants tend to have high intrinsic growth rates which allow them to regain lost propagules quickly (Rejmanek and Richardson 1996; Grotkopp et al. 2002; Pyšek



and Richardson 2007). Therefore, prescribed fire will tend to provide only short-term control, and managers would need to burn regularly enough to maintain control for the long term. Finally, as with cheatgrass, there are many herbaceous perennials that cannot be controlled by fire alone, but fire can kill large quantities of surface-deposited seeds (Vermeire and Rinella 2009), so integrating fire with strategies that kill established plants may enhance control of some perennial invasive plants.

**Herbicides.** Herbicides are very useful for preventing small invasive plant infestations from producing seeds and spreading. They are also effective for controlling weeds during restoration projects so that seeded species have a better chance of establishing (Cione et al. 2002; Huddleston and Young 2005). These uses aside, it is generally not cost-effective to control large invasive plant infestations with herbicides alone because the repeated applications required to maintain control (every 1–3 yr) are too expensive (Lym and Messersmith 1985b; Sheley et al. 1998; Young et al. 1998). Controlling rangeland invasive plants rarely increases forage production enough to offset the herbicide costs (Griffith and Lacey 1991; Bangsund et al. 1996). Furthermore, invasive annual grasses often proliferate after herbicides kill associated invasive forbs, so controlling invasive broadleaved forbs often just replaces undesirable forbs with undesirable annual grasses (Shinn and Thill 2003).

Another problem with large-scale herbicide treatments is that they often kill associated native forbs and shrubs (Erickson et al. 2006; Sheley and Denny 2006; but see, Rice et al. 1997a). Whereas invasive weeds usually recover from herbicide quickly, a recent study of leafy spurge shows desired native plants can fail to recover from herbicides regardless of the length of the recovery period (Rinella et al. 2009). In that study, leafy spurge filled niches left vacant after herbicides removed native plants. Paradoxically, when herbicides damage native plants, invasive plants may ultimately become more abundant in response. Although herbicides play a critical role in weed prevention and restoration, the scientific literature causes us to question their use as

stand-alone tools for controlling expansive invasive plant infestations.

**Prescribed Grazing.** Prescribed grazing encourages the targeted use of invasive plants by manipulating timing, intensity, and frequency of herbivory and selecting animal classes based on their dietary preferences. For example, goats prefer trees and shrubs and forbs compared to grasses, so they are sometimes stocked on grasslands invaded by pines and junipers or invasive forbs, such as knapweed (Campbell et al. 2007).

The key difference between prescribed grazing and other invasive plant management strategies is that it can be affordably used on an annual basis to reduce invasive plant standing crop and biomass production in many situations. Sheep and goats can be economically profitable in well-managed operations or they can serve as additional revenue sources in cattle operations (Williams et al. 1996; Bangsund et al. 2001). Additionally, using sheep or goats to graze and reduce exotic forb standing crop can increase the amount of forage accessible to cattle and increase overall forage utilization (Lym and Kirby 1987).

Desirable rangeland species generally increase after land management practices reduce invasive plant biomass (Lym and Messersmith 1985a; Belcher and Wilson 1989; Sheley et al. 2000). As a consequence, prescribed grazing is more worthwhile when it reduces subsequent invasive plant biomass in addition to reducing standing vegetation. Invasive plant biomass responses to grazing depend on the timing, intensity, and frequency of grazing, as well as the class of livestock. For example, three studies reported no effect of sheep grazing on leafy spurge production because most grazing occurred during and after leafy spurge seed production, and at only a single time during the season (Lacey and Sheley 1996; Olson and Wallander 1998; Seefeldt et al. 2007). Alternatively, four other studies reported fairly consistent declines in leafy spurge over time, and in these studies, grazing occurred multiple times prior to seed production (Johnston and Peake 1960; Lym et al. 1997; Jacobs et al. 2006; Rinella and Hileman 2009). Controlling invasive plants using livestock requires the development of relatively complicated,



Although herbicides play a critical role in weed prevention and restoration, the scientific literature causes us to question their use as stand-alone tools for controlling expansive invasive plant infestations.”





Prescribed grazing and biological control can be effective treatments when implemented into an integrated management program. (Photo: Sharon Bingham)

strategically designed strategies for each species based on their tolerance and/or resistance to herbivory. Detailed investigations are needed to identify prescribed grazing strategies that are effective for specific invasive species in particular environments.

**Biological Control.** Biological control agents can clearly damage individual plants (Pecinar et al. 2007; Thomas and Reid 2007; Zalucki et al. 2007), but unfortunately, these effects often fail to cause appreciable reductions in undesirable vegetation (DeLoach 1991). Many agents are released in hopes that one or a combination of them will prove effective (McEvoy and Coombs 1999). Unfortunately, this lottery approach is rarely effective in controlling large populations. Many invasive plants remain highly problematic despite being the target of many releases of

biological control agents for decades (Zalucki et al. 2007; Story et al. 2008). Furthermore, the risks of deleterious off-target effects increase with the number of releases (Louda et al. 2005; Pearson and Callaway 2005).

Risks and failures aside, biological control is occasionally extraordinarily successful against invasive weeds. The most-cited examples include two introduced beetles that reduced St. Johnswort density by greater than 99% in much of its introduced range (Harris and Maw 1984), and three insects that substantially reduced ragwort in western Oregon (McEvoy et al. 1991; Denslow and D'Antonio 2005). Although these are the best-studied examples, other rangeland weed species have been targeted, with quite varied results, including recent introductions that show promise for

controlling salt cedar (*Tamarix ramosissima* Ledeb.; Hudgeons et al. 2007). In those few special cases, biological control has the unique benefit of providing relatively inexpensive partial weed control over expansive areas for indefinite periods of time.

### *Mechanical Control and Seeding.*

Mechanical methods of herbaceous weed control include tillage and mowing. Mechanical control treatments of tillage and mowing can cause substantial reductions in invasive plant standing vegetation, but they are only truly effective when future biomass production is reduced. Invasive plant responses to mowing have been mixed, with some studies reporting appreciable decreases in weed biomass production (Benefield et al. 1999; Rinella et al. 2001) and other studies reporting no detectable change (Benz et al. 1999; Renz and DiTomaso 1999). Collectively, studies indicate the responses of undesirable vegetation to mowing depend on species, timing of mowing, and other factors. Finally, because invasive weeds quickly recover after mowing is discontinued, mowing must occur frequently to provide continuous control.

Tillage alone can be used to control invasive plants on rangeland under some circumstances. In one study, tillage alone provided no sustained control of perennial pepperweed (*Lepidium latifolium* L.; Young et al. 1998), and in another study, repeated tillage prior to a serious frost reduced leafy spurge well (Lym and Messersmith 1993). Invasive weeds tend to recover quickly when tillage is discontinued.

In addition to controlling invasive plants, tillage provides safe sites for seeded species, and some studies report that competition from seeded species has partially controlled undesirable vegetation (Lym and Tober 1997; Bottoms and Whitson 1998; Sheley et al. 2001; Thompson et al. 2006). However, other studies have reported that seeded species provided no weed control (Sheley et al. 1999; Mangold et al. 2007). In the latter case, it is possible data were collected before the seeded species grew large enough to compete with the reemerging invasive plants. Theoretically, when seeded species develop self-sustaining populations, these populations should suppress undesirable vegetation indefinitely through

resource competition. Therefore, despite the lack of evidence, there are likely to be distinct advantages to integrating seeding with other practices that provide only short-term control, such as herbicides and tillage.

### *Conclusions and Management Implications.*

Many questions remain regarding control of invasive plants, and many of these questions pertain to inconsistencies in the responses of undesirable vegetation to various controls. For example, there are cases in which individual treatments, such as herbicides, grazing, or fire have reduced invasive weeds. There are also cases in which treatments have failed to alter the abundance of invasive plants, and there is even some evidence that invasive weed control occasionally increases weed species. Furthermore, except for a few situations using biological control or annual repeated grazing, it is often questionable whether or not individual invasive weed control strategies are worthwhile because the control they provide is so ephemeral and expensive.

Presumably, integrating multiple control strategies should lead to more consistent, longer-lasting suppression of invasive plants. But integrated strategies are more costly, and there is still no guarantee that the level and longevity of invasive plant control will be satisfactory (Sheley et al. 2001; Lym 2005). Much research is needed to identify affordable, consistently effective strategies for controlling undesirable vegetation.

### **Create a Desired Plant Community**

On most sites, the species that invasive plants suppress or displace comprise both nonweedy exotic species and natives (Enloe et al. 2007). Some of the suppressed natives and nonnatives are often valuable forage plants, and increased forage production often provides the impetus for controlling invaders (Lym and Messersmith 1985a). In many cases, controlling undesired species does not lead to a desired plant community. A variety of other objectives may also be met by creating desired plant communities including increasing native species diversity, increasing habitat for wildlife, improving soil and water quality, and reducing reinvasion. In this section, we investigate how desired, especially native, species respond to invasive weed control and examine efforts

A desired plant community should include a variety of species that fill multiple niches which reduces the likelihood of reinvasion. (Photo: Brett Bingham)







These studies suggest that prescribed grazing may have potential for restoring desired species, but invaded communities are likely to quickly regress to their pregrazing weedy state when prescribed grazing is discontinued.”

to reestablish desired species from seeds. The choice of desired species depends upon management goals. In this document, the goal is considered to be to establish and maintain a healthy, functioning plant community that is resistant to invasion and meets other land use objectives (Sheley et al. 1996).

**Prescribed Grazing.** To our knowledge, only two studies have provided detailed assessments of plant community responses to prescribed grazing of invasive plant–infested rangeland. Olson and Wallander (1998) studied responses of a leafy spurge–infested plant community to prescribed sheep grazing. The authors concluded that grazing reduced leafy spurge stem height without affecting stem density, so grazing presumably lowered leafy spurge biomass production. Grazing increased the density and frequency of several native (Idaho fescue, western wheatgrass [*Pascopyrum smithii* {Rydb.} A. Löve], Sandberg bluegrass [*Poa secunda* J. Presl]) and nonnative (Kentucky bluegrass, annual bromes) grasses, and decreased density of a nonnative dandelion (*Taraxacum officinale* Weber). In a similar study, on spotted knapweed-infested rangeland, Olson and Wallander (1997) found that sheep grazing reduced spotted knapweed rosette and adult plant density. As in the leafy spurge study, grazing increased density and frequency of native Idaho fescue and nonnative Kentucky bluegrass. The native forb arrowleaf balsamroot (*Balsamorhiza sagittata* [Pursh] Nutt.) was not influenced by grazing. These studies suggest that prescribed grazing may have potential for restoring desired species, but invaded communities are likely to quickly regress to their pregrazing weedy state when prescribed grazing is discontinued. To be successful, prescribed grazing will likely need to be carried out indefinitely. It is unfortunate that so few studies have evaluated native species responses to prescribed grazing of weed-infested rangeland.

**Biological Control.** Denslow and D’Antonio’s (2005) review of the literature clearly demonstrates that successful biological control of rangeland invaders can, but does not always, have positive effects on suppressed desired species. A classic successful example is the control of St. Johnswort by the leaf-beetles *Chrysolina quadrigemina* Suffrian

and *Chrysolina hyperici* Forster in California rangeland. Within 5–10 yr of leaf-beetle introduction, St. Johnswort had virtually disappeared from sites in several California counties (Huffaker and Kennett 1959). It was reduced to less than 1% of its prerelease cover, and replaced by a combination of native and exotic grasses, which greatly increased available forage. Similarly, a combination of several insects reduced ragwort to a fraction of its former abundance in several California sites (Pemberton and Turner 1990), and throughout western Oregon (McEvoy et al. 1991). In both situations, control agents persisted, ragwort remained under control for more than a decade, and desired species responded favorably. More recently, a suite of biological control agents led to successful control of diffuse knapweed (*Centaurea diffusa* Lam.) at sites in Colorado, Montana, Oregon, Washington, and British Columbia (Myers 2004; Smith 2004; Seastedt et al. 2007), and successful or partial control of spotted knapweed in sites in Colorado and Montana with some responses by desired species (Story et al. 2006; Seastedt et al. 2007). Similarly, several flea beetles (*Apthona* spp.) have displayed variable, but sometimes quite successful control of leafy spurge and subsequent release of desired species in South Dakota, North Dakota, and Montana (Larson and Grace 2004; Butler et al. 2006; Cornett et al. 2006). Additionally, Lesica and Hanna (2004) provide an example of positive native plant community responses to biological control.

**Herbicides.** Rangeland herbicides tend to selectively kill either grasses or forbs. Therefore, native grasses are typically not damaged by the herbicide used to control invasive forbs. In fact, many native grasses increase following herbicide control of invasive forbs (Sheley et al. 2000; Laufenberg et al. 2005; Sheley and Denny 2006). Similarly, native forbs often increase after herbicides kill invasive grasses (Cione et al. 2002; Wilcox et al. 2007).

Herbicides are sometimes used to control invasive grasses even though herbicide-sensitive native grasses are present (Kyser et al. 2007). Likewise, herbicides are used to control invasive forbs growing with native forbs and shrubs (Fuhlendorf et al. 2002). Sometimes native species escape extensive damage by herbicides



or are able to quickly recover from damage. For example, when the broadleaf herbicide picloram was applied to spotted knapweed–infested rangeland during the summer-dormant period of most native forbs, Rice et al. (1997b) found the herbicide had only mild transient effects on native forbs. Similarly, Erickson et al. (2006) found that herbicide control of leafy spurge with quinclorac did not damage the threatened prairie fringed orchid, although imazapic damaged the orchid. Also, Simmons et al. (2007) found the “nonselective” herbicide glyphosate provided substantial short-term control of an invasive grass while damaging native grasses to a lesser extent or not at all. Finally, Barnes (2007) found that grass-specific herbicides promoted native warm-season grasses by reducing abundance of the exotic grass, tall fescue.

In contrast to these examples, there are other cases demonstrating extensive herbicide damage to natives. For example, Sheley and Denny (2006) concluded that any of three herbicides used to control sulfur cinquefoil (*Potentilla recta* L.) continued to suppress native forbs 2 yr after application. Similarly, Shinn and Thill (2004) found that imazapic, a herbicide that is active against invasive annual grasses, substantially injured native perennial grasses as well. Herbicide damage to native species would not be a problem if the natives consistently recovered, but a recent study showed that herbicide control of rangeland weeds can pose very serious long-term threats to native forb populations (Rinella et al. 2009). In addition to sometimes extensively damaging native species, herbicides are expensive and they generally provide only short-term weed control. Therefore, herbicides alone are unlikely to create desired native plant communities. However, herbicides are critical in preventing spread of small weed infestations and in integrated weed management.

**Prescribed Fire.** Prescribed fire can boost native species and reduce populations of annual invaders by consuming their seeds (DiTomaso et al. 2006b). For example, Harmony (2007) found that fire greatly reduced Japanese brome and increased two native grasses above an unburned control. Also, fire reduced invasive yellow starthistle and three invasive annual grasses while greatly increasing diversity and

species richness of native forbs (Hastings and DiTomaso 1996; DiTomaso et al. 1999). However, depending on fire timing, species identity, and other factors, fire can also boost invaders (Young et al. 1972; Jacobs and Sheley 2003; Travnicek et al. 2005; Thacker et al. 2008) and cause severe damage to native populations, such as big sagebrush (*Artemisia tridentata* Nutt.) growing with cheatgrass (Young and Evans 1978; Knapp 1996).

To our knowledge, no studies have measured long-term invasive weed and native species responses to prescribed burning. Unfortunately, favorable responses likely will be short-lived. Weeds generally invade very quickly following fire (Young and Evans 1978), and native plants are likely to revert to their suppressed preburn state following reinvasion. It is unlikely that managers can burn vegetation regularly enough to maintain native populations, but is possible to integrate fire with other strategies in hopes of providing longer-term restoration of native species (MacDonald et al. 2007).

**Mowing.** A small number of studies have evaluated desired plant responses to mowing of invasive weed–infested rangeland. Wilson and Clark (2001) found mowing invasive grass-infested rangeland for 4 yr greatly restored native prairie grasses. Similarly, MacDougall and Turkington (2007) found that 5 yr of mowing at the time of invasive grass flowering shifted the plant community toward desired forbs and grasses. In contrast to these successes, Simmons et al. (2007) found mowing had little or no detectible effect on an invasive perennial grass and native species, and Brandon et al. (2004) reported mowing increased invasive forb abundances. Collectively, these studies indicate native species responses to mowing depend on the relative susceptibility of desired species and invaders to different timings, heights, and frequencies of defoliation as well as other factors. Finally, when beneficial mowing regimes are identified, they will likely have to be carried out indefinitely or combined in an integrated management strategy to maintain native species because invasive weeds tend to recover quickly when mowing is discontinued.

**Seeding.** In revegetation projects, the desired plant community is in large part dictated by the species in the seed mix. A few



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It has become clear that nonnative grasses have tended to outperform native grasses in revegetation studies.”

revegetation efforts have resulted in fairly diverse assemblages with impressive stability (Blumenthal et al. 2003; Martin et al. 2005; Martin and Wilsey 2006; Foster et al. 2007). However, these successes were obtained from areas with relatively innocuous weed species, not highly tenacious rangeland invaders such as leafy spurge and spotted knapweed.

Where invaders dominate rangelands, managers often strive for the humble goal of introducing one or a few desired or native species in hopes they will provide forage, wildlife habitat, and other values. Seeding is very expensive, so successful seedling establishment is critical. Seedling establishment is most effective when seeding has been integrated with herbicides and/or tillage (Lym and Tober 1997; Ferrell et al. 1998; Whitson and Koch 1998; Masters et al. 2001; Huddleston and Young 2005; Simmons 2005), but establishment failures are common (Lym and Tober 1997; Masters et al. 2001; Sheley et al. 2001; Wilson et al. 2004). Seedling establishment depends on a myriad of factors, not the least important of which is interannual climatic variation (MacDougall et al. 2008), which cannot be controlled or effectively predicted. Beyond establishment, seeded species must maintain viable populations over the long term for seeding to be worthwhile. Unfortunately, desired native species are rarely measured for more than a few years after sowing into invaded rangelands. Among the longer-term investigations, Bottoms and Whitson (1998) found that seeded thickspike wheatgrass (*Elymus lanceolatus* [Scribn. & J.G. Sm.] Gould) and western wheatgrass maintained healthy stands that suppressed Russian knapweed 5 yr after seeding. Similarly, Ferrell et al. (1998) found that seeded wheatgrasses produced large quantities of biomass and continued suppressing leafy spurge 6 yr after seeding. Also, a mixture of four native warm-season grasses contributed substantially to biomass production 6 yr after being sown into spotted knapweed-infested plots (MacDonald et al. 2007). To our knowledge, Ferrell et al. (1998) document the longest-term measurements of seeded species in invasive plant-infested rangeland. The authors found that two grass species remained fairly abundant and partially suppressed leafy spurge 10 yr after seeding. It has become clear that nonnative grasses

have tended to outperform native grasses in revegetation studies (Ferrell et al. 1998; Asay et al. 2001; Sheley et al. 2001).

Given the expense of seeding rangelands, the lack of long-term measurements is troubling. Future restoration research should focus on determining whether or not native species can persist with invasive species beyond a few years after seeding. To some extent, restoration efforts are predicated on the assumption that improper land management causes weed invasions. If proper management does not prevent invaders from dominating the original community, then we should not expect proper management to prevent invaders from dominating the restored community. It is sobering to consider that rangeland restoration may be doomed to fail over the long term wherever invaders have displaced natives despite good range management.

#### *Conclusions and Management Implications.*

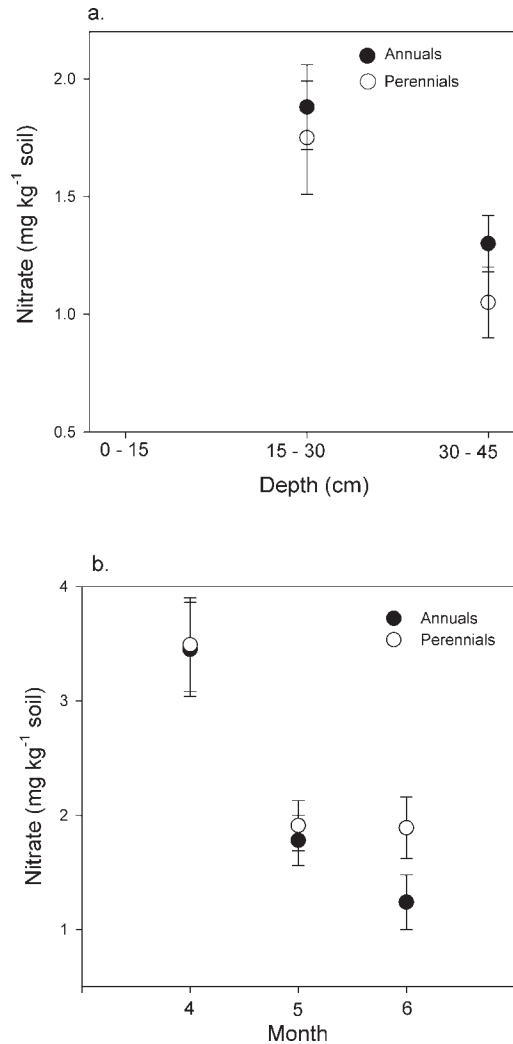
Many studies provide no information on desired or native species responses to weed management, and a few others provide only cursory information. Biological control is relatively inexpensive to implement once developed, and a few biological control programs have restored natives to an impressive extent. However, biological control sometimes fails completely, and it has many risks. More research is needed to elucidate the risks and benefits of biocontrol. Desired and native species responses to prescribed grazing have been limited, but two studies indicate that annually applied prescribed grazing can shift plant communities toward a desired state. Herbicides often control target invaders while allowing associated species to increase in abundance. However, herbicides are quite expensive and herbicides alone provide only short-term weed control. Furthermore, herbicides pose considerable risks to some desired species. Herbicides are useful for preventing spread of weed infestation, and thus helpful in maintaining a desired plant community that has not been invaded. Prescribed fire can be beneficial to desired species, but it can also harm natives and increase invasion. Effective prescribed fire regimes will likely need to be repeated regularly, and can be used occasionally to restore desired plant communities prior to invasion to help

keep them resistant. Effective mowing will probably need to be carried out often to be successful, and in some cases, the plant material may need to be removed. A few studies have reported that seeded native species remained abundant and suppressed tenacious rangeland invaders 5 yr or 6 yr after seeding. This is promising, but longer-term measurements are desperately needed to determine if the benefits of seeding warrant its high costs.

### Change Underlying Causes of Weed Invasion

The benefits of invasive plant control depend in large part on the longevity of control and the resulting desired plant community. In turn, this depends on the ecological causes underlying the original invasion and our ability to alter those causes in favor of desired species. Because invasive species are rarely eradicated, if the original causes of invasion are not repaired, reinvasion is likely (Sheley and Krueger-Mangold 2003). Furthermore, given the large areas and low economic returns per unit area typical of rangeland, temporary invasive weed control is rarely economically sustainable. In this section, we consider the durability of invasive species control and desired community restoration.

*Durability Depends on Original Cause of Invasion.* Invasion is often the result of changes to an ecosystem that inhibit native species, and thereby reduce the competition faced by invasive species (Facon et al. 2006). Perhaps the most important barrier to invasion is the presence of desired species. Desired species garner much of the water, nitrogen, light, and other resources that would otherwise be available to invaders. This resistance to invasion is often described as “biotic resistance” (Maron and Vila 2001; D’Antonio and Thomsen 2004; Levine et al. 2004), and depends a great deal on environmental conditions (Shea and Chesson 2002). Because native species are, by definition, adapted to historical environmental conditions (Landres et al. 1999), changes to these conditions are likely to make them less well-adapted, and less able to resist invasion. Consequently, changes in environmental conditions appear to be a common cause of invasion (Daehler 2003; Facon et al. 2006). These changes can be dramatic, such as soil tillage or improper grazing, or subtle, such



**FIGURE 3. a,** Exotic annual species (●) are less effective than native perennial grasses (○) at extracting soil nitrate at all depths in May. **b,** Monthly sampling was conducted at the 0- to 15-cm depth. Error bars = 1 SEM;  $n = 32$ .

as nitrogen deposition or loss (Vasquez et al. 2008). Where such changes underlie invasion, the key question facing managers is whether the change can be reversed. Among the many underlying causes of invasion, managers have had the most success reversing the following three: past disturbances, reversed via successful restoration; returning grazing to systems, and enemy release, through biological control.

*Restoration as a Long-Term Solution to Previous Novel Disturbances.* A key question for determining the likelihood of long-term invasive plant control is the degree to which a particular invasion is caused by a novel disturbance—a disturbance to which native species are not well adapted. Where such disturbances underlie the invasion, and can be prevented in the future, it is much more likely that a stable, desired plant community





...reestablishing proper grazing regimes that maintain vigorous plants and healthy plant communities can limit invasion.”

can be restored. A combination of invasive weed control (typically with herbicides) and restoration can potentially reestablish native species. Long-lived perennial species often garner much of the water, nitrogen, and other resources available in grassland ecosystems (Wedin and Tilman 1990; Tilman and Wedin 1991; Baer et al. 2002; Seabloom et al. 2003; Fig. 3). Consequently, the presence of perennial species can provide substantial biotic resistance against invasion (Blumenthal et al. 2003, 2005; Seabloom et al. 2003; Bakker and Wilson 2004; Levine et al. 2004). Where disturbance damages or removes resident species, it can increase resource availability and therefore provide opportunities for invasive species (Hobbs and Huenneke 1992; Davis et al. 2000). However, where desired or native species can be restored, their presence may be sufficient to reduce resource availability and keep invasions from recurring (Blumenthal et al. 2003; Seabloom et al. 2003).

Few studies have actually tested whether restoration leads to persistent invasive plant control. Ferrell et al. (1998) studied the response of leafy spurge to herbicides, tillage, and then seeding of several native and nonnative grass monocultures. Five years postseeding, one grass was very rare, but the others substantially suppressed leafy spurge. The two most effective species (which were nonnative) were reassessed 10 yr postseeding, and these species continued to suppress leafy spurge. Similarly, Bottoms and Whitson (1998) found that herbicides and tillage followed by seeding of several native and nonnative grasses greatly suppressed Russian knapweed 5 yr postseeding. Herbicide, tillage, and seeding of native tallgrass prairie species greatly suppressed weeds in an old field 7 yr after seeding (Blumenthal et al. 2003, 2005). Although most of the weed species in this study are not considered aggressive invaders of rangeland, two of the species inhibited by restoration can be desired species or invaders: Kentucky bluegrass and smooth brome (*Bromus inermis* Leys.). Finally, Seabloom et al. (2003) found that 5 yr after restoration, native perennials comprised the majority of the plant biomass in an area otherwise dominated by exotic annuals. These studies suggest seeding may sometimes provide cost-effective, long-term weed control. However, additional longer-term measurements

are desperately needed to better evaluate the long-term benefits of seeding.

#### *Altering Disturbance Regimes as a Solution to Changing Environmental Conditions.*

Where invasion is caused by past disturbance, long-term weed control and restoration may often be achieved with a combination of invasive weed control and seeding. Where invasion is caused by ongoing disturbance, however, or where past changes have led to new stable states, it may also be necessary to change the disturbance regime (Suding et al. 2004). The most important example of this problem in North American rangeland ecosystems is the increase in fire frequency that is both caused by and helps to perpetuate cheatgrass invasion (D'Antonio and Vitousek 1992). Because shrub-steppe ecosystems are not well adapted to frequent fire, the new disturbance regime appears to preclude successful restoration. Consequently, fire suppression is required to allow native species to compete against cheatgrass and other fire-tolerant invaders (Brooks et al. 2004). Fire suppression may or may not be sufficient to allow native species to compete against cheatgrass. If cheatgrass invasion is caused by a combination of fire and past disturbance, then fire suppression and restoration may be sufficient. If cheatgrass invasion is also driven by ongoing changes, such as nitrogen deposition or the amount/timing of grazing, then fire suppression and restoration may not be sufficient. In contrast to shrub-steppe ecosystems, tallgrass prairie evolved with frequent fire. There, the absence of fire can lead to invasion, and prescribed fire can be a long-term solution to invasion (Smith and Knapp 1999; Copeland et al. 2002). Similarly, grazing can be a disturbance that either favors or inhibits invasive plants, depending on the grazing history of the site (Mack and Thompson 1982; Milchunas et al. 1988, 1992; Bock et al. 2007).

#### *Grazing as a Cause and Solution to Invasion.*

Invasive species have evolved with grazing animals, creating a complex relationship among grazing preferences and plants' abilities to resist and tolerate grazing (Heitschmidt and Stuth 1991). Invasion can increase when competitive, desired species are defoliated beyond their ability to recover by the following growing season (Sheley et al. 1997).

Thus, reestablishing proper grazing regimes that maintain vigorous plants and healthy plant communities can limit invasion (Sheley et al. 2008). Similarly, lack of grazing of invaders can stimulate invasion. Since goats and sheep dominate most grazing in areas where many invasive weeds evolved, introducing them into areas with serious infestations of these species can reduce invasive weeds (Olson 1999).

*Biological Control as a Solution for Invaders that have Escaped Natural Enemies.* It is also possible that invaders succeed, overcoming biotic resistance, without any change in environmental conditions. Invaders may have an advantage over native species for other reasons. For example, the enemy release hypothesis proposes that invasive plants have an advantage over native species because they have escaped natural enemies when introduced to a new range (Maron and Vila 2001; Keane and Crawley 2002; Mitchell and Power 2003; Colautti et al. 2004; Blumenthal 2006). When enemy release is driving invasion, biological control, reuniting invasive species with specialized enemies, may be the most durable control method (Fig. 4). The reversal of enemy release by biological control is inexact. Only a subset of the enemies from the original range is introduced. The introduced enemies, however, are often missing their own predators, and may influence the invasive weed more strongly than they would have in their native range (Keane and Crawley 2002).

*Long-Term Solutions When Causes of Invasion Cannot Be Reversed.* All of the above sections discuss situations in which reversing the cause of invasion should help native species compete effectively against invasive species. However, not all causes of invasion can be reversed. For example, if atmospheric CO<sub>2</sub> enrichment causes invasion of otherwise healthy native plant communities (Smith et al. 2000), there may be no way to manage the environment to enable the native plant community to resist invasion. Other types of global change, such as nitrogen deposition and altered precipitation, may lead to similar problems (Dukes and Mooney 1999; Brooks 2003; Vila et al. 2007; Blumenthal et al. 2008). In such situations, efforts to increase biotic resistance, and therefore achieve long-term weed control, may require the use



Effects of flea beetles (*Apthona* spp.) on leafy spurge in MT

**Before flea beetles - 1994**



**After flea beetles - 1996**

**FIGURE 4.** Flea beetle (*Apthona* spp.) (right photo) impact on leafy spurge in Montana (bottom photo; 1996) 3 years following their introduction in 1994 (top photo).

of plant species from outside the local plant community (Seastedt et al. 2008). There may also be situations in which long-term solutions simply do not exist. If invasive species are better adapted to local conditions than are desired species and the local conditions cannot be altered, management choices may be limited to relatively expensive ongoing control or learning to manage the invasive species as novel ecosystems. Given the frequency of novel disturbances, opportunities for grazing management, and enemy release, it appears possible that the causes of invasion could be reversed in many cases. Where possible these approaches are likely to lead to more persistent invasive weed control and more stable desired plant communities.

### **Restore Desired Vegetative Cover to Protect Soils, Control Erosion, Reduce Sediment, Improve Water Quality and Quantity, and Enhance Stream Flow**

Plant community structure and composition are important drivers of ecosystem function and services, including protection and conservation of soil and water resources (Chapin et al. 2000). Our ability to reestablish desired vegetation on invasive plant-infested

Invasive species can have a significant impact on soil resources by altering plant cover, litter inputs, and the amount and distribution of bare ground. (Photo: Alex Boehm)



rangeland is extremely limited, particularly in areas with low precipitation. The proportions of plant cover and bare ground are generally the most important factors that determine the degree to which soil and water resources are protected and conserved by a plant community. To this end, when desired vegetation has been reestablished on degraded rangeland with low total vegetative cover and a high proportion of bare ground, protection and conservation of soil and water resources generally increase (Pyke et al. 2002; Pierson et al. 2007a). The bulk of restoration programs, however, have not evaluated if reestablishment of desired vegetation has effectively impacted soil or water resources. Of the few synthetic efforts to date on rangeland, there is little evidence to suggest that reseeding efforts are successful in establishing enough plant cover to significantly improve protection of soil and water resources beyond what is attained through natural site recovery processes (Pyke et al. 2003; Byers 2004). Of equal importance is the notion that weedy plant communities and desired plant communities may not necessarily differ in their ability to protect and conserve soil and water resources. In some instances, weedy plant communities may be important in rapidly stabilizing heavily disturbed communities on steep slopes, preventing loss and damage to soil and water resources (Pierson et al. 2007b). Nevertheless, there are a number of instances where restoring desired vegetation cover in weed-infested communities may benefit soil and water resources. Although empirical data are limited, some general principles have emerged that should allow reasonable prediction of when restoring desired vegetative cover on weed-infested rangeland may achieve these benefits.

**Soil Resources.** Although a few studies show that establishing a desired vegetation cover on weed-infested rangeland can increase protection of soil resources (Lacey et al. 1989), the bulk of evidence is largely observational (Sperber et al. 2003), limiting our ability to develop generalities. However, processes and factors associated with effective soil conservation are fairly well defined, which may allow relatively accurate predictions of when restoration of desired vegetative cover will provide soil conservation benefits. Plant cover as well as the proportion and connectivity of bare ground are central factors determining erosion and sediment

yield. More canopy cover lowers the effective energy of raindrops as well as the amount of soil exposed to rainfall impact (Blackburn et al. 1994). Large, interconnected patches of bare ground concentrate runoff and increase flow velocities and erosion (Schlesinger et al. 1990; Pierson et al. 2007a). The evidence suggesting that weeds alter soil physical properties is mixed, and appears species-specific (Sperber et al. 2003; Norton et al. 2004). The most significant impacts weeds have on soil resources are related to the degree to which weeds affect plant cover, litter inputs, and the amount and distribution of bare ground (Lacey et al. 1989; Pierson et al. 2007a). Therefore, soil conservation benefits may be achieved if restoring desired vegetation on weed-infested rangeland increases plant cover and/or decreases connectivity of bare patches and plant interspaces. On the other hand, if restoration efforts do not significantly alter these parameters, then establishing desired vegetation may not significantly improve conservation of soil resources.

**Water Resources.** The large negative effect of invasive plant species on water resources and the conservation benefit achieved by restoring desirable species is partially documented in rangeland riparian systems (Zavaleta 2000; Shafroth et al. 2005). However, on upland systems the benefits of restoring desired species on weed-infested rangeland are less well studied and the effects more nuanced. Several case studies examine weed effects on water resources on uplands and there have been extensive studies on individual plant water use patterns (Lambers et al. 2000; Enloe et al. 2004; Kulmatiski et al. 2006). Therefore, although the data on hand are limited, this information can be used to develop some general predictions as to the effects of weeds on water resources as well as the conservation benefits that may be obtained by establishing desired species on weed-infested rangeland. Patterns and rates of plant water use are determined by plant size, phenology, rooting depth, and root densities (Lambers et al. 2000). Weeds that differ significantly from desired vegetation in these traits have the potential to alter the pattern and amount of water available on rangeland. For example, annual grasses that have invaded sagebrush steppe systems initiate growth and use water earlier in the growing season compared to the native perennial bunchgrasses (Kulmatiski



et al. 2006). As a consequence, these weedy plant communities extract water at a faster rate earlier in the growing season than the desired plant community. In this case, eliminating the weeds and restoring desired perennial plants may allow water to remain available to plants for a longer duration during the growing season. As a contrasting example, deep-rooted forbs that invade native bunchgrass communities can deplete soil water at greater depths later in the growing season compared to bunchgrasses (Enloe et al. 2004; Fig. 5). In this case, restoring desired bunchgrasses may help conserve deep soil water.

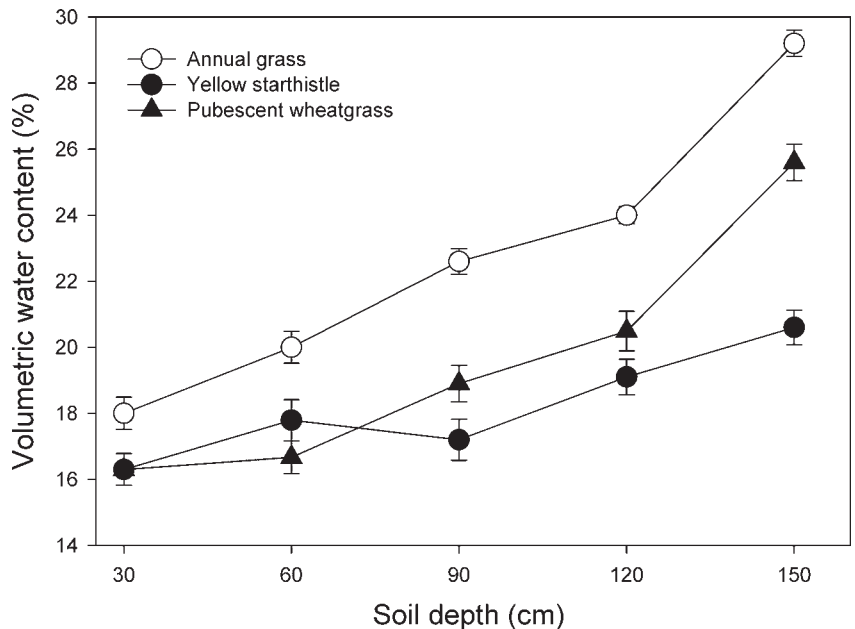
#### Conclusions and Management Implications.

Our ability to restore desired vegetation on arid and semiarid rangeland is limited. Restoring desired species may not always result in a conservation benefit in terms of soil and water resources. Although few studies have examined the conservation benefits of establishing desired species, basic knowledge about soil stability and hydrological processes allows reasonable prediction of scenarios where restoring desired species will benefit soil and water resources. Namely, if restoring desired species increases cover or litter inputs and/or decreases the amount or continuity of bare ground, a soil and water conservation benefit will likely be achieved. The impact of weedy plants on water resources and the benefits achieved by restoring desired species will mainly depend on the degree to which these species groups differ in size, phenology, rooting depth, and root densities. When these species groups exhibit large differences in one or more traits, substantial conservation benefits may be achieved. On the other hand, when these differences are small the conservation benefits may be negligible.

#### Maintain or Enhance Wildlife Habitat Including that Associated with Threatened and Endangered Species

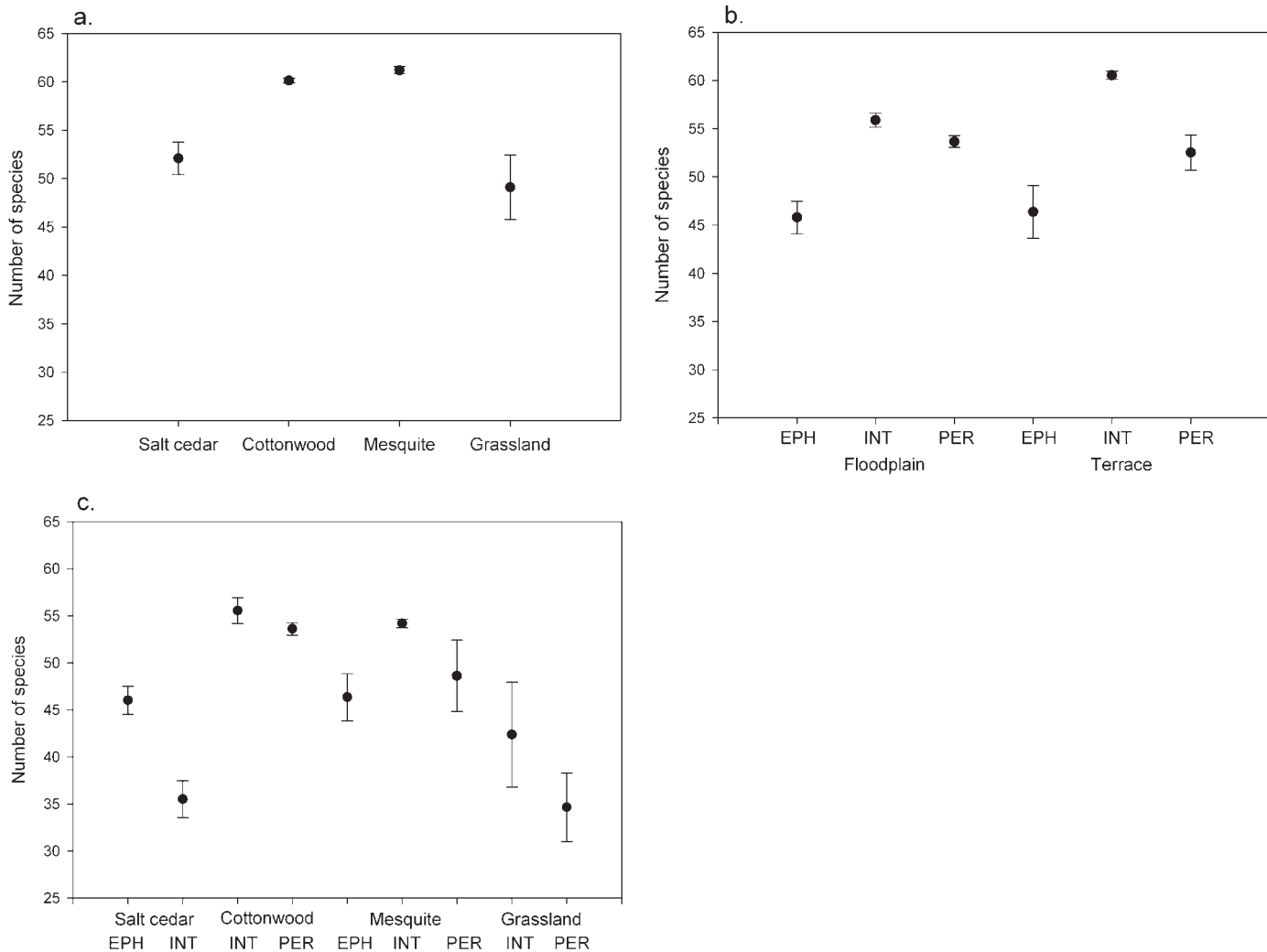
Invasive plant species often change ecosystem structure and function, directly impacting wildlife habitat (DiTomaso 2000; Masters and Sheley 2001). It is not surprising, therefore, that invasive species removal has been shown to benefit wildlife in a number of systems. Relationships between invasive plant species and wildlife, however, are often more complicated, involving both positive and negative effects. For example, saltcedar

**FIGURE 5.** Volumetric soil water content (%; mean  $\pm$  SE) by depth averaged across time of sampling and year. Plant community by soil depth comparisons: yellow starthistle vs. annual grasses ( $F = 3.33$ ,  $P = 0.0262$ ), yellow starthistle vs. pubescent wheatgrass ( $F = 0.27$ ,  $P = 0.8738$ ).



(*Tamarix* spp.) can provide suitable habitat for the endangered southwestern willow flycatcher (*Empidonax traillii extimus* Nelson) and other avian species that nest in midcanopy vegetation, but poor habitat for many other avian species (Dudley and DeLoach 2004; Shafroth et al. 2005; Brand et al. 2008; Fig. 6). The net benefit of invasive species control on wildlife habitat not only depends on the balance between negative and positive effects of invasive plants on wildlife habitat, but also upon the likelihood and time required for successful restoration, as well as the direct impact of invasive species control measures on wildlife and their habitat (Bateman et al. 2008a).

The impact of invasive species on wildlife habitat, and therefore the benefits gained by restoring these habitats, may be relatively predictable based on what is known about the habitat requirements of particular species. For example, deer, elk, and bison rely heavily on grasses. When grasslands are invaded by weedy invasive forbs, grass production declines and animal use of these habitats can decline by up to 80% (Thompson 1996; Rice et al. 1997b; Duncan 2005).



**FIGURE 6.** Avian species richness (mean  $\pm$  SE) as a function of **a**, four vegetation types, **b**, three hydrologic regimes within riparian location (floodplain and terrace), and **c**, nine combined vegetation–hydrologic regime classes on the San Pedro River, Arizona, 1998–2001. EPH indicates ephemeral surface water flow; INT, intermittent surface water flow; and PER, perennial water flow.

Similar effects have been observed on bird populations that prefer open grasslands (Scheiman et al. 2003). In cases where invasive plants alter the preferred forage base or structural characteristics of the native plant community, restoring these systems likely will have a large positive effect on wildlife.

Even if community structure is not altered, restoring natural patterns of plant species abundance may greatly improve habitat. For example, in western Oregon grassland, tall oat grass (*Arrhenatherum elatius* [L.] P. Beauv. ex J. Presl & C. Presl) reduces grassland use and egg laying by the endangered Fender’s blue butterfly (*Icaricia icarioides fenderi*

Macy) (Severns 2008). Oat grass appears to reduce use largely by obscuring the butterfly’s preferred host plant, Kincaid’s lupine (*Lupinus oregonus* A. Heller var. *kincaidii* C.P. Sm.), even when the lupine is present for butterfly use. Plant invasions are likely to have the greatest influence on wildlife when their presence leads to feedbacks that not only change the plant community structure, but also alter ecosystem properties.

For example, in North American rangeland, cheatgrass probably has the most widespread and severe effects on wildlife of any invasive plant. By changing the fire regime, cheatgrass can displace shrub-steppe vegetation and

associated wildlife species. Furthermore, cheatgrass can lead to such changes over extremely large areas (Fig. 7). In the Intermountain West, it has been estimated to occupy 40 million ha (DiTomaso 2000). Of primary concern are rare species such as greater sage grouse (*Centrocercus urophasianus* Bonaparte), Gunnison sage grouse (*Centrocercus minimus*), Brewer's sparrows (*Spizella breweri*), sage sparrows (*Amphispiza belli*), sage thrashers (*Oreoscoptes montanus*), and pygmy rabbits (*Brachylagus idahoensis*). Sage grouse are considered to be sagebrush obligates (Schroeder et al. 2004), and are most likely to persist in large areas with at least 25% sagebrush cover (Aldridge et al. 2008). Both greater sage grouse and Gunnison sage grouse have been proposed for listing under the Endangered Species Act. Although prescribed fire has in the past been suggested as a tool to improve sage grouse habitat, recent studies suggest fire is most often harmful to sage grouse, particularly the frequent fire caused by cheatgrass invasion (Connelly et al. 2000; Baker 2006). Invasion of sagebrush by cheatgrass, and associated increases in fire frequency, appear to be primary causes of sage grouse decline (Knick et al. 2003; Schroeder et al. 2004; Baker 2006).

Other sagebrush-obligate wildlife species are likely to be similarly influenced by cheatgrass invasion and loss of sagebrush. Quantification of habitat requirements shows that many species considered to rely on sagebrush, such as pygmy rabbits, sage thrashers, and sage sparrows, do in fact have habitats that overlap strongly with those of sage grouse (Rowland et al. 2006). In eastern Washington, not only shrub-nesting sage sparrows, but also a variety of ground-nesting birds, were found to be less abundant in areas dominated by cheatgrass than in shrub-grass plant communities (Brandt and Rickard 1994). A variety of avian species have also been shown to prefer native perennial grass seed to cheatgrass seed (Goebel and Berry 1976). Small mammals can also be strongly influenced by cheatgrass invasion. For example bitterbrush-dominated communities have been found to support 3–13 times the densities of small mammals of cheatgrass-dominated communities in central Washington (Gano and Rickard 1982; Gitzen et al. 2001). Similarly, small-mammal densities and richness were

higher in intact sagebrush steppe than in areas with cheatgrass in Idaho's Snake River Plain (Hanser and Huntly 2006). Finally, a study of Townsends ground squirrels (*Spermophilus townsendii idahoensis* Merriam), found higher variation in squirrel burrows in cheatgrass habitats than in shrub-bunchgrass habitats (despite similar mean burrow numbers), suggesting cheatgrass provides an adequate but unstable food resource for this species (Yensen et al. 1992).

**Beneficial Effects of Invasive Plant Management on Wildlife.** Only a handful of studies have actually measured effects of rangeland weed control on wildlife or wildlife habitat. Chemical control of spotted knapweed control was found to release grasses from competition and increase winter forage for elk by 47% in western Montana (Rice et al. 1997b). Other wildlife species rely heavily on native forbs for food. Consequently, restoration of native forbs can be an important objective of invasive species control. Fall wick application of glyphosate controlled Canada thistle (*Cirsium arvense* [L.] Scop.) while increasing shrub biomass, forb biomass, and species richness in a Montana waterfowl production area (Krueger-Mangold et al. 2002). Burning can also favor native forbs if conducted at the right time. In California, burning in the late spring and early summer

**FIGURE 7.** *Bromus tectorum* invasion in shortgrass steppe vegetation near Lander, Wyoming. (Photo: D. Blumenthal)







There are too few direct measurements of wildlife responses to invasive species control to gauge how often control yields benefits for wildlife.”

after many forbs have set seed can favor native forbs over invasive annual grasses (DiTomaso et al. 2006b; Meyer and Schiffman 2008). Similarly, winter–spring cattle grazing appears to reduce annual grass invasion and favor native forbs that provide key habitat for several butterfly species on California serpentine grasslands (Weiss 1999b). A few studies have directly documented increased wildlife use following invasive species control. For example, successful control of spotted knapweed with picloram increased elk foraging in a Montana old field (Thompson 1996). Similarly, mechanical and chemical control of saltcedar and Russian olive (*Elaeagnus angustifolia* L.) in New Mexico floodplain forest increased the abundance of lizards and bats, but decreased the abundance of birds that nest in midstory vegetation (Bateman et al. 2008a, 2008b).

**Detrimental Effects of Invasive Species Management on Wildlife.** Although reduced abundance of invasive species is likely to benefit wildlife, the methods used to reduce invasive species abundance can sometimes harm wildlife. A potentially important example of direct effects of herbicides on wildlife can be found in recent work suggesting that atrazine plays a role in global declines in amphibian populations (Rohr et al. 2008). Both field surveys in Minnesota wetlands and mesocosm experiments showed atrazine to be associated with increased infection by trematodes (a likely proximate cause of amphibian declines) in northern leopard frogs (*Rana pipiens* Schreber). Increased infection, in turn, appears to be caused by both an increase in the abundance of gastropods, which are intermediate hosts for trematodes, and decreased immune responses on the part of the frogs (Rohr et al. 2008).

Invasive species control can also harm wildlife indirectly, through its effects on the plant community. In particular, chemicals that target dicots can decrease plant community diversity, thereby reducing the food available for some wildlife species (Johnson et al. 1996a; Sheley et al. 2007). For example, 2,4-D applied to western Colorado rangeland to favor grasses over forbs and shrubs reduced densities of northern pocket gophers (*Thomomys talpoides* Richardson) and least chipmunks (*Eutamias minimus* Bachman), while increasing densities

of montane voles (*Microtus montanus* Peale; Johnson and Hansen 1969). These effects appear to have been caused by a combination of reduced food availability and changes in vegetation cover. A proposed alternative for controlling woody species without harming wildlife is tebuthiuron, which can reduce woody species without reducing forb abundance and diversity (Johnson et al. 1996b).

#### *Conclusions and Management Implications.*

There is considerable evidence that invasive plants influence wildlife in rangelands. Most often this influence is negative, reducing food and habitat availability for a wide array of wildlife species. Evidence for negative effects on wildlife is particularly strong for invaders that alter ecosystem structure and function, such as weedy forbs invading grasslands and annual grasses invading systems historically dominated by perennial plants. There are too few direct measurements of wildlife responses to invasive species control to gauge how often control yields benefits for wildlife. Rather, wildlife benefits must be inferred from what is known about both the relative value of invasive and native plant species as wildlife habitat, and the effectiveness of management in replacing invasive species with native species.

#### **Protect Life and Property from Wildfire Hazards**

Wildfires are a regular and natural occurrence in many areas of the arid western United States and most of these ecosystems are well adapted to fires (Brooks et al. 2004). These natural ecosystems will return to their preburn state within a few years of a fire under normal conditions. However, other habitats such as riparian corridors, sagebrush scrub, and deserts have longer fire-return intervals because of sparse and discontinuous vegetation. In these areas, the native species are less adapted to fire and are susceptible to a short-duration fire interval (Brooks et al. 2004). Invasion by annual grasses, particularly cheatgrass, red brome (*Bromus rubens* L.), and medusahead, have dramatically shortened the intervals between fires by providing more continuous fuels that are easier to ignite (Brooks et al. 2004). In addition, invasive annual grasses typically reestablish more rapidly than native plants after fires. This can further suppress the

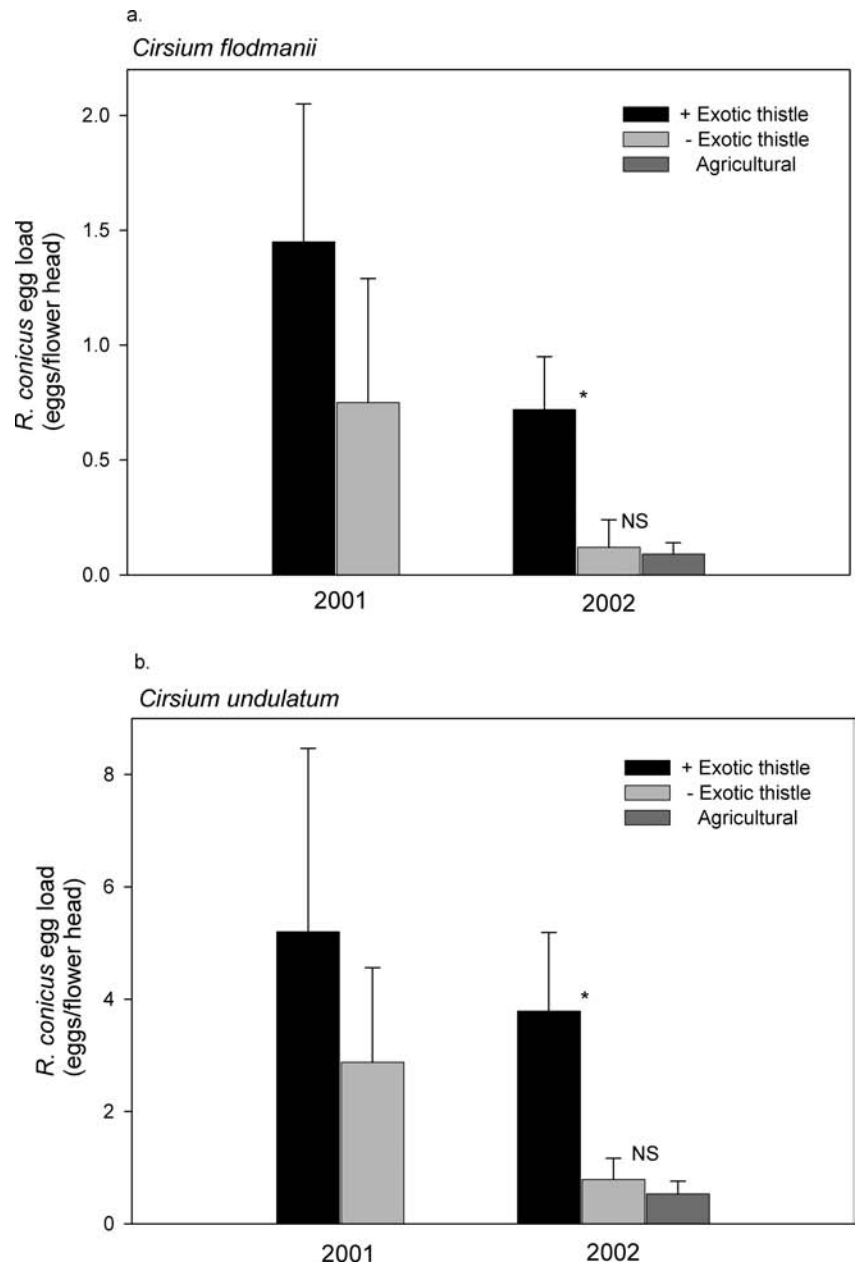
recovery of the natives and allow the weeds to expand their range (Pellant 1990).

More importantly, if fires occur too frequently, some of the native vegetation becomes so severely damaged that recovery is no longer possible (Pellant 1990; Whisenant 1990). This can result in loss of woody species such as sagebrush and other important plants and wildlife species, and effectively convert high-diversity native plant communities into low-diversity nonnative communities (Knick 1999). In some cases, fire exclusion over a period of time can create undesirable conditions for both forest sustainability and human fire hazard (Keeley 2006). This is the situation with some woody species, such as western juniper (*Juniperus occidentalis* Hook; Coultrap et al. 2008), which has expanded its range dramatically in the northwestern United States.

Land management agencies, such as the USDA Forest Service, Bureau of Land Management (BLM), and National Park Service, are required to assess site conditions following wildfire. Where necessary, they can prescribe emergency watershed-rehabilitation measures that can 1) help stabilize soil; 2) control water, sediment, and debris movement; 3) prevent permanent impairment of ecosystem structure and function; and 4) mitigate significant threats to human health, safety, life, property, or downstream values (USDA Forest Service 2010). Each year millions of dollars are spent on emergency post-fire rehabilitation treatments (Robichaud et al. 2000).

In southern California, where chaparral communities are prone to fire at the wildland-urban interface and the societal impacts of accelerated postfire erosion are enormous, there are pressures to treat burned hill slopes with grass seed to protect life and property (Gibbons 1995). It was common to seed such areas with quick growing annual plants, typically nonnative annual ryegrass or collections of native and nonnative forbs. This practice, however, is no longer recommended because the results are often unsuccessful. In some cases, heavy rains can wash away seeds, or inadequate rainfall prevents good seed germination. In addition, some of plants used for reseeding can persist and add to

**FIGURE 8.** *Rhinocyllus conicus* egg load (untransformed means  $\pm$  1 SE) on two native thistle species—**a**, *Cirsium flodmanii* and **b**, *Cirsium undulate*—in grassland patches within two landscape types in 2001, and three landscape types in 2002. Results of planned contrasts comparing landscape pairs are presented above bars for 2002. NS indicates not significant;  $P < 0.05$ .



the invasive plant problem (Bell et al. 2007) by competing with the native vegetation and preventing recovery. Long-term slope stabilization is better achieved by promoting the recovery of deep-rooted perennial shrubs compared to shallow-rooted annuals. This can be accomplished by transplanting shrubs or by

protecting establishing shrubs from herbivory or competition from nonnative species. Shrub recovery can lead to reduced threat of subsequent fires (Bell et al. 2007).

#### *Conclusions and Management Implications.*

There is a considerable amount of evidence to demonstrate the impact of invasive plants, particularly annual grasses, on the frequency of fires in rangeland systems. In addition, it is well recognized that rangeland fires spread by invasive plants can cause significant damage to property and human health. Although few studies have been conducted on the interaction between invasive plants, wildfire, and impacts to wildlife, it stands to reason that these impacts are significant and in most cases detrimental to wildlife. With increased research on methods to control vegetation and protect areas from large catastrophic fires, the economic and ecological damage caused by invasive plants can be substantially reduced in the future.

#### **Minimize Negative Impacts of Pest Control on Soil, Water, Air, Plant, and Animal Resources**

Minimizing negative impacts of pest control on biotic and abiotic resources is an important step in designing economically and ecologically sustainable invasive plant management practices (Sheley et al. 2010). The most commonly applied control strategies for invasive plants on rangeland include herbicides, biocontrol, grazing, fire, or mechanical control such as tilling (Jacobs et al. 1999). Impacts of these control strategies on abiotic and biotic resources have been assessed to varying degrees and in some cases, general ecological patterns and principles are beginning to emerge. For example, the fate and ecological impact of various herbicides on rangelands has been documented and there is much evidence suggesting that as disturbance (e.g., herbicide use, tilling, grazing) intensity increases, invasibility of a system also increases (Hobbs and Huenneke 1992; Davis et al. 2000). Nevertheless, large gaps in our understanding of pest control impacts on abiotic and biotic resources remain. For example, a key component of ecologically based invasive plant management is to apply pest control strategies that reduce the performance of invasive species more than

the performance of desirable species (Sheley et al. 2006). However, a Web of Science query that included the search terms “herbicide” and “rangeland” demonstrated that only 28% (20 of 70) of field studies published between 1976 and 2008 examined herbicide effects on both desirable and weedy vegetation.

#### *Impacts of Control on Soil, Water, and Air Resources.*

Impacts of pest control on soil, water, and air resources vary depending on pest control strategy, but in general, effects are relatively predictable. For example, intense soil disturbances contribute to erosion, decreased water quality, and dust production and also release nutrients, which favors the growth of weeds compared to natives (Greene et al. 1994; Davis et al. 2000; McEldowney et al. 2002; Zhao et al. 2005). Because of this, current management frameworks for weed-infested rangeland focus on using tools that will minimize disturbance such as no-till drills and moderate grazing in efforts to direct a plant community toward a more desirable state (Mangold et al. 2006).

Concerns over the effects of herbicides on soil, water, and air resources have been raised due to the potential of herbicides to affect soil processes, to contaminate groundwater, or to be transported on wind-eroded sediment and potentially inhaled by humans (Larney et al. 1999; Liphadzi et al. 2005; Borggaard and Gimsing 2008). The impact of these herbicides on these resources is dependent on type of herbicide used, application rate, and soil characteristics, among other factors. For example, glyphosate tightly adheres to soil, which makes it difficult for this compound to leach into groundwater or affect soil biological processes (Borggaard and Gimsing 2008). On the other hand, compounds such as dicamba and picloram are highly mobile in the soil (Krzyszowska et al. 1994). High application rates, high rainfall following application, or direct application of these compounds to water bodies can pose a significant threat to water resources. Overall, careful application of herbicide following recommended procedures coupled with the relatively low application rate of herbicides commonly used on rangeland tends to minimize the negative effects of herbicides on rangeland soil, water, and air resources.



## Impacts of Pest Control on Plant and Animal Resources

**Herbicides.** Only a subset of studies (20 of 70) has examined herbicide effects on both invasive and desirable plant species in the field. Although the responses are dependent on a number of factors, such as mode of herbicide action and site-specific environmental conditions, two important trends have emerged. First, desirable species functionally or taxonomically similar to the invasive plant species targeted for control tend to be more negatively impacted by herbicide application. For example, herbicides such as 2,4-D, clopyralid, or picloram are commonly applied to control broadleaf weeds such as knapweed, leafy spurge, and sulfur cinquefoil on rangeland. Because grasses are capable of metabolizing these compounds, desirable rangeland grasses are generally unaffected by these herbicides (Sheley and Jacobs 1997; Sheley et al. 2002; Laufenberg et al. 2005). However, these herbicides can greatly decrease native forb density and cover (Sheley and Denny 2006). There is evidence suggesting that herbicide effects on native forbs are long-lasting and can drive a local decline in species richness (Fuhlendorf et al. 2002; Rinella et al. 2009). As another example, desirable rangeland grasses have shown varying degrees of susceptibility to imazapic, a herbicide used to control invasive annual grasses, with evidence suggesting grasses within the *Hordeae* tribe may be more tolerant to imazapic than other grass species (Kyser et al. 2007). A second trend is that the impact of herbicides on desirable vegetation depends on the rate and timing of herbicide application. In general, when herbicides are applied several weeks prior to seeding or during a dormant seeding, herbicides have a greater selective effect for weeds compared to seeded species (Jacobs et al. 1999; Kyser et al. 2007; Sheley 2007). Higher herbicide application rates can have negative impacts on seeded species, even during fall dormant plantings (Monaco et al. 2005). Even with a given rate and timing of herbicide application, desirable species response can vary substantially across sites in a given year and across years in a given site (Monaco et al. 2005; Sheley et al. 2007). Beyond a few generalities, the effect of herbicide on desirable vegetation remains difficult to predict.



Effects of herbicides on mammals, birds, and invertebrates are generally identified during the ecological risk assessment prepared with each herbicide and for public land management activities during environmental impact reporting. Although a number of herbicides are available to control weeds on rangeland, 70% of the land treated with herbicides by the BLM uses 2,4-D, glyphosate, picloram, tebuthiuron, or imazapic. Of these, glyphosate, picloram, and imazapic show low toxicity to terrestrial animals whereas tebuthiuron and 2,4-D demonstrate moderate toxicity. The low rates of herbicide applied on rangeland combined with relatively low toxicity and lack of chronic exposure suggest herbicides have minimal effect on terrestrial animal species on rangeland.

**Biocontrol.** Development and release of biocontrols follows international and national guidelines designed to minimize the possibility that biocontrol releases will negatively impact desirable vegetation (FAO 1996; Wilson and McCaffrey 1999). Biological control has been implemented successfully in a number of systems (e.g., Huffaker and Kennett 1959; McEvoy et al. 1991; Lym 2005) and when operating under current protocols there are relatively few documented direct effects of biological control on desirable vegetation given the number of biocontrol releases

The impacts of herbicides on desirable and undesirable species depend on the rate and timing of the herbicide application. (Photo: Rob Wilson)

(Funasaki et al. 1988; Center 1995). There is, however, mounting evidence suggesting that poor monitoring efforts, difficulty in predicting biocontrol effects, and the largely unrecognized indirect effects biocontrols can have on ecosystems contributes to an underestimation of the detrimental effects of biocontrols on desirable vegetation (Simberloff and Stiling 1996; Thomas and Willis 1998; Pearson and Callaway 2008). For example, the bulk of biocontrol monitoring focuses on release sites with little attention paid to offsite biocontrol effects even though there is strong evidence demonstrating landscape-scale variation in biocontrol effects on desirable vegetation (Simberloff and Stiling 1996; Rand and Louda 2004). In addition, it is estimated that less than half of the biological control efforts targeting invasive plants in the United States demonstrated any evidence of control (OTA 1995). Given that our ability to predict biocontrol effects on well-studied target vegetation is so low, some researchers have questioned the ability to predict biocontrol effects on desirable vegetation (Thomas and Willis 1998). Although examples of direct effects of biological control on desirable vegetation in a number of systems supports these concerns (Simberloff 1992), of equal importance is the recent literature showing complex indirect effects of biocontrol on desirable vegetation. For example, following the collapse of the target pest population, intense competition among biocontrol agents can cause a transient increase in host plant range, which results in the biocontrol agents attacking desirable vegetation (Lynch et al. 2002). Alternatively, when biocontrol agents only moderately damage invasive plants they may increase invasive plant competitive ability by stimulating compensatory growth (Callaway et al. 1999). In this situation, moderately damaged invasive plants may serve to maintain biocontrol densities at high levels, increasing biocontrol impacts on desirable vegetation (Rand and Louda 2004; Figs. 8a and 8b). These patterns of responses suggest, at a minimum, that current procedures do not adequately prevent biocontrol efforts from having significant impacts on desirable vegetation.

Concerns over the effects of biocontrol on animal resources largely have been centered on within-guild (e.g., insect) interactions. For

example, if an introduced biocontrol insect shares food sources or parasites with a native insect, then biocontrol can have direct and indirect effects on native insect populations (Louda et al. 1997; Willis and Memmott 2005). An analysis of 17 food webs in Australia showed that a weed biocontrol agent with high weed host specificity was associated with a decline in native insect diversity (Carvalho et al. 2008). Although the magnitude of these direct and indirect effects are difficult to quantify and are generally underreported in the literature, basic community ecology theory predicts that such effects may be common (Holt 1977). In some cases, however, effects of biocontrol on a desirable plant community can be complex and difficult to predict, involving multiple interactions within a food chain. For example, introduction of gall flies to control spotted knapweed dramatically increased deer mouse populations that used gall flies as a food source (Ortega et al. 2004). Because deer mice also use native plant seed as a food source, introducing gall flies increased deer mouse populations which resulted in increased predation on native seeds and overall decrease in native plant density (Pearson and Callaway 2008). Although theory and empirical evidence suggest biocontrols likely will have some negative effect on native animal populations, biocontrol may still be an appropriate option if benefits outweigh the costs. Namely, if biocontrols have a large negative effect on weed populations, this benefit may outweigh moderate negative impacts of biocontrol on native plant and animal populations.

**Grazing.** Prescribed grazing effects on nontarget vegetation depend on a number of factors, including animal species used, timing of grazing relative to the phenology of desirable vegetation, and forage quality and quantity of weedy vegetation relative to desirable vegetation, as well as grazing tolerance of weedy and desirable species. Moderate grazing using animals or mixtures of animals (e.g., sheep and cattle) that demonstrate certain dietary preferences for a particular weed can be used to decrease weed density and increase density of desirable plants (Bowns and Bagley 1986; Sheley et al. 1998). In general, when grazing is limited to periods when weedy species are most susceptible to defoliation and desirable plants are largely dormant, the impact of grazing on



desirable vegetation can be minimized and benefit of grazing for weed control maximized (Kennett et al. 1992). For example, utilization of grasses by sheep in areas infested with knapweed was decreased by timing grazing to occur when knapweed was still growing and vegetative growth of desirable grasses had largely stopped for the season (Thrift et al. 2008). If weedy and desirable vegetation have comparable forage quality, grazing animals largely will consume plants in proportion to their abundance. For example, diets of sheep used to graze spotted knapweed were over 50% grasses in areas with low spotted knapweed density, but were less than 20% grasses in areas with high spotted knapweed density (Thrift et al. 2008). When weeds have much lower forage quality compared to desirable vegetation, grazing animals can have a larger preference for and a much greater negative impact on the desired vegetation (Ralphs et al. 2007). Despite these general guidelines it is difficult to predict the effect of grazing on desirable vegetation. For example, grazing leafy spurge infestations has been found to decrease (Jacobs et al. 2006), increase (Seefeldt et al. 2007), or have no

effect on (Lacey and Sheley 1996) the cover of desirable grasses. Although a portion of this variation may be explained by differences in grazing systems, differences in grazing tolerance between weedy and desirable species at a particular site also may be important (Kennett et al. 1992; Kirby et al. 1997; Olson and Wallander 1997). If weedy species demonstrate a greater tolerance to grazing than desirable species do, then prescribed grazing may be a counterproductive control strategy even if grazing animals demonstrate greater or equal preference for weedy species compared to desirable species (Kimball and Schiffman 2003). Although the value of prescribed grazing for weed control has been demonstrated in a number of systems, negative impacts on desirable vegetation have been demonstrated, highlighting the need to closely monitor prescribed grazing efforts.

#### *Conclusions and Management Implications.*

A key step in designing economically and ecologically sustainable invasive plant management practices is to apply management techniques that minimize negative impacts

Prescribed grazing can be an effective tool in reducing the vegetative growth of invasive species. (Photo: Brenda Smith)



on biotic and abiotic resources. Some general principles are beginning to emerge allowing progress to be made toward this goal, such as our understanding of the relationship between disturbance intensity and invasibility. Although some negative effects of pest control strategies on native plant and animal resources are likely, herbicides, biocontrol, and grazing can be applied in ways that greatly minimize these impacts if ecological processes and mechanisms are considered beforehand and control strategies are adjusted to address these factors. Identifying these processes and mechanisms and making necessary adjustments in management, however, is far from straight forward. Complex direct and indirect effects of control efforts on desirable plant and animal resources occur, requiring careful implementation of control efforts, comprehensive monitoring, and a broad determination of costs and benefits achieved by control efforts that include multiple ecosystem components.

## RECOMMENDATIONS AND KNOWLEDGE GAPS

Our recommendations are centered on three general aspects of invasive plant management. The first revolves around improving and standardizing data collection and risk analysis needed to better inform management decisions. Second, progress toward science-based management of rangeland threatened and/or dominated by invasive species must be greatly accelerated. Third, invasive plant management would greatly benefit from the development and implementation of a comprehensive education and technology transfer program. The objective of this portion of the document is to provide critical recommendations to guide future development of invasive weed management and to identify important knowledge gaps. A brief rationale and justification for each recommendation and knowledge gap are provided as well.

### Standardized Data Collection, Risk Analysis, and Prioritization Procedures

The magnitude and complexity of invasive plant management requires that ecologists garner maximum information from all datasets. Data collection for both invasive plants and desired species is central to

developing appropriate management programs in the future. Standardized data collection will be required in order to allow data comparisons among years and data combinations to conduct meta-analysis needed for development of robust principles for management. Managers need standardized data collection procedures to create accurate vegetation assessments that allow periodic evaluations of their management. Inventory data must be summarized and analyzed to forecast likely future vegetation patterns so ecological and economic risk/benefit analysis can be accurately conducted. Standardized ecological and economic data collection would be critically valuable to determine land areas with characteristics that favor the likelihood of success in response to a particular control strategy.

### Science-Based Solutions to Invasive Plant Management

Just as physics provides the scientific principles for engineering, ecology must provide the scientific principles for invasive plant management. We strongly recommend further development of ecologically based management frameworks that can be used to guide the incorporation and application of ecological principles for invasive plant management. Frameworks must be useful to researchers and managers, so the connection between these complementary endeavors is natural and direct. State-and-transition models that utilize ecological processes and the influence of management on these processes to predict vegetation dynamics represent a viable framework for various ecological site descriptions. A process- and evidence-based approach is central to advancing invasive plant management from misapplied treatments that address only symptoms to management programs that emphasize the underlying cause of invasion, retrogression, and succession.

Complex interrelationships among various components within ecosystems create multiple indirect responses to vegetation management that are very difficult to predict. This creates a strong need to manage invasive plants within the context of the entire ecosystem. Invasive plant management must become more integrated within a systems approach to



facilitate problem solving and the attainment of well-defined goals, rather than only practice-based outcomes. Management must assess the complex interrelationship among ecosystem components and processes and design management strategies that influence the underlying ecological cause of invasion and dominance by invaders with predictable outcomes.

Imposing management that addresses the actual cause of invasion is clear in some cases. For example, the increase in invasive wetland species in flooded waterfowl habitat on the Malheur National Wildlife Refuge requires flooding regimes to be less frequent, allowing substantial dry periods to shift the balance in favor of diverse vegetation. In many cases, the actual causes of invasion are less obvious and may actually be a result of multiple direct and indirect interactions that determine successional dynamics. Thus, weed ecologists and scientists must develop guidelines to evaluate causes of invasion,

succession, and retrogression. Once these guidelines are developed, ecological principles must be developed that provide guidelines for managers to impose tools and strategies to influence conditions, mechanisms, and processes in favor of desired vegetation. As multiple interactive ecological processes require amendment, integrated plant management strategies can be developed and employed much more effectively. In this way, various plant management strategies can be designed based on how the treatments influence the ecological processes that direct ecosystem change. Tools and strategies that are based on sound ecological principles could enhance our ability to employ effective integrated management.

Enhancing our ability to prevent invasion is critical for successful implementation of integrated invasive plant management. Given the complexity and persistence of invasive plants, a proactive approach focused on systematic prevention and early intervention

Invasive species management that addresses the actual cause of invasion and employs ecological principles in management strategies increases the likelihood of success. (Photo: Chris Call)





Practicing prevention of invasive species, such as medusahead, is more economical and more effective than costly restoration. (Photo: Ryan Steineckert)

could be much more effective than the existing reactive approach. Most managers recognize the importance of prevention, but lack the ability to effectively employ it. Science-based prevention strategies that are based on the ecology of seed dispersal are severely needed. Land managers need a conceptual framework and associated tools that assist them in indentifying which vectors are major contributors to invasive species dispersal and propose dispersal management strategies to minimize or interrupt these major vectors. Effective methods for containing existing infestations are also needed.

Invasive plant management is currently applied in a somewhat haphazard way based on political pressure and funding resources. In the future, more emphasis should be focused on prioritizing invasive plant management in areas that have the highest likelihood of success both economically and ecologically. Methods for prioritizing invasive plant management will continue to be increasingly necessary as a means to effectively allocate scarce resources. Moreover, the lack of successful control of invasive species indicates that we may transition toward a management philosophy that minimizes the negative impacts of invasive species and maximizes the ecological and economic benefits garnered from invasive weed management programs. Concepts, such as economic/ecological injury levels, biomass optimization models, and thresholds will need to be carefully developed in a manner

that helps managers prioritize management programs to address invasive species.

### **Comprehensive Education and Technology Transfer Programs**

Although some of the necessary infrastructure to conduct educational programs effectively is in place, a unified, progressive, and outcome-based educational and technology transfer program would have strong synergistic effects on invasive plant management. Educational programs vary widely in their objectives, content, and outcomes. Current programs lack continuity of message and the ability to progressively advance managers' understanding of science-based management. We propose that various ecological societies, managers, and researchers develop a comprehensive science-based curriculum promoting the most state-of-the-art, science-based assessment and management strategies. Once developed, training modules could be developed for various portions of the educational infrastructure having responsibility for natural resource extension and outreach.

Restoration of invasive plant-dominated rangeland is extraordinarily risky and expensive. Based on our assessment, the continued application of "farming system" seeding methods is unlikely to provide sustainable replacement of invasive species. Many ecological barriers to seed germination, seedling establishment, and population development exist in restoration areas where



invasive plants dominate. Managers must have an understanding of these barriers and methods for overcoming them if restoration is to become a useful strategy to restore previously invaded sites and prevent reinvasion in the future. Restoration approaches must be founded upon ecological principles that can be applied to specific sites and varied as environmental conditions vary across landscapes.

Invasive plant problems and solutions are complex and management outcomes are rarely predictable. Ecologists and managers are often uncertain about the best management practices to employ, or if management will actually repair plant communities and the associated ecological processes. In most cases, simple answers to complex situations do not exist and solutions to invasive plant problems are elusive. Managers need scientifically credible methods for testing various management strategies that can be used when management programs are being planned and implemented. These adaptive management strategies should include controls for comparisons and designs that use simple experimental hypothesis testing, in addition to monitoring previous effectiveness. A major strength of adaptive management is that it would allow managers to continuously evaluate the effectiveness of current invasive plant management programs and assist with identification of the most successful management programs.

## CONCLUSIONS

Invasive plants negatively impact rangelands throughout the western United States by displacing desirable species, altering ecological processes, reducing wildlife habitat, degrading systems, altering fire regimes, and decreasing forage productivity. Assessing the influence of conservation practices on the perceived benefits to ecosystems is critical to understanding their usefulness in maintaining sustainable ecological and economic systems. We conducted a comprehensive synthesis of peer-reviewed literature to determine the efficacy of various invasive plant strategies on several anticipated benefits. The literature documented only short-term vegetation responses to invasive plant management and rarely addressed long-term ecological outcomes associated with invasive plant management. Our ability to protect

noninfested lands is encumbered by the lack of early detection techniques and effective eradication efforts once new infestations are identified. Several strategies for maintaining invasion-resistant plant communities are beginning to emerge. Herbicides provided short-term control of most invasive weeds, but without additional management, weeds often return rapidly. Documentation of the efficacy of biological control on plant development is well established, but positive effects on control and vegetation dynamics are exceedingly rare. Grazing management is emerging as a useful method for managing invasive plant species, but the timing, intensity, and frequency of grazing, as well as the class of livestock are only known for a few invasive species. Restoration of infested rangeland is difficult and only successful about 20% of the time when nonnative plant material is seeded and the probability is even less when native species are used. There are cases in which invasive plant management strategies can be effective, and in those cases, the management strategies appear to favorably affect wildlife and other important ecological attributes of ecosystems. However, most strategies are associated with high ecological risks and high risk of failure in the long term. It is clear that more research is necessary if the anticipated benefits of invasive plant management are to be achieved. This synthesis indicates that long-term invasive plant management is lacking for most applications and that ecologically based invasive plant management is desperately needed to meet this escalating problem on rangelands.

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As invasive plants continue to spread across the landscape, a priority should be developing comprehensive education and technology transfer programs. (Photo: Ryan Steineckert)



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