Weed Technology. 2006. Volume 20:535–548

Review Article

Control of Invasive Weeds with Prescribed Burning[1]

JOSEPH M. DiTOMASO, MATTHEW L. BROOKS, EDITH B. ALLEN, RALPH MINNICH, PETER M. RICE, and GUY B. KYSER[2]

Abstract: Prescribed burning has primarily been used as a tool for the control of invasive late-season annual broadleaf and grass species, particularly yellow starthistle, medusahead, barb goatgrass, and several bromes. However, timely burning of a few invasive biennial broadleaves (e.g., sweetclover and garlic mustard), perennial grasses (e.g., bluegrasses and smooth brome), and woody species (e.g., brooms and Chinese tallow tree) also has been successful. In many cases, the effectiveness of prescribed burning can be enhanced when incorporated into an integrated vegetation management program. Although there are some excellent examples of successful use of prescribed burning for the control of invasive species, a limited number of species have been evaluated. In addition, few studies have measured the impact of prescribed burning on the long-term changes in plant communities, impacts to endangered plant species, effects on wildlife and insect populations, and alterations in soil biology, including nutrition, mycorrhizae, and hydrology. In this review, we evaluate the current state of knowledge on prescribed burning as a tool for invasive weed management.

Nomenclature: Barb goatgrass, *Aegilops triuncialis* L. #[3] AEGTR; Canada bluegrass, *Poa compressa* L. # POACO; Chinese tallow tree, *Sapium sebiferum* (L.) Roxb. # SAQSE; downy brome, *Bromus tectorum* L. # BROTE; French broom, *Genista monspessulana* (L.) L. Johnson # TLNMO; garlic mustard, *Alliaria petiolata* Andrz. # ALAPE; Kentucky bluegrass, *Poa pratensis* L. # POAPR; medusahead, *Taeniatherum caput-medusae* (L.) Nevski; red brome, *Bromus madritensis* L. ssp. *rubens* (L.) Husnot # BRORU; ripgut brome, *Bromus diandrus* Roth # BRODI; Scotch broom, *Cytisus scoparius* (L.) Link # SAOSC; smooth brome, *Bromus inermis* Leysser # BROIN; sweetclover, *Melilotus* spp.; yellow starthistle, *Centaurea solstitialis* L. # CENSO.

Additional index words: Fire, integrated vegetation management, rangelands, wildlands.

INTRODUCTION

Most ecosystems have adapted to some degree of fire disturbance. In many areas, natural fire regimes have been influenced by humans. Humans have used fire to manage vegetation since prehistoric times, when fire was used to improve opportunities for hunting and to encourage growth of useful plant species (Vale 2002). More recently, ''prescribed fire'' (controlled burn used to achieve a management objective) has been used to reduce fuel loads, restore historical disturbance regimes, improve forage and habitat, and promote biodiversity. Fire also has been used to manage invasive plant species, either directly or as part of an integrated approach. Much of what we know about using fire to manage vegetation is derived from studies of brush management and crop systems (Pyne 1997; Wright and Bailey 1982). However, there are fundamental differences between cropland and wildland settings (Table 1). The goal of this review is to capture the current state of knowledge on the use of fire to manage invasive plants in wildlands.

CONTROL OF INVASIVE PLANTS WITH PRESCRIBED FIRE

Long-term control of invasive species requires depletion of reproductive structures. To control annual species

[1] Received for publication June 23, 2005, and in revised form October 24, 2005.

[2] Cooperative Extension Specialist, University of California, Department of Plant Sciences, Mail Stop 4, One Shields Avenue, Davis, CA 95616; Research Botanist, Western Ecological Reseaarch Center, U.S. Geological Survey, Henderson, NV 89074; Cooperative Extension Specialist, University of California, Botany and Plant Science, 2129 Batchelor Hall, Riverside, CA 92521; VMP Coordinator, California Department of Forestry and Fire Protection, 6105 Airport Road, Redding, CA 96002; Research Associate, Division of Biological Sciences, University of Montana, Missoula, MT 59812; Staff Research Associate, University of California, Department of Plant Sciences, Mail Stop 4, One Shields Avenue, Davis, CA 95616. Corresponding author's E-mail: jmditomaso@ucdavis.edu.

[3] Letters following this symbol are a WSSA-approved computer code from *Composite List of Weeds*, Revised 1989. Available only on computer disk from WSSA, 810 East 10th Street, Lawrence, KS 66044-8897.

Table 1. Comparison of variables related to the use of fire to control invasive plants in croplands and wildlands.

	Croplands	Wildlands
Timing of fires	Preplant or postharvest	Varies with target species and ecosystem
Fuel types	Crop residual, with a simple fuel structure	Fine and coarse debris, with a complex fuel structure
Fire types	Surface fire	Surface and crown fire
Integrated treatments	Fire preceded by chemical or mechanical treatments, followed by a cover crop	Chemical and mechanical treatments pre- or postfire, and revegetation with competitive species
Type of invasives targeted	Typically herbaceous	Varies widely—grasses, herbs, shrubs, and trees
Ecological complexity	Low	High

with fire, it is critical to either kill plants before their seeds become viable (DiTomaso et al. 1999) or destroy the seeds before they disperse (Allen 1995; Menke 1992). Seeds of many species may be most susceptible to heating before they are fully cured (Brooks 2001). When grassland is burned, seeds on the soil surface generally are not exposed to lethal temperatures (Daubenmire 1968). Ideally, burning should be conducted when the seeds of target species are still in the canopy, and after the seeds of desirable species have dispersed to the ground. However, in some cases seeds retained on target species may be too far above the flames to be affected (e.g., prickly lettuce [*Lactuca serriola* L.] in grassland [M. Brooks, personal observation]). For herbaceous plants with protected meristems, for example, rosettes, the burn must be hot enough to damage these tissues. For perennial species, effective burns must prevent resprouting. With species that readily reproduce sexually, it is critical to deplete the soil seedbank and to prevent new seed production or recruitment. A single burn may temporarily reduce the population of the target plant, but most species tend to recover by the second or third year (e.g., Japanese brome [*Bromus japonicus* Murr.]) (Whisenant 1990; Whisenant and Uresk 1990). A follow-up program should be instituted to prevent escaped or isolated plants from completing their life cycle. Where the seedbank is short-lived, a follow-up program may take only a couple of years; in other cases it may take longer.

Annual grasses. Invasive grasses with long-awned seeds (e.g., medusahead [*Taeniatherum caput-medusae* (L.) Nevski], downy brome [*Bromus tectorum* L.], ripgut brome [*Bromus diandrus* Roth], red brome [*Bromus madritensis* L. ssp. *rubens* (L.) Husnot], and barb goatgrass [*Aegilops triuncialis* (L.)] rely on animal dispersal. In many of these species, the seeds remain in the inflorescence longer than most desirable grasses, where they are susceptible to death by the direct heat of burning. For example, medusahead matures at least a month later than most annual species (Dahl and Tisdale 1975; Young et al. 1970). Many studies have demonstrated good control of medusahead with a single early-summer burn (Furbush 1953; George 1992; McKell et al. 1962; Pollak and Kan 1996; Sharp et al. 1957). However, medusahead was poorly controlled by late-summer burns (Young et al. 1972; Youtie et al. 1998), probably because fire moved rapidly and with low intensity through the dry, late-season vegetation (Sweet 2005). Barb goatgrass has been controlled by early-summer burning in central California (Hopkinson et al. 1999) and was eradicated by two consecutive years of burning in a study in the California coastal foothills (DiTomaso et al. 2001). Japanese brome biomass was reduced 85% in the year after a March burn in South Dakota (Whisenant et al. 1984). Ripgut brome also was controlled with a spring fire (DiTomaso et al. 1999; Kyser and DiTomaso 2002). Ripgut brome matures earlier than medusahead, barb goatgrass, and Japanese brome, but its seeds appear to be very sensitive to heat (Sweet 2005). Downy brome and red brome are difficult to control with burning because their seedheads begin to shatter and the seeds fall to the soil surface before enough fuel is available (Brooks 2002; Young and Evans 1978). However, where fuel loads and soil heating are high, such as within woody shrubs, red brome biomass can be significantly reduced for 2 yr beneath the exterior canopy and 4 yr beneath the interior canopy (Brooks 2002).

Annual forbs. Prescribed burning is most effective on late-season annual forbs. Forbs that complete their life cycle in spring, such as winter annuals, are difficult to control with burning, because adequate fuel is unlikely to have accumulated. Fine fuel is typically not limited when burning is conducted in the summer to control later season plants, including summer annuals and some late-season winter annuals.

Yellow starthistle (*Centaurea solstitialis* L.) is an example of a late-season winter annual that can be controlled by certain burning regimes (DiTomaso et al. 1999; Hastings and DiTomaso 1996; Kyser and DiTomaso 2002). The most effective burn timing is in early summer, after senescence of desirable plant species but before viable seed production in yellow starthistle. Because its seeds can survive for more than 3 yr in the soil and their germination is enhanced by a preceding burn (J. M. DiTomaso and G. B. Kyser, unpublished data), a single year of burning will not control an infestation.

DiTomaso et al. (1999) found that three consecutive years of burning were required to reduce a yellow starthistle seedbank by 99%. In other studies (J. M. DiTomaso and G. B. Kyser, unpublished data; Miller 2003), integrating a first-year burn with a second-year herbicide treatment was the most effective strategy.

Biennials. Biennials are difficult to control with prescribed fire because they usually occur in mixed-aged stands; second-year plants that have bolted are susceptible (Heitlinger 1975), but 1-yr-old plants in the rosette stage have protected meristems. Burning later in spring may give better results provided the burn is conducted before bolted plants have set seed. More intense burns, such as areas with increased thatch, give better control of bolting plants (Heitlinger 1975). When a well-developed seedbank is present, additional burns may be necessary. Although burning can suppress biennial species, combining this strategy with herbicide treatments would likely be even more effective.

Multiple-year burns have been successful against biennial sweetclover (*Melilotus* spp.) in midwestern prairie (Cole 1991; Kline 1983; Schwarzmeier 1984). The first-year burn stimulated germination; the second-year burn was conducted after the plants had bolted but before seed production. Burning also has been used to control garlic mustard (*Alliaria petiolata* Andrz.) in the eastern United States. In areas where thatch and litter were damp, plants resprouted and seedlings survived; however, sequential burns under drier conditions were effective (Nuzzo et al. 1996).

Perennial grasses. In the prairie states, burning can be used to control cool-season nonnative perennial grasses, for example, Kentucky bluegrass (*Poa pratensis* L.), Canada bluegrass (*P. compressa* L.), and smooth brome (*Bromus inermis* Leysser), and to boost warm-season perennial grasses. Selective control depends on a heavy thatch layer and timing the burn so that tillers are elongating on the target species but the warm-season natives are still dormant, that is, mid- to late spring for perennial *Poa* spp. (Becker 1989; Curtis and Partch 1948: Engle and Bultsma 1984) or late spring to summer for smooth brome (Wendtland 1993; Willson 1992). Suppression of cool-season perennial grasses by burning can result in an increase in warm-season grasses (Robocker and Miller 1955). However, if warm-season grasses are not present, invasive grasses may re-infest the area (Schacht and Stubbendieck 1985; Willson and Stubbendieck 1996). Environmental characteristics that promote warm-season grasses can help suppress reinvasion, for example, in South Dakota high soil moisture promoted native grasses that suppressed smooth brome recovery after a burn (Blankespoor and Larson 1994). Some cool-season native species also may be susceptible to fire, for example, green needlegrass (*Stipa viridula* Trin.) (Engle and Bultsma 1984).

Tallgrass prairie may benefit from repeated burning (Smith and Knapp 1999, 2001). In Nebraska, 5 yr of burning reduced Kentucky bluegrass cover but increased cover of native grasses and forbs (Becker 1989). In Minnesota, 10 yr of biennial burns controlled Kentucky bluegrass and stimulated little bluestem [*Schizachyrium scoparium* (Michaux) Nash] and big bluestem (*Andropogon gerardii* Vitman) (Svedarsky et al. 1986). Biennial burning was less expensive than annual burns and left cover for ground-nesting birds between burn years. Burning in Kansas for 5 out of 6 yr (Abrams 1988) or 30 out of 36 yr (McMurphy and Anderson 1965) nearly eliminated Kentucky bluegrass. The latter treatment increased cover of warm-season natives, particularly little bluestem (Towne and Owensby 1984). Smooth brome is poorly controlled by a single burn and requires repeat burning at the tiller elongation stage (Willson and Stubbendieck 1996).

Perennial forbs. Because fires usually promote invasive perennial forbs, management of these plants using burning may require integration of other control options, particularly herbicide applications. Burn timing may be critical. For example, repeated spring burns in May to June suppressed Canada thistle [*Cirsium arvense* (L.) Scop.] in Illinois, but burns earlier in the spring or later in the summer stimulated sprouting and increased the infestation (Hutchison 1992; Morghan et al. 2000; Thompson and Shay 1989). Regardless of timing, prescribed burning has been unsuccessful against leafy spurge (*Euphorbia esula* L.), Dalmatian toadflax [*Linaria dalmatica* (L.) Miller [=*L. genistifolia* (L.) Miller]], and sulfur cinquefoil (*Potentilla recta* L.) (Jacobs and Sheley 2003a, 2003b; Lesica and Martin 2003; Wolters et al. 1994).

Prescribed burning may have some effect on diffuse knapweed (*Centaurea diffusa* Lam.). Although fire does not control the parent plant, the seed are retained in the flowerhead long into the season and are exposed to direct heat from the flames of the burn (Renney and Hughes 1969). In contrast, seeds of spotted knapweed (*Centaurea maculosa* Lam. [=*C. biebersteinii* DC., *C. stoebe* L.]) disperse soon after they mature, and neither spring nor fall burns control this species successfully (Emery and Gross 2005). Absinth wormwood (*Artemisia absinthium* L.) in South Dakota is susceptible to repeated spring burning (Steuter 1988), because its new buds develop close to the soil surface and can be killed by an intense fire. In the Southwest, given adequate fuel, pricklypear (*Opuntia* spp.) may be damaged by burning in rangeland. Mortality is usually low in the first year after burning but increases in subsequent years (Ueckert et al. 1988). However, fire is not often used for pricklypear control because the damage to desirable forage is considered unacceptable.

Woody species. Most woody species are difficult to control with burning. Japanese honeysuckle (*Lonicera japonica* Thunb.), tree-of-heaven [*Ailanthus altissima* (Miller) Swingle], Russian-olive (*Elaeagnus angustifolia* L.), and saltcedar (*Tamarix ramosissima* Ledeb.) are favored by fire because they resprout from the base. Sweetbriar rose (*Rosa eglanteria* L.), Himalaya blackberry (*Rubus armeniacus* Focke [=*R. discolor* Weihe and Nees]), cutleaf blackberry (*Rubus laciniata* Willd.), English hawthorn (*Crataegus monogyna* Jacq.), and common pear (*Pyrus communis* L.) also tend to increase after fire (Pendergrass et al. 1988). In eastern and mid-Atlantic states, prescribed burning for woody plant control is conducted during the dormant season, but it is generally unsuccessful because of resprouting. Burns in the growing season may be more effective because carbohydrate reserves are lowest at this time (Richburg and Patterson 2003). Vines are rarely controlled with prescribed burning.

In California, prescribed burning for the control of shrubs is most widely used on broom species, particularly French broom [*Genista monspessulana* (L.) L. Johnson] and Scotch broom [*Cytisus scoparius* (L.) Link]. Like most legumes, they have long-lived seedbanks. In addition, fire scarifies their seeds and stimulates germination in the following season. Consequently, successful management strategies must be long-term and should integrate methods that can deplete the seedbank (Swezy and Odion 1997). For example, in one study (Odion and Hausbensak 1997), French broom was cut in summer and burned in October, stimulating germination during the next rainy season. The cut and burn treatment was repeated to control seedlings. In a variation on this approach, French broom was cut in fall and the dried stems were burned the following May (Boyd 1996). The burn was hot enough to kill the mature rootstalks. The fire also stimulated seed germination the next fall. In November after the burn, annual grasses were broadcast seeded on the burn site, providing fuel for a July burn, which killed the broom seedlings. These sites would likely require several more years of follow-up control to eliminate French broom.

Fire also can be used to control certain tree species, for example, European privet (*Ligustrum vulgare* L.) and Chinese privet (*L. sinense* Lour.) in Alabama (Batcher 2000) and Chinese tallow tree [*Sapium sebiferum* (L.) Roxb.] in southern coastal prairie (Grace 1998). In rangeland, burning may be used to suppress native woody species that are encroaching after long-term fire suppression (Miller and Tausch 2001), including big sagebrush (*Artemisia tridentata* Nutt.), false broomweed (*Ericameria austrotexana* M. C. Johnston), broom snakeweeds (*Gutierrezia* spp.) (Mayeux and Hamilton 1988; McDaniel et al. 1997), junipers (*Juniperus* spp.) (Mitchell et al. 2000), and mesquite (*Prosopis* spp.) (DiTomaso 2000). Mesquite is susceptible to burning when plants are young ($<$4 yr old), but older plants are much more difficult to kill.

USING PRESCRIBED BURNING IN INTEGRATED STRATEGIES

Although repeated burning can be effective for the control of several invasive plant species, this is often prohibited or impractical. Even when possible, such an approach may not be the most appropriate strategy. In the case of tallgrass prairie, repeated burning suppressed exotic weeds and stimulated native warm-season grasses (Smith and Knapp 1999, 2001). However, most ecosystems did not evolve with such a short fire frequency interval. The creation of unnatural fire regimes can favor nonnative species (Brooks et al. 2004; Cowling 1987) and can impact other ecosystem properties such as soil characteristics and nutrient cycling.

Because few invasive weeds are effectively managed by a single year of prescribed burning, it is often necessary to incorporate other control methods into a long-term management strategy (Kyser and DiTomaso 2002). These methods can include mechanical, cultural, biological, and chemical options. Burning may enhance the effectiveness of other techniques, particularly with invasive perennials such as brooms, gorse (*Ulex europaea* L.), saltcedar, leafy spurge, giant reed (*Arundo donax* L.), catclaw mimosa (*Mimosa pigra* L.), and perennial pepperweed (*Lepidium latifolium* L.).

It is not always possible to restore a community to a pristine condition. For example, grasses from Central America and Africa dominate seasonally dry habitats in Hawaii, leading to shorter fire intervals and regional loss of native forest (D'Antonio et al. 2000). Thus, restoration programs are creating new native plant communities that are fire tolerant and can coexist with native grasses (Tunison et al. 2001). In other situations, nonnative desirable species can be introduced to outcompete more detrimental nonnative species. For example, crested wheatgrass [*Agropyron desertorum* (Fischer) Schultes], a nonnative perennial grass, was seeded into a postfire rangeland in the Great Basin desert to suppress cheatgrass and reduce fuel continuity and flammability (Hull and Stewart 1948).

In many areas, prescribed burning is not permitted at all. However, wildfires (uncontrolled burns) do occur and may provide an opportunity to control invasive species. To take advantage of such a situation, other control options can be used after the wildfire. For example, after an August wildfire in a site in Utah infested with squarrose knapweed [*Centaurea triumfettii* All. [=*C. squarrosa* Willd. or *C. virgata* Lam. var. *squarrosa* (Willd.) Boiss.]], the site was treated with picloram plus 2,4-D the following fall. Nearly 3 yr after the herbicide treatment, squarrose knapweed control was 98 to 100%, compared to 7 to 20% in an adjacent unburned site (Dewey et al. 2000).

Situations Conducive to Integrated Approaches. *Burning followed by herbicide treatment.* An initial burn in a management program can stimulate seed germination, depleting the seedbank and making seedlings available for control. This has been shown with yellow starthistle (summer burn followed by clopyralid in winter) (J. M. DiTomaso et al., unpublished data) and Lehmann lovegrass (*Eragrostis lehmanniana* Nees) (Biedenbender et al. 1995). The germination peak in these species may be due to removal of the litter layer (DiTomaso et al. 1999). Burning also stimulates germination of brooms, gorse, and other legumes, probably by scarification (Boyd 1996). Other evidence indicates that smoke from burning plant materials can contain a water-soluble butenolide that acts as a germination stimulant in some plant species (Brown and van Staden 1997; Flematti et al. 2004).

Prescribed burning can be used to improve access to an area with a high density of an invasive species such as saltcedar or other *Tamarix* species. In saltcedar, cutting followed by burning can remove the biomass and provide access for a secondary chemical or mechanical treatment (Friederici 1995; Taylor and McDaniel 1998). Removal of litter by burning or other means can improve visibility for follow-up control, as with rosettes of wild parsnip (*Pastinaca sativa* L.) (Eckardt 1987), and can improve application of preemergence herbicides to the ground (Winter 1993). Litter can tie up a preemergence herbicide and reduce its activity. After burning, improved activity has been observed for imazapic on medusahead and downy brome (J. M. DiTomaso et al., unpublished data; Washburn et al. 1999) and on tall fescue (Rhoades et al. 2002; Washburn et al. 2002).

Burning perennials can improve the effect of foliar herbicides by removing biomass, thatch, and older plant tissues. The recovering vegetation is more exposed and more succulent with a less-developed cuticle, resulting in improved foliar deposition and better uptake. Examples include fennel (*Foeniculum vulgare* Miller) on Santa Cruz Island, California, where burning was followed with triclopyr (Klinger and Brenton 2000); tall fescue burned in spring and followed with glyphosate or imazapic (Washburn et al. 1999); and Japanese honeysuckle and kudzu [*Pueraria montana* (Lour.) Merr. var. *lobata* (Willd.) Maesen and S. Almeida] burned in winter or early spring before herbicide treatments (Brender 1961; Shipman 1962).

Herbicide treatment followed by burning. An herbicide pretreatment may be used to enhance the fuel load to carry a burn (Glass 1991). Treated plants may themselves become fuel; for example, catclaw mimosa (Paynter and Flanagan 2004), gorse (Rolston and Talbot 1980), and mesquite (Queensland Government 2004) were better controlled by burning after pretreatment. Pretreatment also may produce fuel indirectly. Treatment was used to suppress a dense population of yellow starthistle, thus increasing grasses and facilitating a complete burn in the second year (J. M. DiTomaso et al., unpublished data). Chemical or mechanical killing of large Chinese tallow trees in southern prairie allowed development of an understory that served as fuel to burn out small trees (Grace 1998). In North Dakota, fall application of picloram followed by a spring burn gave better control of leafy spurge than either treatment alone (Wolters et al. 1994).

Prescribed fire can be used after mechanical or chemical methods to remove dead biomass and stimulate recovery of the infested site, for example, in saltcedar or giant reed (Bell 1997). In some cases, prescribed burning can be used in combination with other techniques or between two herbicide treatments. Common reed [*Phragmites australis* (Cav.) Steudel] was controlled when plants were herbicide treated, burned, and the resprouting stems treated again (Clark 1998). French broom cover was reduced from 87% to less than 1% when plants were treated with triclopyr, cut and burned a month later, then treated again with glyphosate for 2 yr to control seedlings (Bossard 2000).

Multiple species complexes. Prescribed burning used to control one undesirable species can sometimes select for another, for example, unpalatable native species such as tarweeds (*Hemizonia* spp.). As an example, repeated burning proved effective for management of yellow starthistle (DiTomaso et al. 1999) and medusahead (DiTomaso et al. 2005) but increased the population of nonnative filarees (*Erodium* spp.), which reduce annual grass forage in rangelands. Using integrated approaches, it is possible to select for a more desirable community complex. Integration with herbicide control can help prevent a single species from dominating burned areas.

Burning to decrease dependence on herbicides. Herbicides are the most widely used method of weed control in crops and in many noncrop areas. However, herbicides often do not provide long-term weed control when used alone (Bussan and Dyer 1999). Continuous use can create environmental problems, including off-site chemical movement, selection for tolerant or resistant species, injury to desirable plants and reduction in plant diversity, and changes in the nutrient balance. Integrated management can reduce dependence on herbicides by increasing herbicide efficacy or reducing the number of applications. For example, yellow starthistle control typically requires 3 yr of prescribed burning or clopyralid treatment when either method is used alone. However, the thistle can be controlled in 2 yr by a prescribed burn in summer of the first year followed by application of clopyralid in the following winter.

Burning to prepare for revegetation programs. Removing litter and suppressing invasive species can facilitate establishment of native species. Thatch removal increases solar heating of the soil, which promotes early growth of perennial grasses and legumes (DiTomaso et al. 1999; Ehrenreich 1959). In Nebraska, spring burning to remove the litter layer was followed by herbicide treatment of leafy spurge and by drill seeding of prairie grasses (Masters and Nissen 1998; Masters et al. 1996). In grasslands, burning invasive annual grasses can reduce early-season competition and allow establishment of reseeded desirable species (Goodrich and Rooks 1999). In saltcedar-infested riparian areas, heavy litter accumulation prohibited establishment of desirable plants even after saltcedar was controlled. Integrating a burn treatment can open the soil surface and promote the recovery of natives (Taylor and McDaniel 1998). Revegetation after burning also can help to control soil erosion.

Burning to enhance the efficacy of biocontrol agents. Burning in spring or summer can kill biological control agents whose larvae feed in the seedheads, for example, agents released for yellow starthistle and other knapweed species. However, establishment of the leafy spurge flea beetle (*Aphthona nigriscutis* Foudras) was 230% more successful in burned plots than unburned plots (Fellows and Newton 1999). During the burns, in either mid-May or mid-October, the adult insects were inactive and the juveniles were below ground. The enhanced establishment was attributed to increased colonization in the bare ground of the burn plots. A population of the Klamath weed (St. Johnswort) beetle (*Chrysolina quadrigemina* Suffrian) declined after burns (Briese 1996) but rebounded quickly through offsite recruitment. The increased nitrogen (N) taken up by St. Johnswort in the burn site also benefited the insect populations. Recovery of biocontrol agents also has been observed after burns to control yellow starthistle (M. J. Pitcairn, unpublished data).

Avoiding environmental problems associated with multiple burns. Disturbance regimes affect ecosystem functions such as soil erosion and formation, nutrient cycling, and energy flow (Cowling 1987). Continued disturbance such as repeated burning can result in increased bare ground and risk of erosion, may improve conditions for future plant invasions (Brooks et al. 2004), and can have a direct impact on populations of desirable plants and animals (DiTomaso 1997). Burns will select for plant species that complete their life cycle before the burn. Burning also can enable invasion by species with wind-dispersed seeds, particularly members of the Asteraceae. Integrated control methods may minimize these potential problems.

Manipulating fire characteristics. Fire intensity can impact the level of control and the ability to contain the burn. Burn intensity can be increased by augmenting the fuel load (e.g., by restricting grazing before the burn [George 1992]) or by slowing the fire front (e.g., using backing fires instead of head fires). Intensity can be decreased by decreasing the fuel load (e.g., grazing before a burn), reducing the size of the burn parcel, or speeding up the fire front, for example, by burning in late afternoon or later in the season to ensure the fuel is as dry as possible (McKell et al. 1962).

In low-productivity ecosystems it may take several years to accumulate sufficient fine fuel to carry a fire (DiTomaso et al. 2001). This can be a problem in a repeated burn program, when a first-year burn eliminates litter, leaving insufficient fuel for a second-year burn (Young et al. 1972). With large perennial or woody species, such as giant reed, saltcedar, gorse, and brooms,

Table 2. Effects of fire on Raunkiaer (1934) plant life forms (modified from D. Pyke, M. L. Brooks, and C. M. D'Antonio, unpublished data).

Raunkiaer life form	Example	Perennating or reproductive tissue	Exposure of perennating or reproductive tissue to damage from fire
Therophytes	Annuals	Seeds on or under the soil surface, or on senesced plants	Depends on where seeds are located during fire
Cryptophytes	Bulbs or corms	Perennial tissue well below the soil surface	Protected from fire due to soil insulation above them
Hemicryptophytes	Rhizomatous	Perennial tissue just above or below the soil surface	Depends on the percentage of litter burned and the amount of smoldering combustion
Chamaephytes	Shrubs	Perennial tissue just above the soil surface	Often killed by fire due to their positioning directly in the flame zone of surface fires
Phanerophytes	Trees	Perennial tissue well above the soil surface	Can be killed by crown fire or by surface fire that girdles the trees

mechanical or chemical treatments a few months before the burn can augment the dried biomass, increasing the intensity of the burn and providing better control of resprouting.

EFFECTS ON PLANT COMMUNITIES

Invasive plants are typically managed to achieve two primary goals: to reduce the dominance of the target invasive species, and to increase the dominance and diversity of more desirable species, particularly native plants. This requires a plan to manage the invasive species at the population level and the rest of the plant species at the community level. Information on the use of fire to manage invasive plants tends to focus on the immediate effects of fire on the target invasive species. Few studies have evaluated higher-order effects of these treatments on plant communities, soils, wildlife, or ecosystems in general.

Because invasive plants generally thrive in disturbed environments, they often dominate postfire landscapes unless the native species are also fire-adapted. In some cases, revegetation with native species may be necessary. If the goal of treatments is to produce a self-sustaining fire regime, then the resultant plant community must create appropriate fuels. For example, where woody species have invaded herbaceous communities because of past fire suppression, it may be effective to restore a low-intensity frequent-fire regime fueled by native herbaceous plants. This is only possible if herbaceous fuels reaccumulate rapidly after they burn.

Characteristics of fires. Fires vary in timing, continuity, and intensity. Timing can be important if the target species has a vulnerable window during its development. Fire continuity, that is, whether it is a complete burn or a patchy burn, depends on continuity of the fuelbed. Most significantly, the intensity of the flaming front—rate of spread, residency time, depth, and height—can vary greatly within and among fires. The duration of smoldering after the front has passed can also influence soil heating, which can affect soil properties, roots, and soil seedbanks. The duration of smoldering, and the degree of soil heating, is highest in heavy woody fuels and lowest in light herbaceous fuels (e.g., Brooks 2002). The effects of a single fire are distinct from the effects of a fire regime (a repeated pattern of burning over time). Like single fires, fire regimes vary in type, frequency, intensity, extent and spatial pattern, and seasonality (Brooks et al. 2004; Heinselman 1981; Keeley 1977; Kilgore 1981; Sando 1978).

Effects of fire on individual plants. Fires can damage plant tissue directly when consumed by flames and can disrupt physiological processes indirectly when radiant or convective heating reach high levels (Levitt 1972). Plant tissue that is metabolically inactive or dehydrated can withstand greater heating than active tissue (Whelan 1995). Thus, burning during a plant's active growing season often results in the highest mortality rates. The effect of fire on individual plants also depends on the degree to which perennating tissues are protected from lethal temperatures (Whelan 1995). Raunkiaer (1934) developed a system for classifying plants based on the location of the perennating tissue. D. Pyke, M. L. Brooks, and C. M. D'Antonio (unpublished data) used this classification system to predict responses of different plant species to fire (Table 2).

Effect of burning on nontarget species and plant communities. Burn timing can influence populations of nontarget species. Species that complete their life cycle before burns are conducted will generally be selected for, whereas those that flower and seed later will be negatively impacted. Late-spring or early-summer burns generally have the greatest impact on invasive annuals and are the most beneficial for native forb species (Meyer and Schiffman 1999). Burns in late summer or fall after grasses have senesced can favor native perennials but may not control the targeted annual species (Dyer and Rice 1997). Winter burns, after exotic grasses have emerged, can reduce litter but may not increase native forbs (Meyer and Schiffman 1999).

Most studies show that few nontarget plants respond negatively to summer burning. For example, in Sonoma County, a prescribed burn program for the control of yellow starthistle also reduced the nonnative annual grasses false brome [*Brachypodium distachyon* (L.) Beauv.], ripgut brome, and soft brome (*Bromus hordeaceus* L.). Over 3 yr of burning, only 8% of native plant species showed a decline (DiTomaso et al. 1999; Hastings and DiTomaso 1996).

In some cases invasive plants can increase after fire, for example, Malta starthistle (*Centaurea melitensis* L.) in Sequoia National Park, California (Parsons and DeBenedetti 1984; Parsons and Stohlgren 1989). Most annual grasses experience at most transient effects from prescribed burning. For example, in Sonoma County, wild oat (*Avena fatua* L.), silver hairgrass (*Aira caryophyllea* L.), and little quakinggrass (*Briza minor* L.) increased after a 3-yr burn regime, but returned to preburn levels within 2 yr of the last burn (Kyser and DiTomaso 2002).

Late-spring or early-summer burns tend to favor forbs. For example, three consecutive years of burning increased native forb cover by nearly 400% in Sonoma County (DiTomaso et al. 1999). A number of species can benefit from fire, particularly members of the Fabaceae or Geraniaceae; for example, the native legumes *Lotus wrangelianus* Fischer and C. Meyer, *Lupinus nanus* Benth., and *Trifolium gracilentum* Torrey and A. Gray increased after burning for yellow starthistle control (DiTomaso et al. 1999); and *T. bifidum* A. Gray, *Astragalus gambelianus* E. Sheldon, and *Lotus humistratus* E. Greene increased after control burns for barb goatgrass (DiTomaso et al. 2001). Some nonnative forbs increase after warm-season burns, for example, *Erodium* spp. (Kyser and DiTomaso 2002; Murphy and Lusk 1961). Burning also may increase native perennial grasses such as *Hordeum brachyantherum* Nevski (DiTomaso et al. 2001), western wheatgrass (*Agropyron smithii* Rydb.) (Gartner 1975), and purple needlegrass [*Nassella pulchra* (A. Hitchc.) Barkworth] (DiTomaso et al. 1999; Fossum 1990; Hatch et al. 1991).

In general, prescribed burns increase plant diversity and species richness, usually because of an increase in forbs rather than target species suppression. For example, a single burn in Sonoma County increased plant diversity but did not reduce summer yellow starthistle cover (DiTomaso et al. 1999). Native grassland plants may benefit from removal of the thatch layer by burning (Knapp and Seastedt 1986); this may increase light penetration, soil temperature, and nutrient availability, and discourage survival of pathogens.

IMPACTS ON SOIL CHEMICAL, PHYSICAL, AND BIOTIC PROPERTIES

Although fires can have potential negative effects on soil chemical, physical, and biological properties, these impacts can be minimized by careful burning and site selection. Erosion poses the greatest risk, but this risk can be reduced by burning only small areas, or avoiding burning altogether, in steep terrain (DeBano et al. 1998). Only the hottest fires, as under slash piles or built-up litter, will cause long-term changes in soil properties (Korb et al. 2004; Neary et al. 1999). On the other hand, invasive species also may alter soil properties (Ehrenfeld 2003). Although studies are not available at this time, we suggest that fire can have beneficial effects on soils and may help to restore the negative effects that invasive plants may have caused, especially where invasive species have increased the litter layer and increased soil N.

Effects of Fire on Soil Chemical and Physical Properties. *Fire temperature.* High-temperature fires can have negative effects on surface as well as deeper soils (Korb et al. 2004; Neary et al. 1999), whereas low- and moderate-temperature fires are generally beneficial in fire-adapted ecosystems (DeBano et al. 1998). High temperature fires (700 C or greater) may occur in forest fires, high-productivity shrublands, and in situations where fire suppression has increased the fuel load. Low ground fires in forests and grass fires burn at 200 to 300 C (DiTomaso et al. 1999; Rundel 1983). Fires in desert shrublands can range from approximately 200 C beneath large woody shrubs to 50 C (near ambient) within the interspaces between them (Brooks 2002). Organic matter (OM) consumption by fire begins at 180 C, and all of the soil OM is consumed when the soil is heated to 450 C (DeBano et al. 1998).

A smoldering, slow-moving fire causes more heat damage than a fast-moving fire, and moist soil will conduct more heat than dry soil. Where fuels are patchy, soil heating is spatially patchy as well. For example, soil heating is highest under woody shrubs where the duration of flaming and smoldering combustion is lower than in fine fuels (Brooks 2002). A soil with a low fuel load and a low severity of heating will attain temperatures of only 100 C at the surface and 50 C at 5 cm depth. When the soil surface reaches 700 C, it may be 100 C as deep as 22 cm after a slow-moving fire (Neary et al. 1999). Soil chemical characteristics are little affected by temperatures less than 100 C, thus the impact of a burn may be reduced by managing for a fast-moving fire when the soil is dry.

Nitrogen and organic matter. The impact of fire on soils was extensively reviewed by Neary et al. (1999) and DeBano et al. (1998). In brief, organic material is almost always lost in fires, because carbon volatilizes at 180 C. In 15 semiarid to submesic ecosystems, 9 lost OM after fire, 4 did not change, and 1 low-temperature fire increased OM (E. Allen, unpublished review). Nitrogen begins to volatilize at 200 C. More than one-half of soil N can be lost when the temperature reaches 500 C. However, in cool, dry ecosystems where decomposition is slow, fire is an important agent of N mineralization; in such systems N gains often balance losses. In 20 submesic to semiarid ecosystems, more than one-half of sites reporting N showed a decrease in total soil N after fire, whereas the others showed no change. Decreases generally were caused by erosion.

Other nutrients and pH. Because potassium (K) and phosphorus (P) require temperatures higher than 700 C for volatilization, their loss is usually minimal unless the fire is followed by erosion. Other nutrients such as calcium (Ca), magnesium (Mg), and sodium (Na) volatilize at much higher temperatures. Extractable P was higher after fire in one-half of the studies that reported P, and decreased or showed no change in the others. Of 11 studies reporting pH, 9 had increased pH and 2 showed no change after fire. Elevated pH is typical after fire (Neary et al. 1999), because ash consists primarily of cations (e.g., Ca, Mg, Mn, K, and Na). Dry plant tissue may contain 10% cations, and its ash may have a pH of 9 to 11. After a fire some of the ash may be lost by erosion, causing future nutrient depletion, whereas some ions leach into the soil and increase its pH. The pH drops again as plants grow and take up cations.

Physical properties. Several studies reported either increased or unchanged bulk density after fire. The increase in bulk density is largely caused by the combustion of surface roots and decomposition of roots when shoots are fire-killed (Neary et al. 1999). With the loss of macropores, infiltration is reduced, soil becomes drier, and vegetation recovery may be slowed, especially in semiarid areas (Snyman 2002, 2003). Lower soil moisture also may result from higher temperatures in bare soil and to loss of litter that would normally slow runoff and increase infiltration. In clay soils, crusting after fire may impede infiltration (Mills and Fey 2004). Hydrophobicity, or water repellency, is caused when bare soil surfaces seal under the impact of raindrops, resulting in increased runoff (Ballard 2000; Doerr et al. 1998; Neary et al. 1999). In addition, certain litter types leave hydrophobic organic compounds on the soil surface, although these compounds are destroyed in hot fires where soil temperatures are higher than 290 C (DeBano et al. 1976; Neary et al. 1999). Hydrophobicity contributes to erosion and loss of nutrient-rich ash and topsoil, and is another long-term impact of fire.

Erosion is the most devastating effect of crown fires. Woody vegetation recovers slowly and erosion continues for 3 yr or more after a burn, even with efforts to slow erosion by seeding nonnative grasses (Beyers 2004).

Long-term resilience. Fire-adapted vegetation under normal fire regimes usually recovers rapidly from fire (DeBano et al. 1998). Ecosystems with a high proportion of biomass and nutrients below ground, such as tallgrass prairie, are more buffered from fire impacts than are those with a smaller proportion below ground (Neary et al. 1999). In South African grassland and savanna, annual burns over more than 30 yr resulted in no change in soil OM and a minimal decrease in total soil N (Mills and Fey 2004). In chaparral, most reports found increases or no change in mineral N (decreases occurred primarily through erosion), and others found decreases or no change in total N (DeBano et al. 1998; Neary et al. 1999). Chaparral communities tend to recover quickly because early successional species often include N-fixers such as *Lotus scoparius* (Nutt.) Ottley and *Ceanothus* spp.; a large proportion of their biomass is below ground; and at a 2.5-cm depth in the soil profile, temperatures drop below 200 C under a fire, thus reducing the loss of soil N (DeBano et al. 1977). However, chaparral in slash piles or other high-fuel settings recovers poorly because of the effect of temperature on soil characteristics (Korb et al. 2004), the seedbank (Haskins and Gehring 2004), and erosion.

Effects of fire on soil microbiological properties. Nutrient availability to plants is regulated by soil microorganisms, so their survival and recovery are essential to restoring burned sites. Soil temperatures higher than 100 C kill most microorganisms, at least in moist soils. Soil is a good insulator, and the temperature at a 2.5-cm depth may be only 50 C when the surface is 100 C. However, when the soil surface reaches 700 C, the soil may reach

100 C down to 22 cm (Neary et al. 1999) and kill microorganisms at that depth.

After fire, soil microbial activity (saprotrophic bacteria and fungi, and mycorrhizal fungi) increases as often as it decreases (Acea and Caballas 1996; Anderson et al. 2004; Badia and Marti 2003a, 2003b; Bauhus et al. 1993; Fonturbel et al. 1995; Garcia-Oliva et al. 1998; Mabuhay and Nakagoshi 2003). Saprotrophic activity after a fire depends upon how much soil OM remains.

Nitrification generally increases after a fire because there is an accumulation of NH_4^+ mineralized by the fire that is then converted to NO_3^- (Anderson et al. 2004; Bauhus et al. 1993; White and Zak 2004). However, after several months, microbial activity often drops below the immediate postfire level, and may not recover entirely until soil organic matter recovers to preburn levels (White and Zak 2004). A reduction in nitrification does not necessarily limit plant productivity, because most plants can take up NH_4^+ directly. In fact, most studies of wildland fires at ‘‘normal’’ fuel loads show increased productivity of vegetation for one or more years postfire (e.g., Carreira and Niell 1992; Seastedt and Ramundo 1990).

Mycorrhizal inoculum is reduced after high-intensity burns, for example, under a slash pile (Korb et al. 2004); in fire-suppressed lodgepole pine forest in Yellowstone (Miller et al. 1998); or in accumulated litter under trees in pinyon–juniper woodland (Klopatek et al. 1994). However, other studies showed little or no reduction in mycorrhizal infection of plants after fire (Allen et al. 2003; Anderson and Menges 1997; Korb et al. 2003; Rashid et al. 1997). A slash-pile fire in pinon–juniper woodland recovered mycorrhizal inoculum after 5 yr, but was dominated by exotic forbs (Haskins and Gehring 2004). After eucalyptus fires in Australia, rates of infection appeared dependent on soil type (Launonen et al. 1999). The fungal species composition can be changed by fire (Baar et al. 1999; Stendell et al. 1999); recovery may take years (Allen et al. 2003) and microbial species composition may undergo a fire-induced succession.

Impacts of exotic species on soils. Invasive plants can have impacts on soil chemical and microbial properties that are as great or greater than those of fires, and in fact fire may be used to reverse some of these impacts. The following is summarized from Ehrenfeld’s (2003) review of impacts of invasive plants on soils and nutrient cycling. Sixteen of 20 sites reported greater biomass after invasion, especially where woody species or fast-growing grasses replaced native grasslands. Decreases in stand biomass resulted when annual grasses replaced native shrublands. Net primary productivity and growth rate tended to increase in invaded stands. However, litter mass and soil carbon were equally likely to increase or decrease after invasion; thus, elevated biomass was likely offset by an increased decomposition rate. Most sites also reported increased mineralization and increased microbial carbon.

Plant invasions tend to result in increased total N. Invasive N-fixing leguminous or actinorhizal shrubs can cause permanent alterations in nutrient cycling. Alternatively, systems that are invaded by a single species of flammable annual grass, such as ripgut brome, lose soil N over time because of frequent fires and erosion (Evans et al. 2001). Invasive species may preferentially colonize nutrient-rich soils (Bashkin et al. 2003) but also may promote mineralization and higher extractable nutrients (Evans et al. 2001).

Invasive plants may affect the species composition of soil microorganisms. For example, foxtail brome (*Bromus madritensis* L. ssp. *madritensis*) associated with a different mycorrhizal endophyte than did the native shrubs it replaced (Sigüenza et al. 2006). In another study, a shift in arbuscular mycorrhizal composition caused by wild oat (*Avena barbata* Link) was reversed after native grasses were replanted (Nelson and Allen 1993).

SUMMARY

This review provides a general overview of how fire can be used to manage invasive plant species. In general, annual species that produce seeds well after the fire season begins, that have flowering structures embedded within the fuelbed, and that have short-lived seedbanks are most amenable to control using fire. In this example, the current cohort can be killed by fire before their seeds have matured or dispersed to the ground, and follow-up treatments are only necessary for a few years until the seedbank is depleted. In contrast, perennial species with perennating tissue that is either below ground or well above the fuelbed (and thus protected from heating), and that resprouts readily because they are adapted to fire or some other form of recurrent disturbance, are not generally amenable to control by fire. Invasive plants that alter the fuelbed structure making it less flammable also may be difficult to control with fire because they produce fuelbeds that are relatively inflammable.

In all cases, follow-up monitoring and plans for retreatment are required. For maximum effectiveness, in most cases, fire should be integrated with other control methods. The ultimate net effects of any treatment plan on the entire plant community, higher tropic levels, and ecosystem properties need to be considered before a treatment plan is implemented. Fuel loads created by invasive weeds will rarely be sufficient to affect soil chemical and physical properties adversely, insofar as fire-induced erosion can be avoided. It is always possible that the results of intensive land management may be worse than the effects of inaction. For this reason, it is important to understand the range of affects fire and other management treatments can have on ecosystems.

ACKNOWLEDGMENTS

This report is based in part on discussions held at a workshop March 29–30, 2004, hosted by the U.S. Geological Survey in Henderson, NV. We thank the following individuals for their participation in the workshop and contribution report and manuscript: James Bartolome, David Boyd, Mike Connor, Steve Enloe, James Grace, Marla Hastings, Diana Kimberling, Eric Lane, Kurt McDaniel, Laura Moser, Mike Pellant, Tim Prather, Steve Radosevich, Lisa Rew, Brianna Richardson, Maria Ryan, Mandy Tu, Rob Wilson, Jim Young, and Kris Zouhar. We also acknowledge contributions by Doug Johnson, Elizabeth Brusati, and Gina Skurka of the California Invasive Plant Council, Berkeley, CA. We thank the Center for Invasive Plant Management, Bozeman, MT, for their financial support of this project, including the workshop. We also thank the Renewable Resources Extension Act for financial assistance.

LITERATURE CITED

Abrams, M. D. 1988. Effects of burning regime on buried seed banks and canopy coverage in a Kansas tallgrass prairie. Southwest. Nat. 33:65–70.

Acea, M. J. and T. Carballas. 1996. Changes in physiological groups of microorganisms in soil following wildfire. FEMS Microbiol. Ecol. 20:33–39.

Allen, E. B. 1995. Restoration ecology: limits and possibilities in arid and semiarid lands. *In* B. A. Roundy, E. D. McArthur, J. S. Haley, and D. K. Man, eds. Proceedings, Wildland Shrub and Arid Land Restoration Symposium, Las Vegas, NV. Ogden, UT: U.S. Department of Agriculture, Forest Service, General Technical Rep. INT-GTR-315. Pp. 7–15.

Allen, E. B., M. F. Allen, L. Egerton-Warburton, L. Corkidi, and A. Gomez-Pompa. 2003. Impacts of early- and late-seral mycorrhizae during restoration in seasonal tropical forest, Mexico. Ecol. Appl. 13:1701–1717.

Anderson, M., A. Michelsen, M. Jensen, and A. Kjoller. 2004. Tropical savannah woodland: effects of experimental fire on soil microorganisms and soil emissions of carbon dioxide. Soil Biol. Biochem. 36:849–858.

Anderson, R. C. and E. S. Menges. 1997. Effects of fire on sandhill herbs: nutrients, mycorrhizae, and biomass allocation. Am. J. Bot. 84:938–948.

Baar, J., T. R. Horton, A. M. Kretzer, and T. D. Bruns. 1999. Mycorrhizal colonization of *Pinus muricata* from resistant propagules after a stand-replacing wildfire. New Phytol. 143:409–418.

Badia, D. and C. Marti. 2003a. Plant ash and heat intensity effects on chemical and physical properties of two contrasting soils. Arid Land Res. Manag. 17:23–41.

Badia, D. and C. Marti. 2003b. Effect of simulated fire on organic matter and selected microbiological properties of two contrasting soils. Arid Land Res. Manag. 17:55–69.

Ballard, T. M. 2000. Impacts of forest management on northern forest soils. For. Ecol. Manag. 133:37–42.

Bashkin, M., T. J. Stohlgren, Y. Otsuki, M. Lee, P. Evangelista, and J. Belnap. 2003. Soil characteristics and plant exotic species invasion in the Grand Staircase—Escalante National Monument, Utah, USA. Appl. Soil Ecol. 22:67–77.

Batcher, M. S. 2000. Element stewardship abstract for *Ligustrum* spp., privet. http://tncweeds.ucdavis.edu/esadocs/documnts/ligu_sp.html. Accessed: June 1, 2005.

Bauhus, J., P. K. Khanna, and R. J. Raison. 1993. The effect of fire on carbon and nitrogen mineralization and nitrification in an Australian forest soil. Austr. J. Soil Res. 31:621–639.

Becker, D. A. 1989. Five years of annual prairie burns. *In* T. B. Bragg and J. Stubbendieck, eds. Proceedings, 11th North American Prairie Conference—Prairie Pioneers: Ecology, History and Culture. Lincoln, NE: University of Nebraska. Pp. 163–168.

Bell, G. P. 1997. Ecology and management of *Arundo donax*, and approaches to riparian habitat restoration in southern California. *In* J. H. Brock, M. Wade, P. Pysek, and D. Green, eds. Plant Invasions: Studies from North America and Europe. Leiden, The Netherlands: Backhuys. Pp. 103–113.

Beyers, J. L. 2004. Postfire seeding for erosion control: effectiveness and impacts on native plant communities. Conserv. Biol. 18:947–956.

Biedenbender, S. H., B. A. Roundy, and L. Abbott. 1995. Replacing Lehmann lovegrass with native grasses. *In* B. A. Roundy, E. D. McArthur, J. S. Haley, and D. K. Mann, eds. Proceedings, Wildland Shrub and Arid Land Restoration Symposium, Las Vegas, NV. Ogden, UT: U.S. Department of Agriculture, Forest Service, General Technical Rep. GTR-INT-315. Pp. 52–56.

Blankespoor, G. W. and E. A. Larson. 1994. Response of smooth brome (*Bromus inermis* Leyss.) to burning under varying soil moisture conditions. Am. Midl. Nat. 131:266–272.

Bossard, C. C. 2000. *Genista monspessulana*. *In* C. C. Bossard, R. M. Randall, and M. C. Hoshovsky, eds. Invasive Plants of California's Wildlands. Berkeley, CA: University of California. Pp. 203–208.

Boyd, D. 1996. Use of fire to control French broom. Proc. Calif. Exotic Pest Plant Counc. Symp. 2:9–12.

Brender, E. V. 1961. Control of Honeysuckle and Kudzu. Asheville, NC: U.S. Department of Agriculture, Forestry Service, Southeastern Forestry Experimental Station. Statistics Paper 120. 9 p.

Briese, D. T. 1996. Biological control of weeds and fire management in protected natural areas: are they compatible strategies? Biol. Conserv. 77: 135–141.

Brooks, M. 2001. Fire and invasive plants in the wildlands of California. Nox. Times 3(4):4–5.

Brooks, M. L. 2002. Peak fire temperatures and effects on annual plants in the Mojave Desert. Ecol. Appl. 12:1088–1102.

Brooks, M. L., C. M. D'Antonio, D. M. Richardson, J. B. Grace, J. E. Keeley, J. M. DiTomaso, R. J. Hobbs, M. Pellant, and D. Pyke. 2004. Effects of invasive alien plants on fire regimes. BioScience 54:677–688.

Brown, N.A.C. and J. van Staden. 1997. Smoke as a germination cue: a review. Plant Growth Regul. 22:115–124.

Bussan, A. J. and W. E. Dyer. 1999. Herbicides and rangeland. *In* R. L. Sheley and J. K. Petroff, eds. Biology and Management of Noxious Rangeland Weeds. Corvallis, OR: Oregon State University. Pp. 116–132.

Carreira, J. A. and F. X. Niell. 1992. Plant nutrient changes in a semiarid Mediterranean shrubland after fire. J. Veg. Sci. 3:457–466.

Clark, K. H. 1998. Use of prescribed fire to supplement control of an invasive plant, *Phragmites australis*, in marshes of southeast Virginia. *In* T. L. Pruden and L. A. Brennan, eds. Proceedings, Fire in Ecosystem Management: Shifting the Paradigm from Suppression to Prescription. Tall Timbers Fire Ecology Conference. Tallahassee, FL: Tall Timbers Research Station. P. 140.

Cole, M.A.R. 1991. Vegetation management guideline: white and yellow sweet clover [*Melilotus alba* Desr. and *Melilotus officinalis* (L.) Lam.]. Nat. Areas J. 11:214–215.

Cowling, R. M. 1987. Fire and its role in coexistence and speciation in Gondwanan shrublands. S. Afr. J. Sci. 83:106–112.

Curtis, J. T. and M. L. Partch. 1948. Effect of fire on the competition between blue grass and certain prairie plants. Am. Midl. Nat. 39:437–443.

Dahl, B. E. and E. W. Tisdale. 1975. Environmental factors related to medusahead distribution. J. Range Manag. 28:463–468.

D'Antonio, C. M., J. T. Tunison, and R. Loh. 2000. Variation in impact of exotic grass fueled fires on species composition across an elevation gradient in Hawai'i. Aust. Ecol. 25:507–522.

Daubenmire, R. 1968. Ecology of fire in grasslands. Adv. Ecol. Res. 5:209–266.

DeBano, L. F., P. H. Dunn, and C. E. Conrad. 1977. Fire's effect on physical and chemical properties of chaparral soils. *In* Proceedings, Symposium on the Environmental Consequences of Fire and Fuel Management in Mediterranean Ecosystems. Washington, DC, U.S. Department of Agriculture, Forestry Service, General Technical Rep. WO-3. Pp 65–74.

DeBano, L. F., D. G. Neary, and P. F. Ffolliot. 1998. Fire Effects on Ecosystems. New York: Wiley and Sons. 333 p.

DeBano, L. F., S. M. Savage, and S. M. Hamilton. 1976. The transfer of heat and hydrophobic substances during burning. Soil Sci. Soc. Am. J. 40: 779–782.

Dewey, S. A., R. W. Mace, L. A. Buhler, and K. Andersen. 2000. The interaction of fire and herbicides in the control of squarrose knapweed. Proc. West. Soc. Weed Sci. 53:8–9.

DiTomaso, J. M. 1997. Risk analysis of various weed control methods. Proc. Calif. Exotic Pest Plant Counc. Symp. 3:34–39.

DiTomaso, J. M. 2000. Invasive weeds in rangelands: species, impacts and management. Weed Sci. 48:255–265.

DiTomaso, J. M., K. L. Heise, G. B. Kyser, A. M. Merenlender, and R. J. Keiffer. 2001. Carefully timed burning can control barb goatgrass. Calif. Agr. 55(6):47–53.

DiTomaso, J. M., G. B. Kyser, and M. S. Hastings. 1999. Prescribed burning for control of yellow starthistle (*Centaurea solstitialis*) and enhanced native plant diversity. Weed Sci. 47:233–242.

DiTomaso, J. M., G. B. Kyser, N. K. McDougald, R. N. Vargas, M. P. Doran, R. Wilson, and S. Orloff. 2005. Medusahead control. Proc. Calif. Weed Sci. Soc. 57:145–148.

Doerr, S. H., R. A. Shakesby, and R.P.D. Walsh. 1998. Spatial variability of soil hydrophobicity in fire-prone eucalyptus and pine forests, Portugal. Soil Sci. 163:313–324.

Dyer, A. R. and K. J. Rice. 1997. Intraspecific and diffuse competition: the response of *Nassella pulchra* in a California grassland. Ecol. Appl. 7: 484–492.

Eckardt, N. 1987. Element stewardship abstract for *Pastinaca sativa*, wild parsnip. http://tncweeds.ucdavis.edu/esadocs/documnts/pastsat.pdf. Accessed: June 1, 2005.

Ehrenfeld, J. G. 2003. Effects of exotic plant invasions on soil nutrient cycling processes. Ecosystems 6:503–523.

Ehrenreich, J. H. 1959. Effect of burning and clipping on growth of native prairie in Iowa. J. Range Manag. 12:133–137.

Emery, S. M. and K. L. Gross. 2005. Effects of timing of prescribed fire on the demography of an invasive plant, spotted knapweed *Centaurea maculosa*. J. Appl. Ecol. 42:60–69.

Engle, D. M. and P. M. Bultsma. 1984. Burning of northern mixed prairie during drought. J. Range Manag. 37:398–401.

Evans, R. D., R. Rimer, L. Sperry, and J. Belnap. 2001. Exotic plant invasion alters nitrogen dynamics in an arid grassland. Ecol. Appl. 11:1301–1310.

Fellows, D. P. and W. E. Newton. 1999. Prescribed fire effects on biological control of leafy spurge. J. Range Manag. 52:489–493.

Flematti, G. R., E. L. Ghisalberti, K. W. Dixon, and R. D. Trengove. 2004. A compound from smoke that promotes seed germination. Science 305: 977.

Fonturbel, M. T., J. A. Vega, S. Bara, and I. Bernardez. 1995. Influence of prescribed burning of pine stands in NW Spain on soil microorganisms. Eur. J. Soil Biol. 31:13–20.

Fossum, H. C. 1990. Effects of prescribed burning and grazing on *Stipa pulchra* (Hitchc.) seedling emergence and survival. M.S. thesis. University of California, Davis, CA.

Friederici, P. 1995. The alien saltcedar. Am. For. 101:45–47.

Furbush, P. 1953. Control of medusa-head on California ranges. J. For. 51: 118–121.

Garcia-Oliva, F., R. L. Sanford, and E. Kelly. 1998. Effect of burning of tropical deciduous forest soil in Mexico on the microbial degradation of organic matter. Plant Soil 206:29–36.

Gartner, F. R. 1975. Wind Cave Grassland Ecology: Final Report. Missoula, MT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station Fire Science Laboratory. 46 p.

George, M. R. 1992. Ecology and management of medusahead. Univ. Calif. Davis Dept. Agron. Range Sci. Range Sci. Rep. 32:1–3.

Glass, W. D. 1991. Vegetation management guideline: cut-leaved teasel (*Dipsacus laciniatus* L.) and common teasel (*Dipsacus sylvestris* L.). Nat. Areas J. 11:213–214.

Goodrich, S. and D. Rooks. 1999. Control of weeds at a pinyon-juniper site by seeding grasses. *In* S. B. Monsen and R. Stevens, eds. Proceedings, Ecology and Management of Pinyon–Juniper Communities within the Interior West, Provo, UT. RMRS-P-9. Rocky Mountain Research Station. Ogden, UT: U.S. Department of Agriculture, Forest Service. Pp. 403–407.

Grace, J. B. 1998. Can prescribed fire save the endangered coastal prairie ecosystem from the Chinese tallow invasion? Endang. Spec. Update 15(5):70–76.

Haskins, K. E. and C. A. Gehring. 2004. Long-term effects of burning slash on plant communities and arbuscular mycorrhizae in a semi-arid woodland. J. Appl. Ecol. 41:379–388.

Hastings, M. S. and J. M. DiTomaso. 1996. Fire controls yellow star thistle in California grasslands: test plots at Sugarloaf Ridge State Park. Restor. Manag. Notes 14:124–128.

Hatch, D. A., J. W. Bartolome, and D. S. Hillyard. 1991. Testing a management strategy for restoration of California's native grasslands. *In* Proceedings, Symposium on Natural Areas and Yosemite: Prospects for the Future. Denver, CO: U.S. National Park Service. Pp. 343–349.

Heinselman, M. L. 1981. Fire intensity and frequency as factors in the distribution and structure of northern ecosystems. *In* H. A. Mooney, T. M. Bonnicksen, N. L. Christensen, J. E. Lotan, and W. A. Reiners, eds. Proceedings, Fire Regimes and Ecosystem Properties. U.S. Department of Agriculture, Forest Service, General Technical Rep. WO-26. Pp. 7–57.

Heitlinger, M. E. 1975. Burning a protected tallgrass prairie to suppress sweetclover, *Melilotus alba* Desr. *In* M. K. Wali, ed. Prairie: A Multiple View. Grand Forks, ND: University of North Dakota. Pp. 123–132.

Hopkinson, P., J. S. Fehmi, and J. W. Bartolome. 1999. Summer burns reduce cover, but not spread, of barbed goatgrass in California grassland. Ecol. Restor. 17:168–169.

Hull, A. C. and G. Stewart. 1948. Replacing cheatgrass by reseeding with perennial grass on southern Idaho ranges. Agron. J. 40:694–703.

Hutchison, M. 1992. Vegetation management guideline: Canada thistle (*Cirsium arvense* (L.) Scop.). Nat. Areas J. 12:160–161.

Jacobs, J. S. and R. L. Sheley. 2003a. Prescribed fire effects on dalmatian toadflax. J. Range Manag. 56:193–197.

Jacobs, J. S. and R. L. Sheley. 2003b. Testing the effects of herbicides and prescribed burning on dalmatian toadflax (Montana). Ecol. Restor. 21: 138–139.

Keeley, J. E. 1977. Fire dependent reproductive strategies in *Arctostaphylos* and *Ceanothus*. *In* H. A. Mooney and C. E. Conrad, eds. Proceedings, Symposium on the Environmental Consequences of Fire and Fuel Management in Mediterranean Ecosystems. U.S. Department of Agriculture, Forest Service, General Technical Rep. WO-3. Pp. 371–376.

Kilgore, B. M. 1981. Fire in ecosystem distribution and structure: western forests and scrublands. *In* H. A. Mooney, T. M. Bonnicksen, N. L. Christensen, J. E. Lotan, and W. A. Reiners, eds. Proceedings, Fire Regimes and Ecosystem Properties. U.S. Department of Agriculture, Forest Service, General Technical Rep. WO-26. Pp. 58–89.

Kline, V. M. 1983. Control of sweet clover in a restored prairie (Wisconsin). Restor. Manag. Notes 1(4):30–31.

Klinger, R. C. and R. Brenton. 2000. The relative effectiveness of prescribed burning and herbicide application for controlling fennel on Santa Cruz Island, California. *In* W. K. Moser and C. F. Moser, eds. Proceedings, Fire and Forest Ecology: Innovative Silviculture and Vegetation Management. Tall Timbers Fire Ecology Conference. Tallahassee, FL: Tall Timbers Research Station 21. P. 137.

Klopatek, C. C., C. F. Friese, M. F. Allen, and J. M. Klopatek. 1994. Comparisions of laboratory and field burning experiments on mycorrhizae distribution, density and diversity. J. Soc. Am. For. 94:762–776.

Knapp, A. K. and T. R. Seastedt. 1986. Detritus accumulation limits productivity of tallgrass prairie. BioScience 36:662–668.

Korb, J. E., N. C. Johnson, and W. W. Covington. 2003. Arbuscular mycorrhizal propagule densities respond rapidly to ponderosa pine restoration treatments. J. Appl. Ecol. 40:101–110.

Korb, J. E., N. C. Johnson, and W. W. Covington. 2004. Slash pile burning effects on soil biotic and chemical properties and plant establishment: recommendations for amelioration. Restor. Ecol. 12:52–62.

Kyser, G. B. and J. M. DiTomaso. 2002. Instability in a grassland community after the control of yellow starthistle (*Centaurea solstitialis*) with prescribed burning. Weed Sci. 50:648–657.

Launonen, T. M., D. H. Ashton, and P. J. Keane. 1999. The effect of regeneration burns on the growth, nutrient acquisition and mycorrhizae of *Eucalyptus regnans* F. Muell. (mountain ash) seedlings. Plant Soil. 210: 273–283.

Lesica, P. and B. Martin. 2003. Effects of prescribed fire and season of burn on recruitment of the invasive exotic plant, *Potentilla recta*, in a semiarid grassland. Restor. Ecol. 11:516–523.

Levitt, J. 1972. Responses of Plants to Environmental Stresses. New York: Academic. 697 p.

Mabuhay, J. A. and N. Nakagoshi. 2003. Microbial biomass and abundance after forest fire in pine forests in Japan. Ecol. Res. 18:431–441.

Masters, R. A. and S. J. Nissen. 1998. Revegetating leafy spurge (*Euphorbia esula*)–infested rangeland with native tallgrasses. Weed Technol. 12:381–390.

Masters, R. A., S. J. Nissen, R. E. Gaussoin, D. D. Beran, and R. N. Stougaard. 1996. Imidazolinone herbicides improve restoration of Great Plains grasslands. Weed Technol. 10:392–403.

Mayeux, H. S. and W. T. Hamilton. 1988. Response of false broomweed and associated herbaceous species to fire. J. Range Manag. 41:2–6.

McDaniel, K. C., C. R. Hart, and D. B. Carroll. 1997. Broom snakeweed control with fire on New Mexico blue gramma rangeland. J. Range Manag. 50:652–659.

McKell, C. M., A. M. Wilson, and B. L. Kay. 1962. Effective burning of rangelands infested with medusahead. Weeds 10:125–131.

McMurphy, W. E. and K. L. Anderson. 1965. Burning Flint Hills range. J. Range Manag. 18:265–269.

Menke, J. W. 1992. Grazing and fire management for native perennial grass restoration in California grasslands. Fremontia 20(2):22–25.

Meyer, M. D. and P. M. Schiffman. 1999. Fire season and mulch reduction in a California grassland: a comparison of restoration strategies. Madroño 46:25–37.

Miller, J. R. 2003. Integrated Management of Yellow Starthistle (*Centaurea solstitialis*) at Fort Hunter Liggett, Monterey County, California. M.S. thesis. University of California, Davis, CA.

Miller, R. F. and R. J. Tausch. 2001. The role of fire in juniper and pinyon woodlands: a descriptive analysis. *In* K.E.M. Galley and T. P. Wilson, eds. Proceedings, Invasive Species Workshop: The Role of Fire in the Control and Spread of Invasive Species. Fire Conference 2000: First National Congress on Fire Ecology, Prevention, and Management. Tallahassee, FL: Tall Timbers Research Station. Pp. 15–30.

Miller, S. L., T. M. McClean, N. L. Stanton, and S. E. Williams. 1998. Mycorrhization, physiognomy, and first-year survivability of conifer seedlings following natural fire in Grand Teton National Park. Can. J. For. Res. 28:115–122.

Mills, A. J. and M. V. Fey. 2004. Frequent fires intensify soil crusting: physicochemical feedback in the pedoderm of long-term burn experiments in South Africa. Geoderma 121:45–64.

Mitchell, R., C. Britton, B. Racher, E. Fish, and E. Atkinson. 2000. Prescribed fire costs on juniper-infested rangeland. Rangelands 22(5):7–10.

Morghan, K.J.R., T. R. Seasted, and P. J. Sinton. 2000. Frequent fire slows invasion of ungrazed tallgrass prairie by Canada thistle (Colorado). Ecol. Restor. 18:194–195.

Murphy, A. H. and W. C. Lusk. 1961. Timing medusahead burns. Calif. Agr. 15:6–7.

Neary, D. G., C. C. Klopatek, L. F. Debano, and P. F. Ffolliott. 1999. Fire effects on belowground sustainability: a review and synthesis. For. Ecol. Manag. 122:51–71.

Nelson, L. L. and E. B. Allen. 1993. Restoration of *Stipa pulchra* grasslands: effects of mycorrhizae and competition from *Avena barbata*. Restor. Ecol. 1:40–50.

Nuzzo, V. A., W. McClain, and T. Strole. 1996. Fire impact on groundlayer flora in a sand forest 1990–1994. Am. Midl. Nat. 136:207–221.

Odion, D. C. and K. A. Haubensak. 1997. Response of French broom to fire. *In* M. Morales and T. Morales, eds. Proceedings, Fire in California Ecosystems: Integrating Ecology, Prevention and Management. San Diego, CA: Associaton for Fire Ecology. Pp. 296–307.

Parsons, D. J. and S. DeBenedetti. 1984. Effects of prescribed fire on annual grasslands in Sequoia National Park. Bull. Ecol. Soc. Am. 65:258.

Parsons, D. J. and T. J. Stohlgren. 1989. Effects of varying fire regimes on annual grasslands in the southern Sierra Nevada of California. Madroño 36:154–168.

Paynter, Q. and G. J. Flanagan. 2004. Integrating herbicide and mechanical control treatments with fire and biological control to manage an invasive wetland shrub, *Mimosa pigra*. J. Appl. Ecol. 41:615–629.

Pendergrass, K. L., P. M. Miller, and J. B. Kauffman. 1988. Prescribed fire and the response of woody species in Willamette Valley wetland prairies. Restor. Ecol. 6:303–311.

Pollak, O. and T. Kan. 1996. The use of prescribed fire to control invasive exotic weeds at Jepson Prairie Preserve. *In* C. W. Witham, ed. Proceedings, Ecology, Conservation and Management of Vernal Pool Ecosystems. Sacramento, CA: California Native Plant Society. Pp. 241–249.

Pyne, S. J. 1997. America's Fires: Management on Wildlands and Forests. Durham, NC: Forest History Society. 54 p.

Queensland Government. 2004. Natural Resources and Mines. Mesquite (*Prosopis* spp.). Fact Sheet. Queensland Government Web page: http://www.nrm.qld.gov.au/factsheets/pdf/pest/PP37.pdf. Accessed: June 1, 2005.

Rashid, A., T. Ahmed, N. Ayub, and A. G. Khan. 1997. Effect of forest fire on number, viability and post-fire re-establishment of arbuscular mycorrhizae. Mycorrhiza 7:217–220.

Raunkiaer, C. 1934. The Life Forms of Plants and Statistical Plant Geography; Being the Collected Papers of C. Raunkiaer. Translated by H. Gilbert-Carter, A. I. Fausbøll, and A. G. Tansley. Oxford, U.K.: Clarendon Press. 632 p.

Renney, A. J. and E. C. Hughes. 1969. Control of knapweed *Centaurea* species in British Columbia with Tordon herbicide. Down to Earth 24:6–8.

Rhoades, C., T. Barnes, and B. Washburn. 2002. Prescribed fire and herbicide effects on soil processes during barrens restoration. Restor. Ecol. 10:656–664.

Richburg, J. A. and W. A. Patterson III. 2003. Can northeastern woody invasive plants be controlled with cutting and burning treatments? *In* K. P. Bennett, A. C. Dibble, and W. A. Patterson III, eds. Proceedings, Using Fire to Control Invasive Plants: What's New, What Works in the Northeast? Portsmouth, NH: University of New Hampshire Cooperative Extension. Pp. 1–3.

Robocker, W. C. and B. J. Miller. 1955. Effects of clipping, burning and competition on establishment and survival of some native grasses in Wisconsin. J. Range Manag. 8:117–120.

Rolston, M. P. and J. Talbot. 1980. Soil temperature and regrowth of gorse burnt after treatment with herbicides. N. Z. J. Exp. Agric. 8:55–61.

Rundel, P. W. 1983. Impact of fire on nutrient cycles in Mediterranean-type ecosystems with reference to chaparral. *In* F. J. Kruger, D. T. Mitchell, and J.U.M. Jarvis, eds. Mediterranean-Type Ecosystems: The Role of Nutrients. New York: Springer-Verlag. Pp. 192–220.

Sando, R. W. 1978. Natural fire regimes and fire management—foundations and direction. West. Wildlands 4(4):34–42.

Schacht, W. and J. Stubbendieck. 1985. Prescribed burning in the Loess Hills mixed prairie of southern Nebraska. J. Range Manag. 38:47–51.

Schwarzmeier, J. A. 1984. Sweet clover control in planted prairies: refined mow/burn prescription tested (Wisconsin). Restor. Manag. Notes 2:35–36.

Seastedt, T. R. and R. A. Ramundo. 1990. The influence of fire on belowground processes of tallgrass prairie. *In* S. L. Collins and L. L. Wallace, eds. Fire in North American Tallgrass Prairies. Norman, OK: University of Oklahoma. Pp. 99–117.

Sharp, L. A., M. Hironaka, and E. W. Tisdale. 1957. Viability of medusahead (*Elymus caput-medusae* L.) seed collected in Idaho. J. Range Manag. 10: 123–126.

Shipman, R. D. 1962. Establishing Forest Plantations on Areas Occupied by Kudzu and Honeysuckle. South Carolina Agricultural Experiment Station, Forest Research Service 5. 21 p.

Sigüenza, C., L. Corkidi, and E. B. Allen. 2006. Feedbacks of soil inoculum of mycorrhizal fungi altered by N deposition on the growth of a native shrub and an invasive annual grass. Plant Soil In press.

Smith, M. D. and A. K. Knapp. 1999. Exotic plant species in a C_4-dominated grassland: invasibility, disturbance, and community structure. Oecologia 120:605–612.

Smith, M. D. and A. K. Knapp. 2001. Size of the local species pool determines invasibility of a C_4-dominated grassland. Oikos 92:55–61.

Snyman, H. A. 2002. Fire and the dynamics of a semi-arid grassland: influence on soil characteristics. Afr. J. Range Forage Sci. 19:137–145.

Snyman, H. A. 2003. Short-term response of rangeland following an unplanned fire in terms of soil characteristics in a semi-arid climate of South Africa. J. Arid Environ. 55:160–180.

Stendell, E. R., T. R. Horton, and T. D. Bruns. 1999. Early effects of prescribed fire on the structure of the ectomycorrhizal fungus community in a Sierra Nevada ponderosa pine forest. Mycol. Res. 103:1353–1359.

Steuter, A. A. 1988. Wormwood sage controlled by spring fires (South Dakota). Restor. Manag. Notes 6:35.

Svedarsky, W. D., P. E. Buckley, and T. A. Feiro. 1986. The effect of 13 years of annual burning on an aspen–prairie ecotone in northwestern Minnesota. *In* G. K. Clambey and R. H. Pemble, eds. Proceedings, 9th North American Prairie Conference—The Prairie: Past, Present and Future. Moorhead, MN, and Fargo, ND: Tri-College University. Pp. 118–122.

Sweet, S. B. 2005. Effects of seed maturity on the resistance of weed seeds to simulated fire: implications for timing prescribed burns. M.S. Thesis. University of California, Davis, CA. 56 pp.

Swezy, M. and D. C. Odion. 1997. Fire on the mountain: a land manager's manifesto for broom control. *In* M. Kelly, E. Wagner, and P. Warner, eds. Proc. Calif. Exotic Pest Plant Counc. Symp. 3:76–81.

Taylor, J. P. and K. C. McDaniel. 1998. Restoration of saltcedar (*Tamarix* sp.)–infested floodplains on the Bosque del Apache National Wildlife Refuge. Weed Technol. 12:345–352.

Thompson, D. J. and J. M. Shay. 1989. First-year response of a *Phragmites* marsh community to seasonal burning. Can. J. Bot. 67:1448–1455.

Towne, G. and C. Owensby. 1984. Long-term effects of annual burning at different dates in ungrazed Kansas tallgrass prairie. J. Range Manag. 37: 392–397.

Tunison, J. T., C. M. D'Antonio, and R. K. Loh. 2001. Fire and invasive plants in Hawai'i Volcanoes National Park. *In* K.E.M. Galley and T.P. Wilson, eds. Proceedings, Invasive Plant Workshop: The Role of Fire in the Control and Spread of Invasive Species. Fire Conference 2000: First National Congress on Fire Ecology, Prevention, and Management. Tallahassee, FL: Tall Timbers Research Station Misc. Pub. 11. Pp. 122–131.

Ueckert, D. N., J. L. Petersen, R. L. Potter, J. D. Whipple, and M. W. Wagner. 1988. Managing Pricklypear with Herbicides and Fire. Texas Agricultural Experiment Station Rep. 4570. Pp. 10–15.

Vale, T. R., ed. 2002. Fire, Native Peoples, and the Natural Landscape. Washington, DC: Island Press. 238 p.

Washburn, B. E., T. G. Barnes, C. C. Rhoades, and R. Remington. 2002. Using imazapic and prescribed fire to enhance native warm-season grasslands in Kentucky, USA. Nat. Areas J. 22:20–27.

Washburn, B. E., T. G. Barnes, and J. D. Sole. 1999. No-till establishment of native warm-season grasses in tall fescue fields: first-year results indicate value of new herbicide. Ecol. Restor. 17:144–149.

Wendtland, K. J. 1993. Fire history and effects of seasonal prescribed burning on northern mixed prairie, Scotts Bluff National Monument, Nebraska. M.S. thesis. University of Wyoming, Laramie, WY. 188 p.

Whelan, R. J. 1995. The Ecology of Fire. New York: Cambridge University. 346 p.

Whisenant, S. G. 1990. Postfire population dynamics of *Bromus japonicus*. Am. Midl. Nat. 123:301–308.

Whisenant, S. G., D. N. Ueckert, and C. J. Scifres. 1984. Effects of fire on Texas wintergrass communities. J. Range Manag. 37:387–391.

Whisenant, S. G. and D. W. Uresk. 1990. Spring burning Japanese brome in a western wheatgrass community. J. Range Manag. 43:205–208.

White, L. L. and D. R. Zak. 2004. Biomass accumulation and soil nitrogen availability in an 87-year-old *Populus grandidentata* chronosequence. For. Ecol. Manag. 191:121–127.

Willson, G. D. 1992. Morphological characteristics of smooth brome used to determine a prescribed burn date. *In* D. D. Smith and C. A. Jacobs, eds. Proceedings, 12th North American Prairie Conference: Recapturing a Vanishing Heritage. Cedar Falls, IA: University of Northern Iowa. Pp. 113–116.

Willson, G. D. and J. Stubbendieck. 1996. Suppression of smooth brome by atrazine, mowing, and fire. Prairie Nat. 28:13–20.

Winter, B. 1993. Leafy spurge control in a tallgrass prairie natural area. Leafy Spurge News 15(1):3.

Wolters, G. L., C. H. Sieg, A. J. Bjugstad, and F. R. Gartner. 1994. Herbicide and fire effects on leafy spurge density and seed germination. Fort Collins, CO: U.S. Department of Agriculture, Forest Service Research Notes RM-526. 5 p.

Wright, H. A. and A. W. Bailey. 1982. Fire Ecology: United States and Southern Canada. New York: J. Wiley. 501 p.

Young, J. A. and R. A. Evans. 1978. Population dynamics after wildfires in sagebrush grasslands. J. Range Manag. 31:283–289.

Young, J. A., R. A. Evans, and B. L. Kay. 1970. Phenology of reproduction of medusahead. Weed Sci. 18:451–454.

Young, J. A., R. A. Evans, and J. Robison. 1972. Influence of repeated annual burning on a medusahead community. J. Range Manag. 25:372–375.

Youtie, B., J. Ponzetti, D. Salzer, and J. Soll. 1998. Controlling annual exotic grasses on relict grasslands (Oregon). Restor. Manag. Notes 16:209.