The Use of Fire as a Tool for Controlling Invasive Plants

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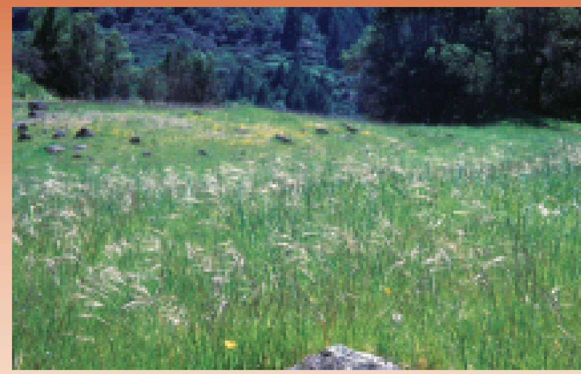
Much of what we currently know about using fire to manage vegetation—and to control invasive plant species in particular—has been derived from studies of cropland systems. However, there are many fundamental differences between cropland and wildland settings, and our ability to use effects observed in croplands to predict effects that may occur in wildlands is limited. Some of these fundamental differences include the timing of fires, fuel types, fire types, other treatments that come before or after burning, and the types of invasive plants that are targeted.

The goal of this report is to capture the current state of knowledge on the use of fire as a tool to manage invasive plants in wildlands. By providing a more thorough source of information on this topic, we hope this review facilitates improved decision making when considering the use of prescribed burning for the management of invasive plants.

—from the Introduction by Matthew L. Brooks, U.S. Geological Survey, Western Ecological Research Center



***Purple needlegrass** (Nassella pulchra) and other native plants increased greatly in abundance following consecutive prescribed burns for control of yellow starthistle at Sugarloaf Ridge State Park. Sonoma County, CA. (Photo by J.M. DiTomaso, University of California, Davis)*

***Post-burn monitoring** documents changes to plant community. US Department of Agriculture rangeland scientists measure regrowth after a burn. (Photo by Stephen Ausmus, USDA, Agricultural Research Service).*

On the cover:
***Grassland ignition** with a drip torch. Application is perpendicular to wind direction, with flames blowing toward areas already burned. (Photo by Jennifer Chapman, National Park Service)*

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THE USE OF FIRE AS A TOOL FOR CONTROLLING INVASIVE PLANTS

California Invasive Plant Council

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science for a changing world

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To obtain copies of this report, or to submit additional information for future revisions, contact the California Invasive Plant Council through its website: www.cal-ipc.org.

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This report is based on input from an expert committee of scientists and land managers experienced in the use of fire as a tool to manage invasive plants. The following individuals participated in a workshop March 29-30, 2004, hosted by the U.S. Geological Survey in Henderson, NV.

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Introduction

Matthew L. Brooks
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Fire is one of the oldest tools used by humans to manage vegetation. Its use can be traced back to pre-historic times when it was used to manipulate vegetation to improve opportunities for hunting wildlife and to increase production of plant species that were used for food, textiles, shelter, and other practical applications (Vale 2002).

Modern use of fire in wildland areas increased during the latter part of the 1900s. "Prescribed fire" has been used to reduce hazardous fuel loads, restore historical disturbance regimes, improve forage and habitat for game and livestock species, and promote biodiversity. In some cases, fire has also been used to manage invasive plant species.

Much of what we currently know about using fire to manage vegetation—and to control invasive plant species in particular—has been derived from studies of cropland systems. However, there are many fundamental differences between cropland and wildland settings, and our ability to use effects observed in croplands to predict effects that may occur in wildlands is limited. Some of these fundamental differences include the timing of fires, fuel types, fire types, other treatments that come before or after burning, and the types of invasive plants that are targeted (Table 1).

For example, fire is normally used in croplands as a technique to remove dead plant material left after harvesting. This is done to facilitate soil work (e.g., disking, plowing), suppress overwintering pathogenic fungal spores, or reduce the seed banks of crop competitors. Cropland fuels are typically dried crop stubble, often supplemented by fossil fuel accelerants. In wildlands, fires may occur anytime during the invasive plant's life cycle, provided fuel moisture and weather are sufficient to carry a fire. Cropland systems are also inherently less complex than wildland systems. With fewer parts and interactions among parts, it is easier to reliably predict the outcome of fires in a cropland setting. Because of these differences, there is a significant need for information specific to the effects of prescribed fire on invasive plants in wildland ecosystems.

The goal of this report is to capture the current state of knowledge on the use of fire as a tool to manage invasive plants in wildlands. It summarizes current literature and observations on: the risks and challenges of conducting prescribed burns; the types of systems where burning, used alone or as part of an integrated approach, can be effective for the management of invasive plants; the impacts of prescribed burning on the broader plant community and soils. By providing a more thorough source of information on this topic, we hope this review facilitates improved decision making when considering the use of prescribed burning for the management of invasive plants.

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

	Croplands	**Wildlands**
Timing of Fires	Pre- or post-harvest	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 or crown fire
Other Integrated Treatments	Fire preceded by chemical or mechanical treatments, followed by a cover crop	Followed by chemical or mechanical treatments, or revegetation with competitive species
Type of Invasives Targeted	Typically herbaceous	Varies widely—grasses, herbs, shrubs and trees
Ecological Complexity	Low	High

CHAPTER 1: Planning and Implementing Prescribed Burns

Ralph Minnich
California Department of Forestry and Fire Protection

Prescribed burning is a valuable tool in the effort to manage invasive plants. Certainly fire can reduce the volume of plant material, and has been shown to reduce the re-establishment of invasive plants. The use of prescribed burning, however, can also create a series of issues and concerns that must be addressed during the planning process and incorporated into the management program.

Motivation and Limitations

The motivation for developing a prescribed burn project is the commitment to reduce or remove invasive plants from a specific area in order to promote the native vegetation community. Prescribed burning is viewed as a means of treating natural areas—sometimes large—in a cost effective manner. This motivation includes the belief that clear benefits will result from the project and that obstacles can be overcome. Factors that can limit a successful outcome include restrictions on allowable burn area due to smoke impacts, lack of a suitable time window for completing the burn, opposition from neighbors and the community, unwillingness of employees to assume the additional workload or responsibility, lack of commitment at higher levels of an organization, and lack of support from regulatory agencies.

Responsibility and Liability

There are three major responsibilities associated with planning and implementing a prescribed burn, and these responsibilities carry significant legal liability. First, an organization or individual must be designated as responsible for the project. The responsible party, sometimes called the lead agency, sets the objectives for the proposed project and determines if the use of prescribed burning is an appropriate tool to accomplish the objectives. The responsible party conducts a comprehensive review to identify, and if necessary mitigate, factors that could influence the burn. The responsible party needs to ensure that concerns of regulatory agencies are addressed and that fees are paid. The responsible party will also either assume full legal liability for the project, share liability with others, or provide the terms for limited liability. Legal liability applies to personnel, any injuries that result from the burn, property damage, vehicle use, directing the resources, and conducting the burn within prescription parameters.

Secondly, the responsible agency is required to appoint a coordinator who develops and obtains approval for the plan. For approved projects, the coordinator may also be tasked with ensuring that preparations for conducting the project are accomplished.

The third responsibility is the designation of a qualified fire manager to execute the project. The duties of the prescribed fire manager include ensuring that resources are available, determining that fuel and weather conditions are within the parameters of the burn prescription, informing regulatory agencies and media of the burn, managing the activity and

Agencies coordinate leadership responsibilities for prescribed burns. *On this burn to control yellow starthistle at Sugarloaf Ridge State Park, personnel from State Parks and the California Department of Forestry and Fire Protection coordinated team activities including preparation of fire lines, equipment crews, and burners and beaters. Sonoma County, California, early July, 1995. (Photo by Joe DiTomaso, University of California, Davis)* resources during the burn, and conducting the burn in a safe manner to achieve the objectives.

Availability of Resources

Prescribed burning requires extensive resources, but the benefits can also be extensive. A successful prescribed burn utilizes resources efficiently, both during the preparation and implementation phases. On site, qualified individuals are necessary to coordinate and manage the project. Hand crews and bulldozers prepare containment lines at pre-determined locations. A combination of personnel and igniting equipment (ranging from hand-held torches to helicopter firing devices) is necessary for firing the area. Resources for containing the fire, consisting of fire crews, fire engines, bulldozers and crews must be immediately available to suppress embers, spots, and fire outside the line. Some agencies have these resources available internally, while others have agreements with other agencies for access to these resources.

Most of the available resource funding within an agency may be devoted to fire suppression and not for conducting prescribed burns. Furthermore, the ideal timing for a prescribed burn, particularly in the summer and fall, may not coincide with adequate availability of resources due to other wildland fire activity, competing projects, or limited funds. Private vendors may have fire crews and engines available for hire, but these resources are usually very expensive and must be scheduled with sufficient lead time to assure staffing.

Training and Qualifications

The training and qualifications requirements for all personnel have become increasingly important. Many disciplines must collaborate if a burn is to be successful. It is crucial that personnel assigned to manage the activity have training in the application of prescribed fire as well as experience in "Incident Command System" organizational structure, fire behavior, fuels management, and communications. The Incident Commander must have expertise in burning under wildland fuel conditions. Each person involved in the burn must either meet the qualifications of the assignment or be supervised by a qualified person.

The cost of training, especially for classes outside of the employee's organization, can be high. Training for a desired function often requires prerequisite training and experience. Often knowledge and expertise is gained on the job, incrementally with years of experience in a number of projects.

Containment strategies are implemented before a burn. *Control lines around the edge of the burn unit can be set by wet-lining with water and foam. (Photo by Jennifer Chapman, National Park Service)*

Some agencies choose to hire or subcontract a vendor or other agency to conduct prescribed burns. While this may be expedient, the cost is fairly high and liability for the burn remains with the responsible party.

The availability of qualified personnel to command, coordinate and supervise a prescribed burn project can be highly variable. The loss of personnel due to retirements, re-assignments, promotions and other factors can lead to an inadequate pool of personnel to fill the needed functions. Unless an organization is proactively providing training classes and experience opportunities, fewer qualified people will be ready to assume positions.

Opportunities to complete an approved project can be limited, especially if burn prescription parameters are very narrow. For instance, a prescribed burn involving target vegetation surrounded by highly flammable fuels will require strict adherence to a cooler prescription, and a prescribed burn in a confined environment, such as within an urban or suburban setting, requires a higher level of coordination and more resources to ensure that surrounding properties are protected. Finding and responding promptly to a favorable time window in these circumstances require an individual familiar with the project and the burn prescription to closely monitor fuels conditions and weather forecasts in anticipation of such a window.

Safety Equipment

Personnel assigned to conduct prescribed burning must be supplied with and wear the appropriate Personal Protective Equipment (PPE). Heavy leather boots, helmet and gloves must be worn. Although 100% cotton pants and shirt may be acceptable, PPE usually includes flame retardant pants and shirt. Organizations should provide, or even require, the highest level of burn protection for their employees, particularly for scheduled burn projects. For fire suppression resources, an emergency fire shelter is also mandatory according to OSHA and Cal OSHA regulations.

Budget Limitations

Many organizations, public and private, are faced with reductions in budgets or demands that existing budgets be stretched to accomplish a wider range of projects. The outcome of a "do more with less" approach is usually less getting done. In addition, budget reductions can result in a proposed solution that is not the best alternative, but merely the cheapest, or follow-up efforts that are reduced or nonexistent, which can negate the benefits of initial efforts. Many organizations lack a long-term commitment for funding to ensure treatments of invasive plants over an extended period. The prohibition on carrying money from one fiscal year to the next, even though dedicated to an approved project, seriously hampers multi-year efforts. Some organizations are able to develop partnerships with other public and private entities, allowing them to capitalize on cost-sharing to achieve the long-term desired results.

As budget and staffing limitations arise, managers must choose among project proposals with particular attention to the opportunity costs of each project. Limitations dictate that efforts be channeled toward projects that address the most damaging invasive plants with the greatest likelihood of success. As a result, response to new invaders is typically neither immediate nor adequate.

Two national programs, Invasive Species Control and Fire Plan, have complimentary goals and objectives. However, competition for funding between the two groups and narrow exclusionary focus within each agency reduces the potential for mutual benefits. There is a definite need to integrate all issues—cultural values, natural resources and wildland fire—into a cohesive framework of opportunity. Data on the cost of prescribed burning need to be available for reference during the planning process to assist with developing budget requests and allocations, and as comparisons for evaluating the efficiency and cost-effectiveness of the project.

Project Review Process

The owners and land managers involved in the planning phase of a prescribed burn project usually have specific objectives and desired outcomes, as well as concerns about the impacts of burning. In some cases these concerns can be alleviated, while in other cases they can prevent the project from moving forward. It is important to identify all negative concerns early in the process in order to develop proposals for mitigating adverse impacts.

Multiple ownerships add a significant degree of difficulty when trying to gain consensus for the objectives of the burn. Some land-use classifications may prohibit the implementation of prescribed fire. Project size can also be a factor, as some efforts to treat invasive plant species require that all infected acreage be included in the prescribed burn.

Many agencies can be involved in the review of a proposed project. These typically include the US Fish and Wildlife Service, US Environmental Protection Agency, state Fish and Game agencies, Water Quality Control Boards, Air Quality Management Districts, advocates for threatened and endangered species, archaeological and historical organizations, Native American cultural groups, and various proponents and opponents of prescribed

Mop-up activities take place after a burn. *Care is taken to make sure hot spots of smoldering fuel won't flare up. (Photo by Chuck Schoendienst, California Department of Forestry and Fire Protection)* fire. These organizations each review projects from a narrow perspective, which can be diametrically opposite to the input from other organizations. Thus, project review comments provided by one organization may be in direct conflict with other organizations. The time, energy and patience needed to develop consensus and conduct a prescribed burn usually exceed estimates.

Agreements

Because planning and implementation of a prescribed burn typically involve multiple pubic and private parties, some governmental agencies have developed general agreements which ease the way for multi-party involvement (such as the Vegetation Management Program in California, through Public Resources Code and the VMP Handbook). Absent an established planning and agreement process, individual agreements and processes must be created to assure compliance with environmental review and to accommodate comment periods. This individualized process consumes a considerable amount of time and makes agencies less willing to commit the effort necessary to complete projects.

Education

Suppression of wildfires ignited by natural causes (lightning), human activity, or other sources rarely receives public complaints. Strangely enough, however, a prescribed burning project whose goal is to solve an identified problem, such as controlling invasive plant species, often results in a barrage of public complaints and obstacles. This reaction can result from a lack of understanding of the problem, disagreement with the method being proposed, a reluctance to take a pro-active approach, distrust of the organization proposing the project, or transference of previous problems or concerns.

To overcome these obstacles and proceed with the project, the lead agency must provide as much information and education as possible for the cooperators, neighbors and community. This is best achieved by encouraging participation in the planning process and accepting comments from interested individuals. Concerns need to be addressed and, as necessary, mitigated. Urban dwellers generally have less inclination to support projects involving burning, mostly due to the potential impacts of smoke and property damage. People in rural areas, especially those allied with agriculture, usually understand the impacts of invasive plants and are more supportive of control efforts. When burns are conducted along roadways, the traveling public is prone to voicing complaints regarding any impact on traffic flow.

Air quality and road closures are major public concerns. *Smoke does not disperse well when a high-pressure system keeps hot air at the surface. This burn at Point Reyes National Seashore was shut down because stable atmospheric conditions did not change. (Photo by Jennifer Chapman, National Park Service)*

Data on Species Response to Fire

Research on the preferred growing conditions of many plant species has been ongoing for many years. However, the responses of many of these species to fire parameters, including intensity, duration, return interval and time of year, have not been clearly delineated and in many cases not included in the studies. Adding to the challenge of studying the effects of burning is the relative gain or decline of native species in relation to invasive species. This topic is covered in more detail in later chapters of this report.

Alternatives

The alternatives to prescribed burning are limited. Handwork is expensive, proceeds slowly and may need to be repeated. In addition, mechanical methods using large equipment do not allow selective targeting of specific plant species. Chemical treatments are increasingly limited due to decreased availability of suitable chemicals, stricter regulation, and opposition from groups and individuals. Thus the utility of prescribed fire relative to other alternatives is likely to become increasingly acknowledged.

CHAPTER 2: Control of Invasive Plants With Prescribed Fire

Joseph M. DiTomaso
University of California, Davis

Objectives of a Prescribed Burn

Most natural plant ecosystems have adapted to fire as a frequent or infrequent disturbance. In many areas, these natural fire regimes have been altered by humans. In some cases the fire frequency interval is shortened and in others the intervals between fires has lengthened. For example, cheatgrass or downy brome (*Bromus tectorum*) has invaded much of the western United States and has increased fire frequency to the point that native shrub species cannot survive or persist (Brooks *et al.* 2004).

Prescriptive burning can be used for a number of reasons. In some areas, particularly roadsides or forested regions, it may reduce accidental fire hazard by reducing fuel loads. It is also used in efforts to return an area to more natural fire regimes, which can reduce catastrophic wildfire events.

Increasingly, prescribed fire is being used as a tool to increase native plant species and provide wildlife habitat. For instance, perennial grasslands can be burned on a regular interval to increase native bunchgrasses or to stimulate the germination of legumes by reducing the suppressive thatch layer. A major aspect of using fire as a habitat restoration tool is its role in the management of invasive plants, which can include annuals, perennials and woody species. This can be through direct damage and suppression of the target species, or as part of an integrated approach in which fire facilitates more effective use of another control strategy, including mechanical, cultural, or chemical options. The discussion in this chapter focuses on the use of fire to directly manage invasive plants. The subsequent chapter in this volume (Chapter 3) discusses fire as part of an integrated approach in conjunction with other control techniques.

Fire can significantly reduce invasive plant populations. *Three consecutive years of burning in the area to the left of the road have reduced the yellow starthistle seedbank by more than 90%. Grasses and legumes have replaced starthistle in burned areas, while areas in the foreground remain densely infested. Sugarloaf Ridge State Park, Sonoma County, California, July, 1998. (Photo by Joe DiTomaso, University of California, Davis)*

Information Sources on the Effect of Prescribed Burning on Invasive Plant Management

Dr. Peter Rice recently completed a review of the literature on the use of prescribed burning for control of invasive species. This comprehensive literature review can be found on the Center for Invasive Plant Management website (www.weedcenter.org).

In addition, the US Forest Service's Fire Effects Information System (FEIS) provides current scientific and technical website information about fire effects on plants and animals, including invasive species (www.fs.fed.us/database/feis/plants/weed/weedpage.html). This site contains information on one thousand species and was developed at the Forest Service's Rocky Mountain Research Station, Fire Sciences Laboratory in Missoula, Montana.

Included in this information is a thorough evaluation of the interactions between fire and non-native invasive plant species, particularly with respect to (1) the role of fire in enabling plant invasions, (2) altered fire regimes following plant invasion, (3) the use of fire to control plant invasions, and (4) background information on the biology, ecology, phenology, distribution, and management of invasive plants. Published information on how to control weeds is augmented by personal observations. The invasive species entries are cross-referenced with dominant species or vegetation types, with information on the fire regimes of the vegetation types, and the effects of fire on natives.

Growth Forms

Life history of invasive plants can often determine their direct susceptibility to fire. In the western United States, fire is most effective on annual species, both grasses and broadleaf weeds. However, many annual grasses and forb species of rangelands and wildlands in the western states are winter annuals that generally germinate with the first rainfall events of fall and mature in mid- to late spring. These species are difficult to control with prescribed fire because fuel loads are typically not sufficient before seeds have matured, and conversely, once these annuals have cured and can be burned, their seeds have already dispersed and are no longer susceptible to destruction by grassland fires. Species selectivity is also difficult to achieve in these cases.

Some winter annual species, however, have a longer life cycle and do not mature until early to midsummer. These include some top invasive plants, such as the annual grasses medusahead (*Taeniatherum caput-medusae*), red brome (*Bromus madritensis* ssp. *rubens* [=*B. rubens*]), Japanese brome (*Bromus japonicus*), and barb goatgrass (*Aegilops triuncialis*), as well as some forb species, particularly yellow starthistle (*Centaurea solstitialis*). For these species, timely burns can be used as an effective control strategy once combustible fuel is sufficient, but prior to seed maturation or dispersal.

Biennial species, which complete their life cycle within two years, are herbaceous plants that germinate and typically exist as basal rosettes in the first year, then bolt, flower, and die in the second year. Single burn events do not control biennial species. In some cases they can be managed by fire, but this requires either multiple-year burns or an integrated approach using other control options.

Herbaceous perennial grasses and forbs survive for more than two years. They are more difficult to control with fire than invasive annuals. This is particularly true for perennial forbs, for which there are few examples where fire alone has proven successful. In contrast, there have been several reports, mainly from eastern states, of perennial grasses being controlled by prescribed burning. Most reports are for the management of Kentucky bluegrass (*Poa pratensis*), Canada bluegrass (*Poa compressa*), or smooth brome (*Bromus inermis*). Proper timing is required to achieve successful and selective control of these species.

Timing is key to controlling plants and seeds. *Though they appear intact, these medusahead seedheads from the canopy have been killed by the heat of a prescribed burn. This study by UC Davis found that burning before seed dehiscence is one of the most effective controls for medusahead in California's Central Valley region. Bobcat Ranch, Yolo County, California, June, 2003. (Photo by Joe DiTomaso, University of California, Davis)*

Woody species can be shrubs, vines, or trees. Most of the problematic invasive woody species resprout from the base when injured. These species are difficult to control with prescribed burning and often require very hot burns, multiple burns, or a combination of tools to be successful. Woody plants that do not resprout, such as junipers (*Juniperus* spp.), are generally easier to control with an intense fire than are other woody plants.

Reproduction

Long-term successful control of invasive species with prescribed burning requires suppression and depletion of all reproductive structures, both vegetative and sexual. For all species, including herbaceous annual, biennial, and perennial species, as well as woody invasives, it is critical to reduce the seedbank to a manageable level. This can be achieved by both depleting the existing soil seedbank and by preventing new seed production or off-site seed recruitment. Thus, it is important to understand the timing of seed maturation, mechanisms of new seed recruitment, and longevity of existing seeds in the soil. Understanding these components of invasive plant biology can not only determine the potential success of a prescribed burn program, but can also lead to more effective strategies using integrated control options. In addition, this information can dictate the length of time necessary for an effective management program.

For perennial species, both herbaceous and woody, effective burns must prevent vegetative recovery due to resprouting from the base of the stems or from root or underground stem structures. In the case of some herbaceous plants that have protected meristems close to the soil surface, including biennials in the first year or plants with basal rosettes, the burn temperature must be hot enough to damage these tissues and prevent vegetative recovery.

Annuals

To control annual species effectively, it is critical to either destroy the seeds with fire before they shatter (Allen 1995, Kan and Pollak 2000, Menke 1992) or to kill the plant before the seeds become viable (DiTomaso *et al.* 1999). Since seeds on the soil surface are not generally exposed to high enough temperatures to cause mortality in a grassland environment (Daubenmire 1968), burn timing is most effective after desirable species have dispersed their seeds, but when target invasive species have their seedheads directly exposed to the flames. Burn temperatures are considerably higher in a fine fuel canopy (500-900^{o}C) compared to temperatures at the soil surface 150-350^{o}C (DiTomaso *et al.* 1999).

EFFECT OF HEAT ON SEED SURVIVAL

Long-awned invasive grasses (e.g., medusahead, downy brome, ripgut brome (*Bromus diandrus*), red brome, barb goatgrass) rely on animal dispersal for long distance seed dissemination. Consequently, the seeds remain attached in the inflorescence longer than most desirable perennial and annual grasses. Medusahead matures at least a month later than most annual species, including grasses (Dahl and Tisdale 1975, Young *et al.* 1970). This directly exposes seeds to intense heat of fire flame when the senesced vegetation of other species or medusahead litter provides adequate fire fuel.

Even when exposed to direct flames, control of some annual grasses may be poor. This may be related to seed moisture content and the timing of the burn. For example, although most studies demonstrate over 90% medusahead reduction with a single prescribed burn (George 1992, McKell *et al.* 1962, Pollak and Kan 1996, DiTomaso *et al.* unpublished data), three annual burns near Alturas, California did not decrease medusahead (Young *et al.* 1972). These inconsistent results with burning were attributed to improper timing of the burn. It was suggested that medusahead seeds with moisture content below 30% are not effectively killed by fire exposure typical of a grassland burn (Young *et al.* 1972).

More recent results (Sweet, unpublished research in progress) indicate that seed moisture content in medusahead and barb goatgrass is not correlated with sensitivity to direct flame exposure. Rather, it is likely that fire moves more quickly through the dried vegetation later in the season and this reduces the time of direct exposure of inflorescences to the flames. This could account for the poor control when prescribed burns are conducted in late summer.

In many species it is still possible that burns conducted before seeds are fully cured and viable increases the embryo susceptibility to heating (Brooks 2001). Burning before seeds become viable is certainly the most effective timing and gives 100% reduction in germination. In addition, duration of heat exposure to the inflorescence can be increased by use of backing fires, which are directed into a low breeze and thus move more slowly.

Not all annual grass species respond in the same way to direct flame exposure. For example, ripgut brome seeds are more susceptible to direct flame than both medusahead and barb goatgrass. A one-second exposure to 400^{o}C dramatically reduced seed germination even when seeds were mature (10% seed moisture) (Sweet, unpublished data).

There are other methods of providing enough heat to kill mature seeds. Downy brome is difficult to control with a prescribed burn because its seeds begin to shatter shortly after the culms cure and before enough combustible fuel is available. Because most of the seeds are on the surface of the soil, considerable fuel is required to destroy the seed and necessary fire duration cannot be achieved by herbaceous plants. Only woody fuels can extend the duration of heating long enough to destroy seeds at the surface (Evans and Young 1987, Young and Evans 1978). Consequently, downy brome seeds can be destroyed when they are under a shrub canopy and not in the interspaces where there is only herbaceous fuels. Unfortunately, reinvasion of the areas under the shrubs is likely to be rapid from new downy brome seeds produced in the shrub interspace areas. A similar response was reported for red brome under and within the interspace of creosote (*Larrea tridentata*) (Brooks 2002).

The intensity of the burn can be manipulated in a number of ways, including delaying a burn until later in the season (Hansen 1986), conducting the burn in the late afternoon (McKell *et al.* 1962), using backing fires instead of headfires, and reducing the size of the burn parcel (D'Antonio *et al.* 2003). In addition, deferring grazing for a year can increase the fuel load and thus the heat of the burn (George 1992). In low productivity ecosystems it may take several years of litter and standing dead accumulation to produce sufficient fine fuel to carry an effective fire (DiTomaso *et al.* 2001). This can also be a problem when conducting multiple years burning where combustible fuel is unavailable in the second year burn because of the elimination of the litter fuel following the first year burn (Young *et al.* 1972).

CONTROL SUCCESS FOR ANNUAL GRASSES

In most examples, both published and anecdotal, prescribed burning has shown great success in the management of medusahead (Miller *et al.* 1999). One of the earliest reports by Furbush (1953) in California demonstrated good control for at least three years following a single year burn (June) of a medusahead infestation. At this timing the annual grasses had cured but medusahead seed was still in the milk to early dough stage. To achieve this level of success a complete burn is necessary. There was no germination of medusahead from seeds whose awns were consumed and the lemma tips charred, but germination was 87% from uncharred seeds although the fire had consumed the culm up to the seedhead (Sharp *et al.* 1957).

In a study conducted in Fresno and Yolo Counties, California, prescribed burning alone was a very effective tool for the control of medusahead. In Fresno County, a single year of burning reduced medusahead from 50% average cover to <1% the following year (99% reduction), and in Yolo County this reduction was from 79% cover in the year of the burn to 11% the following year (86% reduction) (DiTomaso *et al.*, unpublished data).

There are a few examples where burning for the control of medusahead has failed or achieved only partial success (Young *et al.* 1972, Ponzetti 1997, Youtie 1997, Youtie *et al.* 1998). In these cases the timing of the burn was too late and although most of the medusahead seeds were consumed or at least charred, a small percentage was unburned or survived the heat exposure.

Prescribed burning was also shown to be effective on barb goatgrass, however, a single year of burning was not sufficient even when the seeds were still in the inflorescence. In a central California burn conducted for one year in early summer, the cover of barb goatgrass was reduced by about 50%, but three years after the burn the infested acreage increased ten-fold (Hopkinson *et al.* 1999). In another study, barb goatgrass was burned two consecutive years in the central coastal foothills. The first burn was conducted in May 1997, a drought year (DiTomaso *et al.* 2001). Although the seedbank of barb goatgrass was dramatically reduced by the first year of burning, there was no effect on goatgrass cover. In the second year (1998) the same area was burned in July. This was an *El Niño* season and, although the timing was very different, the phenology of the noxious annual grass was similar to the previous burn timing. The effectiveness of the two-year burn regime was excellent. Barb goatgrass was not detected in the site in the spring and summer after the second year burn. It is important to note, however, that in a nearby site on predominantly serpentine soil, a second year burn was not complete and barb goatgrass was not effectively controlled in this area. Consequently, despite the completeness of the first year burn in all sites, it was the effectiveness of the second year burn that determined the success of the two-year burn program. Based on these results, the seedbank of barb goatgrass does not appear to survive more than two years.

Japanese brome has also been successfully controlled with prescribed burning. Unlike downy brome and other annual grasses, Japanese brome retains its seed in the inflorescence for an extended period. Using a single year of prescribed burn in March, the biomass of Japanese brome was reduced by 85% in the year following the burn (Whisenant *et al.* 1984). The suppression of Japanese brome was still at 50% in the second year after the burn. Similar results were also obtained in a South Dakota mixed grass prairie burned in either fall or spring (Gartner 1975, Gartner *et al.* 1979). Again, they found suppression of the grass in the second year following the burn.

Ripgut brome can also be controlled with a spring fire (Kyser and DiTomaso 2002). Unlike medusahead, barb goatgrass and Japanese brome, ripgut brome matures earlier and most prescribed burns in late spring and early summer are after the majority of the seeds have been disseminated. However, as previously discussed, the seeds of this species appear to be very sensitive to heat and only a one-second exposure to direct flames is sufficient to cause mortality.

Prescribed burns can be counterproductive. *The National Park Service temporarily suspended prescribed burns in lower elevation yellow pine forests of the southern Sierra Nevada because the burn prescription being used aided invasion by cheatgrass* (Bromus tectorum). *(Photo by John R. Matchett, US Geological Survey)*

CONTROL SUCCESS FOR ANNUAL FORBS

Of the major forbs weeds, more is known of the effect of prescribed burning on yellow starthistle than any other species in the western United States (DiTomaso *et al.* 1999, Hastings and DiTomaso 1996, Kyser and DiTomaso 2002, Miller 2003). The success of burning depends on proper timing. The best time for burning is usually in early to mid-summer (late June to July depending on the area) following seed dispersal and senescence of desirable grasses and forbs, but prior to viable seed production in yellow starthistle. The dried vegetation of other species can provide the fuel to carry the fire. Unfortunately, this timing is when the risk of escaped fires is very high.

Unlike most annual grasses, the seeds of yellow starthistle can survive for three or more years in the soil. Consequently, a single year of burning can reduce the seedbank by about 75%, but this will not be sufficient to reduce the infestation in most cases. In addition, yellow starthistle seed germination in the fall seems to be stimulated by a preceding burn (DiTomaso and Kyser, unpublished data). A second year of burning will further reduce the seedbank, but this too may not provide adequate control. Three consecutive years of burning were required to reduce the yellow starthistle seedbank by 99% and the summer cover by 91% (DiTomaso *et al.* 1999). In other studies, integrating a first year burn with a second year herbicide treatment was the most effective strategy (see Chapter 3 for further discussion).

EFFECT OF BURNING ON NON-TARGET SPECIES AND PLANT DIVERSITY

The timing of burns to control annual grasses or forbs can greatly influence the population of other non-target species. Species that complete their life cycle before the burns are conducted will generally be unharmed and selected for, while those that flower later and whose seeds mature later than the burn will often be negatively impacted. For most invasive annual species, late spring or early summer burns are the most effective. This timing is also most beneficial for native forb species, primarily because it is when the phenology of invasives and native species, fire intensity, and removal of the surface thatch are optimal (Meyer and Schiffman 1999). Burns conducted in the late summer or fall after all the grasses have senesced can favor native perennials, but may not provide effective control of the desired invasive annual species (Dyer and Rice 1997). Winter burns, after exotic grasses have emerged, can reduce the litter and thatch layer but may not increase native forb populations (Meyer and Schiffman 1999). Multiple-year burns can also impact the species composition, particularly in areas with uneven topography, as they will increase the potential for soil erosion, thus selecting for some species at the expense of others.

Interestingly, most studies show that few non-target plants respond negatively to prescribed summer burning and the ones that do are generally non-native species (DiTomaso *et al.* 1999, Hastings and DiTomaso 1996). For example, in Sonoma County, a prescribed burn program for the control of yellow starthistle also significantly reduced false brome (*Brachypodium distachyon*), ripgut brome, and soft brome (*Bromus hordeaceus*), all of which are non-native annual grasses. In another study, soft brome germination and seedling establishment were reduced when the thatch layer was removed by the preceding burn (Smith 1970). Of all the species monitored over a three consecutive year burn period, only 8% of the native species showed a decline. One native species that decreased in abundance in the burned sites was *Clarkia purpurea*, a late season native herbaceous flowering plant with a similar life cycle as yellow starthistle (DiTomaso *et al.* 1999).

Although most species that benefit from burns are desirable plants, in some cases invasive perennials can increase following fire. Fall or springs burns conducted in Sequoia National Park in central California achieved control of invasive annual grasses, but the noxious exotic annual forb Malta starthistle (*Centaurea melitensis*), which was not found in the pre-burn areas, was found in abundance in the post-burn sites (Parsons and DeBenedetti 1984, Parsons and Stohlgren 1989). Most annual grasses experience no or only transient effects from prescribed burning. For example, in Sonoma County, wild oat (*Avena fatua*), silver hairgrass (*Aira caryophyllea*) and little quakinggrass (*Briza minor*) increased after a three-year burn regime, but returned to their unburned control levels within one or two years of the last burn (Kyser and DiTomaso 2002).

Late spring or early summer burning has been shown to have a more pronounced positive effect on forb species, both native and non-native (DiTomaso *et al.* 1999). Three consecutive years of prescribed burning increased native forb cover in spring by nearly 400% in Sonoma County, boosting the native proportion of total forbs from 17% live canopy to 67% (DiTomaso *et al.* 1999). A number of individual species can benefit from fire, including the native forbs *Linanthus bicolor* and *Minuartia californica* (DiTomaso *et al.* 2001), but most often it is members of the Fabaceae (pea or bean family) or Geraniaceae (geranium or filaree family). In the same Sonoma County study site, the native legumes *Lotus wrangelianus*, *Lupinus nanus*, and *Trifolium gracilentum* increased their cover values by 250% to 2,300% after burning compared to adjacent unburned areas. In Mendocino County, control burns for barb goatgrass significantly increased the abundance and cover of three native legume species, *Trifolium bifidum*, *Astragalus gambelianus* and *Lotus humistratus* (DiTomaso *et al.* 2001). Other non-native forbs can also increase dramatically following warm season burns. The cover and abundance of nearly all species of *Erodium* are stimulated following late spring or summer burns (DiTomaso *et al.*, unpublished, Kyser and DiTomaso 1999, Murphy and Lusk 1961).

In addition to benefits to native legume species, the most important positive impact of prescribed burning for invasive weed control is the potential in-

Fuels impact fire characteristics. *Before test burns, US Geological Survey staff collect biomass to assess fuel loads and fuel moisture. Mono Lake, CA. (Photo by John R. Matchett, US Geological Survey)* crease in native perennial grasses. Burns designed to control barb goatgrass increased the native perennial grass *Hordeum brachyantherum* from 1% to 10% cover (DiTomaso *et al.* 2001). The spring South Dakota burns that controlled Japanese brome led to the increased production of western wheatgrass (*Agropyron smithii*), a native rhizomatous perennial (Gartner 1975). In Sonoma County, spring cover of purple needlegrass (*Nassella pulchra*) initially decreased following the first burn, but increased by three-fold after three consecutive years of burning. The decline following the first year burn was likely due to a reduction in the size of bunchgrass clumps rather than to clump mortality (DiTomaso *et al.* 1999). Other studies have also shown increases in purple needlegrass following grassland fires (Fossum 1990, Hatch *et al.* 1991).

In general, prescribed burns increase plant diversity and species richness, particularly of native plants. Most studies show that this is due to an increase in forbs. This increase has not been shown to be a response to invasive target species suppression. For example, a single burn in Sonoma County did not significantly reduce summer yellow starthistle cover, but dramatically increased plant diversity compared to the unburned site (DiTomaso *et al.* 1999). There have been several suggestions as to why native plants benefit from periodic burning in grasslands, but most center on the removal of the thatch layer (DiTomaso *et al.* 1999, Knapp and Seastedt 1986, Vogl 1974). Removal of thatch may affect several abiotic factors and can increase light penetration, soil temperature, and nutrient availability. Seeds of many native endemic forbs may require light exposure to germinate or higher light levels for seedlings to survive. In addition, increased solar radiation in a burned area warms the soil to a greater degree much earlier in the growing season and this could benefit the establishment of native species. Native seedlings (non-legume species) could use nutrients released by a prescribed burn to increase survivorship (DiTomaso *et al.* 1999). It is also possible that thatch (which is removed by burning) may provide an environment conducive to the survival of pathogens that could increase the mortality of native forbs, particularly in the seedling stage.

FOLLOW-UP PROGRAMS

Single-year burns have been shown to be effective with medusahead, but in most other cases do not provide effective control. Even when a reduction in the infestation is observed in the first year after the burn, recovery often occurs by the second or third year. For example, a one time burn for control of Japanese brome reduced density and seed production in the first growing season after the burn, but the density of the annual grass returned to pre-burn levels by the second growing season as the seedbank became fully replenished (Whisenant 1990, Whisenant and Uresk 1990).

Follow-up burns, grazing, or selective herbicide treatments can slow the recovery of exotic annual grasses and maintain forb cover (Bainbridge and D'Antonio 2003). The most critical aspect to the management of invasive annual grasses and forbs is to control the existing soil seedbank and prevent new seed recruitment. The goal of a successful follow-up program is to prevent escaped or isolated plants from completing their life cycle. In some cases where the seedbank is short-lived, a follow-up program may take only a couple of years, whereas in other cases it may require longer.

Biennials

Prescribed fire generally is not successful for controlling biennials. The life cycle of biennial plants means that they typically exist as uneven-aged stands. Only those plants that have bolted in the second year of growth are susceptible to fire mortality (Heitlinger 1975). One-year-old plants in the basal rosette stage have meristems protected from grassland fire damage and escape the fire to produce seed in the second year. However, by initiating the burn later in the spring it is possible to obtain better results provided the burn is conducted before the bolted plants have set and dispersed seed.

Similar to burns for annual species, the ultimate goal of a prescribed burn program for biennials is to control the existing seedbank and to prevent new seed set and off-site recruitment. To achieve this, multiple year burns can be used to eventually deplete the seedbank. This approach has proven successful with biennial sweetclover (*Melilotus* spp.) (Kline 1983, 1984). In even-aged stands of white sweetclover (*Melilotus albus*) and yellow sweetclover (*Melilotus officinalis*) in a midwestern prairie, two or more sequential burns were necessary for effective management (Cole 1991, Kline 1983, 1984). The first year burn in fall or spring scarified the seed and stimulated germination. The second year burn was conducted later in the season, after the plants had bolted but before seed production. When a well-developed seedbank is present, additional burns may be necessary.

More intense burns give better control of bolting plants. These burns can occur with increased thatch (Heitlinger 1975). In a study on a tallgrass prairie in Wisconsin, this approach reduced sweetclover from 91% in the unburned site to 5% in the treated area (Schwarzmeier 1984). Although reports indicate success controlling sweetclover with prescribed burning when infestations are even-aged, unevenly-aged stands are more difficult to control using this approach because first year plants escaped damage from the first burn and set seed before the second year burn, thus perpetuating the seedbank.

Prescribed burning has also been used to control garlic mustard (*Alliaria petiolata*) in the eastern United States. However, in areas where the thatch and litter layer were damp, plants resprouted and seedlings survived. Sequential burns under drier conditions were very effective and reduced garlic mustard to about 2% cover, whereas populations in unburned areas doubled every two years (Nuzzo *et al.* 1996). Although burning can suppress sweetclovers, garlic mustard and other biennial species, combining this strategy with timely herbicide treatments would likely prove to be even more effective (Martin and Parker 2003).

Perennials

GRASSES

In the western United States, prescribed burning for the control of invasive annual species can increase the frequency and abundance of perennial grasses, particularly native species. In the North Central and Midwestern states, burning can also be used as a tool to control cool season non-native perennial grasses in tall grass prairies previously dominated by warm season natives. Although a number of undesirable cool season perennial grasses may be susceptible to burning, the vast majority of studies have been conducted on Kentucky bluegrass, Canada bluegrass, and smooth brome.

A heavy thatch layer and proper burn timing are critical for selective control of cool season grasses and stimulation of warm season grasses. Optimal timing is when the tillers are elongating on the cool season grasses but the warm season natives are still dormant. This typically occurs from mid- to late spring. For Kentucky and Canada bluegrass, the best timing in a tall grass prairie was mid-April to early May when plants were 4-10 inches tall (Becker 1989, Curtis and Partch 1948, Engle and Bultsma 1984). However, this timing did not control quackgrass (*Elytrigia repens* [=*Agropyron repens*]) growing in the same area. Quackgrass tillers elongate later than Kentucky bluegrass and this was critical to the success of the burn. Thus, it is important to recognize that the optimal burn timing for one target may increase another invasive with a slightly different phenology (Murphy and Lusk 1961). Similarly, the response to burn timing was different for Kentucky bluegrass and smooth brome in an Illinois prairie. Kentucky bluegrass growth begins in early April and peaks by mid-May, whereas smooth brome does not begin its growth until mid-April and peaks in mid-July. A late April burn timing was optimal for Kentucky bluegrass control but because smooth brome tillers had not yet elongated into the fuel bed, its growing points were not as susceptible to damage and its population was only slightly reduced (Blankespoor 1987, Old 1969). Smooth brome is most susceptible when tiller elongation elevates the growing point to a height where it is directly exposed to fire. For smooth brome control, late spring (late May) to summer burns were more effective than mid-spring burns (Wendtland 1993, Willson 1992). Burning at the flowering stage can also suppress smooth brome.

The suppression of cool season perennial grasses by burning can result in an increase in warm season grasses (Robocker and Miller 1955). However, in the absence of warm season grasses the invasive cool season grasses would quickly re-infest the area. Consequently, most examples of successful control of cool season invasive perennials by burning occur in prairies where warm season natives were still a major component of the community. For example, properly timed fire suppressed smooth brome, but in the absence of warm season grass competition secondary tiller growth allowed the smooth brome to recover (Willson and Stubbendieck 1996). Similar results were also reported in a South Dakota prairie, where high soil moisture promoted native warm season grasses that suppressed smooth brome recovery after a burn

(Blankespoor and Larson 1994). Prescribed burning was compared in two Minnesota prairies, one with a mix of warm season natives and cool season invasive perennials grasses (Kentucky and Canada bluegrass) and the other dominated by Kentucky bluegrass with only a small component of native warm season grasses (Schacht and Stubbendieck 1985). In the site with very few warm season perennials, the cool season grasses were not suppressed compared to the site with a significant component of warm season grasses. Although most cool season perennial grasses are non-native, there are some native species that may also be susceptible to fire. For example, spring burns designed to control Kentucky bluegrass in South Dakota also injured the cool season native green needlegrass (*Stipa viridula*), but the burning increased warm season grasses for both years (Engle and Bultsma 1984).

Repeated annual burning of tall grass prairie greatly suppresses exotic species richness and abundance while stimulating the native warm season C_4 grasses (Smith and Knapp 1999, 2001). In a Nebraska prairie, five years of burning dramatically reduced Kentucky bluegrass cover but increased the cover of the native warm season grasses and forbs (Becker 1989). A similar result was reported in a Minnesota prairie, where burning was conducted every other year. Repeated burning provided excellent control of Kentucky bluegrass and stimulation of the warm season native perennials little bluestem (*Schizachyrium scoparium*) and big bluestem (*Andropogon gerardii*). As the population shifted more in favor of the less matted growth form of bluestems, the subsequent burns were hotter (Svedarsky *et al.* 1986). The benefits of this program were most evident nearly 10 years after starting the burning. The biennial burning regime was less expensive compared to annual burns and also provided better cover for ground nesting birds every other year between the burns.

Two burning regimes for the control of Kentucky bluegrass were compared in a Kansas prairie (Abrams 1988). In the unburned area, the canopy cover of Kentucky bluegrass was 30%. By comparison, the site burned twice in the first and fifth year of a six year period had 7% cover of Kentucky bluegrass, whereas the cool season perennial grass was completely eliminated in the site burned five consecutive years. In perhaps the most dramatic example, a burning program was sustained for 36 years in a Kansas tallgrass prairie except for one six-year suspension (McMurphy and Anderson 1965). The burn timing was just prior to the initiation of spring growth of the native warm season perennial little bluestem. This optimal timing nearly eliminated Kentucky bluegrass and increased the total basal cover of warm season natives, particularly little bluestem (Towne and Owensby 1984).

Invasives can change fire regimes. *Historic fire intervals in Great Basin sagebrush steppe communities have been shortened significantly by invasion of cheatgrass* (Bromus tectorum). *When the fire interval is shorter than the recovery time of sagebrush vegetation, habitat type conversion to annual grassland can occur. US Geological Survey and Bureau of Land Management personnel study fire behavior in test burns to learn more about natural fire regimes. Mono Lake, CA. (Photo by John R. Matchett, US Geological Survey)*

For smooth brome, single burns generally result in partial to full recovery. To maintain low smooth brome abundance, it is necessary to repeat burning at the tiller elongation stage in late spring, with late May being the optimal timing for the burn (Willson and Stubbendieck 1996).

FORBS

In most cases, successful control of invasive herbaceous perennial forbs using prescribed burn involves integration of other control options, particularly herbicide applications. Typically, controlled fires or wildfires promote invasive perennial forbs. As an exception, repeated spring burning in a mixed grass prairie in South Dakota reduced the invasive

Mediterranean native absinth wormwood (*Artemisia absinthium*) by 96% (Steuter 1988). In the southwestern states, a pricklypear cactus (*Opuntia* spp.) can be a rangeland problem. Although these species are not themselves combustible, they are easily damaged by high heat or direct flames. Adequate fuel is necessary for effective control, and the hotter the fire, the greater the control. Mortality of pricklypear cactus is usually fairly low in the first year after burning but increases in subsequent years (Ueckert *et al.* 1988). Fire is not often used for prickly pear cactus control since the damage to desirable forage is considered unacceptable.

Prescribed burning may have some effect on diffuse knapweed (*Centaurea diffusa*) populations. While fire does not control these plants directly, the seeds are retained in the flowerhead long into the season and are, therefore, exposed to direct heat from the flames of the burn (Renney and Hughes 1969). In contrast, spotted knapweed (*Centaurea maculosa* [=*C. biebersteinii*, *C. stoebe*]) seeds are dispersed soon after they mature and neither spring nor fall burns have been successful for the control of this species (Emery and Gross 2003).

Although most studies show little to no success with prescribed fires on invasive herbaceous perennial forbs, burn timing may be critical to their management. For example, repeated spring burns in May to June provided some level of control of Canada thistle (*Cirsium arvense*) in Illinois, but burns conducted earlier in the spring or later in the summer stimulated sprouting and increased the infestation of the invasive weed (Hutchison 1992, Morghan *et al.* 2000, Thompson and Shay 1989). Even in this example, adequate control was not achieved with burning alone, but the frequently burned plots were more resistant to reinvasion (Morghan *et al.* 2000). Prescribed burning has been attempted to manage several other species, including leafy spurge (*Euphorbia esula*), Dalmatian toadflax (*Linaria dalmatica* [=*L. genistifolia*]), and sulfur cinquefoil (*Potentilla recta*). In all these cases, regardless of the timing of the burn, control was unsuccessful and the invasive species generally increased in abundance (Jacobs and Sheley 2003a,b, Lesica and Martin 2003, Wolters *et al.* 1994).

Some species resprout after fire. *Fire-adapted woody species, like the native chaparral shrub chamise* (Adenostoma fasciculatum), *can resprout from the base following a burn. Invasive woody species like broom can also resprout. McLaughlin Reserve in the northern Coast Range of California, Spring, 2005. (Photo by Guy Kyser, University of California, Davis)*

Woody Species

Most woody species are difficult to control with a prescribed burn. Some of the most problematic species, such as Japanese honeysuckle (*Lonicera japonica*), tree-of-heaven (*Ailanthus altissima*), Russian-olive (*Elaeagnus angustifolia*), and saltcedar (*Tamarix ramosissima*) are favored by fire because they readily resprout from the base following mechanical or fire damage. Other major invasive woody species, including sweetbriar rose (*Rosa eglanteria*), Himalaya blackberry (*Rubus armeniacus* [=*R. discolor*]), cutleaf blackberry (*Rubus laciniata*), English hawthorn (*Crataegus monogyna*), and common pear (*Pyrus communis*) also tend to increase following fire (Pendergrass *et al.* 1988). However, other shrub and tree species can be controlled using prescribed burns.

In eastern and mid-Atlantic states, prescribed burning for woody invasive plant control is conducted during the dormant season, but it is generally unsuccessful because of profuse resprouting from the rootstocks. It is likely that growing season burns would be more effective since non-structural carbohydrate reserves are lowest at this time (Richburg *et al.* 2001, Richburg and Patterson 2003). This would potentially deplete the energy reserves of the plants root system, making it more susceptible to subsequent burns or other control options.

Vines are rarely controlled with prescribed burning, but control efforts on invasive shrubs have received some attention. Absinth wormwood can grow as both an herbaceous perennial and a subshrub. As discussed under the perennial forb section, repeated burning was very effective on this species, as the new buds develop fairly close to the soil surface and can be killed by an intense fire (Steuter 1988).

In California, prescribed burning for the control of shrubs is most widely used on broom species in the Fabaceae (pea or bean family), particularly French broom (*Genista monspessulana*) and Scotch broom (*Cytisus scoparius*). Like most other members of the Fabaceae, they have long-lived persistent seedbanks. In addition, their seed coats are scarified by fire, which stimulates germination in the following season. Consequently, any successful management strategy must be long-term and consider integrating methods that can deplete the soil seedbank (Swezy and Odion 1997).

Some strategies take advantage of the stimulation in seed germination following fire. For example, the above ground growth of the plant can be cut in summer or fall and allowed to dry on site. Once the cut materials have dried they can be burned. In one case with French broom (Odion and Hausbensak 1997), burns were conducted in October. The first year fire was intense because of the high density of dried French broom and this burn stimulated broom germination during the next rainy season. The cut and burn strategy was repeated a second time. This second burn provided excellent control of the young seedlings. This strategy would reduce the seedbank and the dense stand of French broom, but continued control efforts would be necessary to effectively manage the infestation. Continued control of mature plants would eventually deplete the seedbank since it takes two or three years before the new plants can set seed.

Another approach to controlling French broom used a fall cutting followed by an intense burn of the dried stems the following May (Boyd 1996). The burn was hot enough to kill the mature French broom rootstalks and prevent resprouting. The fire stimulated seed germination the next fall. In November following the burn, two non-native annual grasses (soft brome and rattail fescue (*Vulpia myuros*)) were broadcast seeded into the infested burn site. These winter annual grasses provided fine fuels for a subsequent burn in July. This summer burn killed the broom seedlings.

Although repeated burns were used in both of these examples, the objective of each burn was similar. The initial burn allowed a higher percentage of broom seed to germinate than would occur without the burn. Consequently, the depletion of the seedbank could be accelerated. However, it would likely require several more years of follow-up control efforts to assure near elimination of the French broom infestation at each of these sites.

In addition to using prescribed burning for the control of shrubs, fire can also be a tool to control certain tree species. In Alabama, repeated burning under drought conditions effectively eliminated European privet (*Ligustrum vulgare*) and Chinese privet (*Ligustrum sinense*) (Batcher 2000). Chinese tallow (*Sapium sebiferum*) is a fire-adapted tree that is difficult to control with fire. However, burns conducted during the growing season on smaller trees at lower density reduced resprouting and prevented its dominance in a southern coastal prairie (Grace 1998, Grace *et al.* 2001). With frequently repeated burns and high fine fuels it was possible to kill even larger trees.

In rangeland, prescribed burning is occasionally used for long-term suppression of native woody species, including big sagebrush (*Artemisia tridentata*), false broomweed (*Ericameria austrotexana*), broom snakeweeds (*Gutierrezia* spp.) (Bunting 1994, Mayeux and Hamilton 1988, McDaniel *et al.* 1997, Senft 1983), junipers (Miller and Tausch 2001, Mitchell *et al.* 2000), and mesquite (*Prosopis* spp.) (DiTomaso 2000). Although these species are often desirable and natural components of ecosystems, they can encroach into other grassland ecosystems with long-term fire suppression (Miller and Tausch 2001).

Western juniper (*Juniperus occidentalis*) and mesquite (*Prosopis* spp.) are two species that are native to the southwestern United States but have encroached into rangelands. In the absence of natural fires, juniper has spread into sagebrush habitats. This species is a basal sprouting, multistemmed evergreen tree growing on rocky slopes with shallow soils. It is native to Texas, Oklahoma and New Mexico. Western juniper does not resprout from the roots so it is susceptible to top kill by mechanical methods of burning. Typically, fine grass fuels are not sufficient to kill large trees, but when there is enough fuel present to carry fire, small controlled burns can kill young trees. It is also possible to control juniper infestations using a propane torch (Lile *et al.* 2004). However, this technique is very cumbersome on rangeland sites and is not very practical on slopes over 20%. Other species of juniper can also be controlled with prescribed fire, including redberry juniper (*Juniperus pinchotii*) (Mitchell *et al.* 2000).

Mesquite are susceptible to prescribed burning when plants are young (<4 years old), but older plants are much more difficult to kill.

Fuel loads, site and weather conditions affect burns. *This patchy burn in the Bald Hills of Redwood National and State Parks in California reduced cover of Scotch broom and stimulated the soil seed bank, but left significant areas unburned. (Photo by Tim Bradley, National Park Service)*

Data Gaps

While some invasive species can be controlled using timely prescribed burning, there are few data on a host of other important invasive annuals, perennials, or woody species. The characteristics of their life cycle and biology can provide clues as to whether they may be good targets for a control burn program. However, the very basic biological information found in the literature may not be reliable enough to predict the success or failure of prescribed burning on a particular species. For example, based on the life cycle of ripgut brome, which is similar to most early season winter annual grasses, it would be predicted that late spring control burns would be ineffective, yet a number of studies have shown excellent control of ripgut brome with a later burn timing (DiTomaso *et al.* 1999, Kyser and DiTomaso 2002). Specific information, such as the sensitivity of mature seed to heat, can greatly assist in predicting impacts of fire on invasive species.

Probably the biggest gap in our understanding of the ecosystem effects of prescribed burning is the impact it has on native vegetation. Only a few perennial grasses and legumes species have been studied in any detail. There is much to know on the effects of burning on other native and non-native species, both short and long-term responses to single fire events or repeated burning. It is particularly important to understand the impact of burning on threatened and endangered plant species. Understanding their response to fire can assist in the decision making process on managing invasive species.

Burn programs designed to control invasive species often show variable results. due to climatic variability, soil type, topography, fire extent or size, burn timing or seasonality, community structure, fuel loads and properties (i.e., both standing litter, thatch, and other sources), or intensity and duration of the burn (e.g., head or back fires, fuel type). Most of these factors are not taken into consideration when developing a prescription burn program and little information exists as to their influence on the efficacy of a burn. In addition, more information needs to be accumulated at the time of the burn, including flame length, humidity and temperature, burn speed and many other characteristics of the fire. This information can help explain the success or failure of the program and greatly help in interpreting control data.

One of the key aspects in the long-term development of a control program, regardless of whether fire is involved, is the seedbank longevity of both the target and non-target species. Although some studies have considered this factor for target invasive species, few, if any, consider the impact of burning on the seedbank of desirable species in the treatment site. This is a key gap in the development of long-term management programs. It is particularly important when burning is used as the sole strategy in a management program, since fire may either stimulate germination or increase seed mortality.

CHAPTER 3: Using Prescribed Burning in Integrated Strategies

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Although repeated burning has been shown to be very effective for the control of several invasive plant species, there are many occasions where it is prohibited or impractical. Since few invasive weeds are effectively managed by a single year of prescribed burning, it is often necessary to incorporate other control options into a long-term management strategy (Kyser and DiTomaso 2002). These methods can include mechanical, cultural, biological and chemical options.

Even when multiple-year burnings are possible, such an approach may not be the most appropriate strategy for a particular ecosystem. For example, repeated burning may create a new shorter fire interval that could be detrimental to endemic or desirable species. In the case of tallgrass prairie, repeated burning suppressed exotic weeds and stimulated native warm season grasses (Smith and Knapp 1999, 2001). However, other ecosystems, including grasslands, did not evolve with as short a fire frequency interval. The creation of unnatural fire regimes can establish opportunities for other species to colonize or expand their cover in sites they could not previously dominate (Brooks *et al.* 2004). Increased fire frequency can also affect other ecosystem properties, including the rate of soil erosion and formation, and the patterns of nutrient cycling. As a result, this alteration in the disturbance regime can lead to the selection of plants and animals with different life forms and histories than the natural composition (Cowling 1987).

While it is desirable to create systems that are similar to their natural state, there are situations where it is simply not possible to restore a community to its pre-invasion or pre-disturbance condition. For example, many fire-promoting invasive tropical grasses from Central America and Africa dominate seasonally dry habitats in the Hawaiian Islands. The shorter fire regime has led to a complete loss of native forest in some regions (D'Antonio *et al.* 2000). In these areas it is not possible to restore the native plant assemblage. Thus, restoration programs are creating new native plant communities that are fire tolerant and can co-exist with native grasses (Tunison *et al.* 2001). In other situations, non-native desirable species, not considered significant invaders, can be introduced to an area to outcompete more detrimental non-native species. For example, *Agropyron desertorum*, a non-native perennial grass, was seeded into a post-fire rangeland in the Great Basin desert of North America to suppress the more noxious annual cheatgrass and reduce fuel continuity and flammability (Hull and Stewart 1948).

In many areas, prescribed burning is not permitted at all, despite its effectiveness in controlling a particular invasive species in that site. However, these sites can experience periodic wildfires. Such a wildfire can be very beneficial to the management of that species, particularly if the timing of the fire is ideal for suppression of the invasive plant. In addition, the scale of these fires is generally larger than

Post-burn seeding can play a role. *As part of an integrated management study for control of medusahead, researchers follow a burn with a rangeland drill seeder to plant native cover. Bobcat Ranch, Yolo County, California, Fall, 2003. (Photo by Joe DiTomaso, University of California, Davis)*

prescribed burns. To take advantage of such a situation, other control options can be used later in the season or the following season. This can provide far greater and more economical control than a management approach in unburned areas. For example, after an August wildfire in a squarrose knapweed (*Centaurea triumfettii* [=*C. squarrosa* or *C. virgata* var. *squarrosa*]) infested site in Utah, the site was treated with a combination of picloram and 2,4-D herbicides the following fall. Nearly three years after the herbicide treatment, squarrose knapweed control was 98 to 100%, but control in an adjacent unburned site was only 7 to 20% (Dewey *et al.* 2000).

Situations Where Integrated Approaches Can Be Used

There are a number of situations where integrated approaches for invasive weed control can be used more effectively, economically, and practically than using a single control option over several years. This is particularly true when infested areas consist of complexes with both desirable and undesirable species. The examples presented here include integrated approaches that incorporate prescribed burning as part of a program.

PRE-TREATMENT BURNING

In many cases where fire alone does not provide sufficient control, it can be used to dramatically enhance the effectiveness of other techniques, usually mechanical or chemical control options. Numerous published examples illustrate the benefits of this strategy. This generally applies to invasive perennials such as French broom, Scotch broom,

Mechanical control can complement burning. *Tractor-drawn mowing and cutting tools such as the rotary deck mower and flail mower (top two photos) may be used to knock down standing vegetation, increasing the availability of dry fuel. Tillage tools like the disc and harrow (bottom two photos) suppress vegetation and reduce litter on the soil surface, resulting in a cooler fire. (Top two photos by Guy Kyser; lower photos by Joe DiTomaso, University of California, Davis)* gorse (*Ulex europaea*), saltcedar, leafy spurge, giant reed (*Arundo donax*), and perennial pepperweed (*Lepidium latifolium*).

An initial burn in a management program can stimulate seed germination, thus depleting the seedbank more rapidly than using mechanical, cultural, or chemical control. With a first year burn, yellow starthistle germination the following fall increased dramatically. These seedlings were subsequently killed with a clopyralid herbicide treatment during late winter or early spring. The increase in germination depleted the seedbank about one year faster that would be achieved with only chemical treatments (DiTomaso *et al.*, unpublished data). A similar approach was used for the control of the invasive Lehmann lovegrass (*Eragrostis lehmanniana*), where the seedbank was rapidly depleted when an initial burn stimulated germination and a follow-up herbicide treatment killed the young plants and the surviving adult lovegrass plants (Biedenbender *et al.* 1995). The stimulation in germination in both these species may be due to removal of the litter layer. This has been shown to increase solar radiation earlier in the season, which increases both light exposure and soil temperatures (DiTomaso *et al.* 1999).

Burning stimulates germination of brooms and other legume seeds, including gorse. In contrast to yellow starthistle and lovegrass, this may be the direct result of seed scarification by the fire. Since new plants can take two or more years to reach reproductive maturity, follow-up treatments with herbicides or fire can not only deplete the seedbank, but can also prevent new seed development (Boyd 1996).

More commonly, however, prescribed burning is used to improve access to an area with a high density of an invasive species. Infestations of saltcedar and other *Tamarix* species are often so dense as to prohibit ground applications of an herbicide or mechanical entry. Burns by themselves are ineffective and can harm native vegetation in mixed stands. However, in solid stands of saltcedar, where aerial herbicide application is prohibited, a prescribed burn or pile burning after mechanical cutting can remove the biomass and provide access for a secondary treatment (Egan 1999, Friederici 1995, Taylor and McDaniel 1998). The timing of this burn may depend upon bird populations in the area. For example, summer burns would be ideal for development of crown fires, but this can overlap with the nesting season. As a result, burns are often conducted in early fall when temperature and humidity conditions are still satisfactory for the development of crown fires. Access to recovering plants is excellent and mechanical removal or treatment with herbicides is much easier following a burn (Racher and Britton 2003, Turner 1974). Removal of the thatch or litter layer can also provide better visibility of the area for a follow-up mechanical control method. This was reported with wild parsnip (*Pastinaca sativa*) where early spring burning eliminated the litter and revealed the location of the recovering rosettes, which were subsequently removed by mechanical means (Eckardt 1987).

Litter or thatch removal can also improve deposition of an herbicide on the target surface. With pre-emergence herbicides, application to bare ground can increase penetration into the soil profile allowing more of the compound to contact the root zone (Winter 1993). This is more effective with the control of annual species. With perennials, burning can remove the biomass, thatch, and older plant tissues. The recovering vegetation is more succulent with a less developed waxy cuticle. Not only is herbicide deposition better on these tissues, but uptake is less restricted.

As an example, a control program for fennel (*Foeniculum vulgare*) on Santa Cruz Island, California, integrated a first year burn with a subsequently triclopyr treatment to the recovering plants. The burn alone did not reduce the fennel population, but the incorporation of the two techniques nearly eliminated fennel from the treatment site and significantly increased the cover and diversity of native species (Klinger and Brenton 2000). In another example, a tall fescue infestation was burned in spring and glyphosate or imazapic was applied one to two months later to recovering plants. This integrated approach nearly eliminated tall fescue in the imazapic treated sites and reduced tall fescue to 6% cover in the glyphosate treated areas. In contrast, tall fescue cover averaged about 90% in the burn only sites (Washburn *et al.* 1999).

Removal of litter also enhances the effectiveness of herbicides for the control of vine-like species. Winter or early spring burning was used as a pretreatment for the control of Japanese honeysuckle and kudzu (*Pueraria montana* var. *lobata*)

(Brender 1961, Everest *et al.* 1991, Moorhead and Johnson 2002, Shipman 1962). Following removal of the canopy litter, herbicides were applied to the young resprouting vegetation. This increased contact with young leaves gave good control of these invasive species.

With some herbicides, particularly imazapic, a thatch or litter layer near the soil surface can tie up the herbicide and dramatically reduce its activity. This has been observed with medusahead, downy brome, and other noxious annual grass control, where imazapic used alone was ineffective but following a prescribed fire gave excellent control (DiTomaso *et al.*, unpublished data, Washburn *et al.* 1999). Even with management of the perennial tall fescue, control was only 37% in the burn-only areas, 60% in the herbicide-only plots, but control was 83% in the areas with the combination of burning and imazapic (Rhoades *et al.* 2002, Washburn *et al.* 2002). In other cases, areas previously burned required a much lower herbicide rate because of the reduced thatch layer (Vollmer *et al.*, personal comm.).

POST-TREATMENT BURNING

Burning is not always the most effective pre-treatment in an integrated management plan. In some cases, other control methods can be used before conducting prescribed burns. For example, a fall application of picloram followed by a spring burn gave better control of leafy spurge in North Dakota than herbicides or burning alone (Wolters *et al.* 1994). Even though burning has not been shown to be an effective control strategy for leafy spurge, the burn was demonstrated to significantly lower its germination.

In areas where the invasive plant infestation is so heavy that little fuel is available to carry an effective fire, an herbicide pre-treatment can provide ample dried biomass (Glass 1991). In a heavily infested yellow starthistle site, adequate fuel may not be available at the ideal burn timing. An herbicide treatment in the previous year can increase the grass population and facilitate a complete burn in the second year. However, when using this strategy, it is important to employ a third year control option since burning can stimulate the germination of yellow starthistle (DiTomaso *et al.*, unpublished data).

A similar strategy can apply to woody species. Treating gorse with an herbicide desiccated the living tissues and increased flammability of the site, which in turn, improved control (Rolston and Talbot 1980). As another example, Chinese tallow tree (*Sapium sebiferum*) can invade prairie ecosystems in the southern states. Mature stands suppress understory surface fuels by shading out the existing vegetation. Removing these stands by mechanical or chemical methods can increase the surface fuels and allow the re-introduction of prescribed burning. Periodic prescribed burning in these systems can stimulate native prairie vegetation and limit Chinese tallow tree encroachment (Grace 1998).

Prescribed fire can also be used after mechanical or chemical methods to remove the dead biomass and stimulate recovery or revegetation of infested site with more desirable species. Such an approach can be effective for management of saltcedar or giant reed (Bell 1997, Dudley 2003). In the southwestern United States large mesquite (*Prosopis* spp.) trees can be very difficult to kill with herbicides or burning (Queensland Government 2004). However, the top vegetation can be killed by herbicides and a subsequently burn can produce intense fires hot enough to kill the root systems.

In some cases, prescribed burning can be used either in combination with other techniques or between two herbicide treatments. Common reed (*Phragmites australis*) was effectively controlled when plants were herbicide treated, then burned and treated again to resprouting stems (Clark 1998). The level of control was considered much better than infestations only treated with herbicides. French broom canopy cover was reduced from 87% to less than 1% when plants were treated with the herbicide triclopyr, then cut and burned a month later, then treated again with glyphosate for two years to control the germinated seedlings (Bossard 2000).

MULTIPLE SPECIES COMPLEXES

In some situations, prescribed burning used to control one invasive species can select for another invasive species or for objectionable native species. For example, burning to control an annual grass can select for a perennial forb. For ranchers, a possible concern may be the selection for unpalatable native species, such as tarweeds (*Hemizonia* spp.), when relying on only prescribed burning as a management tool. Using integrated approaches, it is possible to more effectively select for a more desirable community complex.

As an example, repeated burning alone proved very effective for the management of yellow starthistle (DiTomaso *et al.* 1999) and medusahead (DiTomaso *et al.* 2005). However, burns can dramatically increase the population of non-native filarees, which reduce the quantity of desirable annual grass forage in rangelands. Integration with herbicide control can prevent a single species from dominating these areas and provide better quality and quantity of forage.

BURNING TO DECREASE DEPENDENCE ON HERBICIDES

Herbicides are the most widely used method of weed control in crops and in many non-crop areas. Although herbicides are effective for the control of noxious range weeds, they seldom provide long-term control of weeds when used alone (Bussan and Dyer 1999). Continuous use of herbicides can create environmental problems, including off-site chemical movement via surface water or drift, selection for other tolerant undesirable invasive species, selection for resistance in the target weed, injury to desirable native plants and reduction in plant diversity, changes in nutrient balance that decrease the total vigor of the range, and others.

Integrating management tools can reduce the dependence on herbicides. Under ideal situations, integration with other tools can increase the efficacy of herbicides or even reduce the number of years necessary to achieve adequate control. For example, effective yellow starthistle control typically requires three years of prescribed burning or clopyralid treatment when either method is used alone. However, a similar level of management can be accomplished in only two years when a prescribed burn is conducted in the summer of the first year and clopyralid is applied the following winter or early spring.

BURNING TO PREPARE FOR REVEGETATION PROGRAMS

Removing thatch and suppressing invasive species can greatly facilitate the establishment of native species. In tallgrass prairies, thatch removal with burning increases solar heating of the soil, which stimulates the early growth and vigor of warm season grasses (Ehrenreich 1959). In combination with the reduction in cool season grasses, this can result in rapid site conversion.

Similar results have occurred with the stimulation of native legumes and perennial grasses in burns designed to control yellow starthistle (DiTomaso *et al.* 1999). In grasslands, burning invasive annual grasses can reduce early season competition long enough to allow establishment of reseeded desirable species (Goodrich and Rooks 1999). These successfully established species may increase competition for resources and prevent dominance of the invasive grasses.

Prescribed burns can enhance effectiveness of herbicides. *Pre-emergent herbicides can be more effectively applied to soil when thatch has been burned. Here an ATV-mounted boom sprayer applies imazapic following a summer burn for control of medusahead* (Taeniatherum caput-medusae). *The burn not only prevented new seed recruitment, but it cleared the ground for chemical treatment. Flemming Ranch, Fresno County, CA. (Photo by Joe DiTomaso, University of California, Davis)*

In saltcedar infested riparian areas, the heavy accumulation of ground litter prohibits the establishment of desirable native plants even when the invasive species is controlled with herbicides. Integrating a prescribed burn treatment can open the soil surface and encourage the recovery and restoration of natives (Taylor and McDaniel 1998).

Revegetation programs do not necessarily need to be incorporated into a management program as a weed suppression method. In areas prone to erosion, reseeding after burns can be used primarily to prevent soil erosion during the rainy season. Usually this requires fast growing species that can become well established within a year of planting. In Nebraska, spring burning to remove the abundant litter layer was followed by drill seeding of desirable tallgrass prairie grasses. In conjunction with herbicide treatments to control leafy spurge, this integrated approach promoted the establishment of warm season perennial grasses (Masters 1992, Masters and Nissen 1998, Masters *et al.* 1996).

WORKING AROUND RESTRICTIONS ON BURNING

As described in Chapter 1, the ability of land managers to conduct repeated burns is limited, and they may be fortunate to burn an infested area even once. The timing of this burn, both seasonally and temporally, may be critical to the success of the management program. Necessity requires managers to use integrated approaches that can include prescribed burning.

BURNING TO ENHANCE THE EFFICACY OF BIOCONTROL AGENTS

Prescribed burning in spring or summer can kill biological control agents, particularly those whose larvae are feeding within the seedheads. This includes all the agents released for the control of yellow starthistle and many of the agents specific to other knapweed species. However, establishment of the leafy spurge flea beetle (*Aphthona nigriscutis*) was 230% more successful in burned plots compared to unburned plots (Fellows and Newton 1999). At the time of the burn, in either mid-May or mid-October, the adult insects were not active and the juveniles were below ground during the burn. The enhanced establishment of the insects was attributed to the increase in colonization in the bare ground of the burn plots. In the case of the spring burn, the insects had emerged after the leafy spurge plants had resprouted.

Other insects, including the klamathweed (St. Johnswort) beetle (*Chrysolina quadrigemina*) declined dramatically as a result of prescribed burns (Briese 1996). However, the population of this biocontrol agent rebounded quickly, primarily through influx of the beetles from outside the burn site. The increased available nitrogen taken up by klamathweed (=common St. Johnswort; *Hypericum perforatum*) in the burn site also benefited the insect populations. Similar recover of biocontrol agents have been observed in grassland areas burned for the control of yellow starthistle (*Centaurea solstitialis*) (Pitcairn, unpublished data).

AVOIDING ENVIRONMENTAL PROBLEMS ASSOCIATED WITH MULTIPLE BURNS

Disturbance regimes affect ecosystem properties such as rates of soil erosion and formation, and

Fire can enhance effectiveness of insect biocontrol agents. Larinus planus (Fabricius), *the Canada thistle bud weevil, is shown on Canada thistle* (Cirsium arvense)*. (Photo by Eric Coombs, Oregon Department of Agriculture, from insectimages.org)*

pathways and temporal patterns of nutrient cycling and energy flow (Cowling 1987). Continued disturbance, such as repeated burning, can result in long periods of bare ground exposure and increase the risk of soil erosion, especially in areas with uneven topography (Brooks *et al.* 2004).

As was previously discussed, multiple year burning can have a negative impact on the population of some plants, including desirable native species, as well as animals populations (DiTomaso 1997). Species, both native and invasive, which complete their life cycle before the burn, will be selected for, while those with later flowering times will be selected against.

In some areas, burning can lead to rapid invasion by other undesirable species with wind-dispersed seeds, particularly members of the Asteraceae (sunflower family). Integrating control methods may be necessary to minimize these potential problems while providing effective invasive weed management.

MANIPULATING FIRE CHARACTERISTICS

Fire intensity can be manipulated with a pre-treatment method that increases or decreases the combustible fuel loads. For example, grazing prior to a burn can create a cooler burn (D'Antonio *et al.* 2003). In contrast, deferring or restricting grazing prior to the burn will increase fuel accumulation and fire intensity (George 1992). The fire intensity can impact the level of control, as well as the ability to contain the burn. With large perennial or woody species, such as giant reed, saltcedar, gorse, and the brooms species, mechanical or chemical treatments a few months before the burn can increase the amount of dried biomass. This will dramatically increase the intensity of the burn and provide much better control of root crown resprouting compared to a cooler burn.

Tools Used in an Integrated Control Strategy

Although much of the discussion on the tools used in an integrated approach with prescribed burning focuses on mechanical cutting and chemical applications, there are a number of other options that may be practical and effective. These can be other mechanical methods, several cultural control options, biological control, and chemical application techniques. Each of these techniques is briefly discussed with particular reference to how they might be used with prescribed burning.

MECHANICAL CONTROL

Mechanical methods of control for invasive species consists of two approaches: removing or damaging above-ground tissues, including stems and new shoots, and removing or desiccating below-ground structures, primarily vegetative reproductive structures (roots, root crowns, rhizomes, bulbs, etc.). All of the below-ground control methods also remove or destroy above-ground tissues.

There are several ways to remove above-ground plant tissues, including hand-pulling, mowing, shredding, roller chopping, clipping (wood cutting), and chaining. Hand-pulling can also remove below-ground roots and is not normally used as an intensive control strategy with prescribed burning, except to remove plants that have escaped the burn, or as a follow-up program after the invasive population has been significantly reduced. It is more effective for the control of annuals and can be very labor intensive when targeting perennial species.

Mowing, shredding, roller chopping, clipping, and chaining remove primarily the above-ground vegetation. In some cases, they can provide excellent control of an invasive species. For example, mowing can be a very effective management tool for yellow starthistle control when used just prior to flowering and on plants with a high branching pattern (Benefield *et al.* 1999). In many cases, however, these techniques may not, by themselves, provide control of the invasive species, but can enhance the effectiveness of burning by increasing the amount of dried fuels in grasslands or shrublands. They also have the benefit of preventing seed production and reducing carbohydrate reserves in the target species.

For control of shrubs or trees, mechanical methods can include chaining, roller chopping, wood cutting, chaining, and shredding (Cross and Wiedemann 1985, McHenry and Murphy 1985, Rasmussen 1991). The use of these tools is somewhat limited, as they can only be used in areas with relatively gentle terrain (DiTomaso 2000).

Below-ground mechanical control techniques cut, expose, or remove the vegetative reproductive structures so that they do not resprout and recover. These methods include the use of hand-pulling, hand-held equipment (e.g., hoe, shovel, pick, etc.), grubbing, bulldozers, or tillage equipment (e.g., disk, harrow, knives, etc.). These techniques also injure or remove the above-ground tissues. Bulldozing can be used to kill woody species or large herbaceous plants, such as giant reed, and the dried biomass can be subsequently burned. In addition, bulldozers are often used to remove tree stumps that are capable of resprouting (McHenry and Murphy 1985). Most of these techniques can be used independent of prescribed burning. Tillage is mainly used to control annual species and can reduce the amount of soil surface litter, resulting in a cooler burn when the two techniques are used in combination. Like the tools used for above-ground control, this strategy is generally restricted to fairly gentle terrain accessible to these types of equipment.

CULTURAL CONTROL

Cultural weed control methods are commonly used in agricultural systems. They can include manipulation of fertilizer type and placement, irrigation regimes, row spacing, solarization, mulching, cover crops, intercropping, and many other techniques. In non-crop areas, cultural control methods can include prevention techniques, such as cleaning equipment to prevent seed or plant fragment movement, using weed-free hay, monitoring sensitive sites to detect new infestations, and implementing educational programs to identify invasive species and other important components of the associated ecosystem.

Under some circumstances, the use of nutrient spray carriers, water manipulation, solarization, shading, or mulches can be part of an integrated weed management program in non-crop areas. However, these techniques are rarely used outside of agriculture. The addition of a liquid fertilizer carrier to an herbicide spray solution can provide good weed control, while simultaneously increasing the growth of non-target species, particularly annuals. This will increase fuel loads that can increase the probability of a more complete burn later in the season. Using mulch, either as hay or as living mulch through the planting of ephemeral annual crop grasses (barley or wheat) can also provide fuel for a subsequent burn once the living mulch has cured. This was used for the management of yellow starthistle at Pinnacles National Monument in California (Martin and Martin 1999). In this case, a post-burn seeding of sterile wheat was used to supplement fine fuels to carry a second season burn.

As has been discussed, grazing can be used to manipulate fuel load in a prescribed burn. Grazing can also be used to manage some invasive species, including yellow starthistle, knapweeds, and invasive annual grasses (Olson 1999). The success of grazing can depend up the type of strategy employed and its timing. The ideal time to graze is when the noxious species is most susceptible to defoliation or when the impact on the desirable vegetation is minimal (Kennett *et al.* 1992). Three grazing strategies can be used to manage invasive weeds: (1) moderate grazing levels, to minimize the physiological impact on native plants and to reduce soil disturbance; (2) intensive grazing to counteract inherent dietary preferences of the grazer, resulting in equal impacts on all forage species including weeds; and (3) multi-species grazing which distributes the impact of livestock grazing more uniformly among desirable and undesirable species (Olson 1999).

Establishing more desirable and competitive plant species is the best long-term sustainable method to suppress weed invasions, establishment, or dominance, while providing high forage production (Borman *et al.* 1991, Lym and Tober 1997). One of the most common cultural methods integrated with prescribed burning is a reseeding or revegetation program. Revegetation is usually conducted after a burn, since the burn can recycle nutrients, provide weed suppression, eliminate thatch, increase light availability at the soil surface, and increase solar soil heating. Revegetation techniques include broadcast seeding, drill seeding, or plug planting, but the most economical and successful of these is drill seeding (Jacobs *et al.* 1999).

The choice of species used for revegetation is critical to its success. Seeded species need to be adapted to the soil conditions, mycorrhizal associations, elevation, climate, and precipitation level of the site (Jacobs *et al.* 1999). If livestock grazing is a primary objective of a revegetation program, a perennial grass with high forage production may be the appropriate choice (Jacobs *et al.* 1999). To maintain suppression of invasive weeds, the revegetated species should grow vigorously and be competitive. This can often be a major limitation to revegetation programs. Only a limited number of species have proven to be aggressive enough to displace invasive species and the proper species choice varies depending on the location and objective.

BIOLOGICAL CONTROL

The goal of a biological control program is not to eradicate the target weed, but to exert sufficient environmental stress to reduce its dominance in the plant community (Wilson and MacCaffrey 1999). Insect agents can achieve this by boring into roots, shoots and stems, defoliation, seed predation, or extracting plant fluids. These effects can reduce the competitive ability of the plant relative to the surrounding vegetation. Although biological control agents can include nematodes, pathogens, and vertebrates, the vast majority of those released are insects or mites (Julien 1989). Most of these have targeted non-native forb weeds of rangelands (Julien 1992). Biocontrol can be a cost-effective, long-term, and self-sustaining management option (Blossey *et al.* 1994).

Burn timing can be critical to the survival of biological control agents. Leafy spurge flea beetles are active during the summer. In one study, mid-spring and fall burns did not injure the leafy spurge flea beetle because they were below ground and, in fact, burns increased the population (Fellows and Newton 1999). In most situations, however, the direct effect of prescribed burning is damaging to biological control agents within the burn site. Despite this limitation, biological control can still play a key role in an integrated approach with burning. Since insects and pathogens are mobile organisms, they have the opportunity to readily re-occupy the treated site. This has been shown for agents specific to yellow starthistle (Pitcairn and DiTomaso *et al.*, unpublished data) and klamathweed (common St. Johnswort) (Briese 1996). The combination of other control effects—including prescribed burning—with biological agents can further stress the target plant and not only reduce its competitiveness, but slow its ability to re-establish and dominate a site.

CHEMICAL CONTROL

Most integrated approaches that include prescribed burning also incorporate an herbicide treatment either before or after the burn. Many examples have already been discussed. Generally, these treatments are broadcast applications made by ground or aerial (fixed wing or helicopter) equipment. With patchy infestations it is also possible to use directed applications (spot treatments), wick treatments, or stem treatments, such as basal bark, cut stump, or hack-and-squirt. Stem applications are generally made to woody vegetation. All these treatments can be made before a prescribed burn to increase fuel loads, or after burning to control resprouting vegetation.

Data Gaps

Most often a single method is not effective in the sustainable control of a range weed. A successful long-term management program should be designed to include combinations of mechanical, cultural, biological, and chemical control techniques (DiTomaso 2000). Integrated approaches for weed management have not been well studied. This is true with or without the incorporation of prescribed burning. Innovative land managers have successfully used integrated strategies to control many different species. However, these reports occur mostly as anecdotal stories and do not appear in the literature. A greater exchange of information is needed in all areas related to the use of integrated weed management in rangelands and wildlands.

Burns may clear one weed and stimulate another. *A burn was conducted to control ripgut brome* (Bromus diandrus), *which remains dominant in the unburned foreground. Beyond the firebreak, a nonnative forb, black mustard* (Brassica nigra), *has taken over after the grassland burned. (Photo by Andrew Moyes, Santa Monica National Recreation Area)*

CHAPTER 4: Effects of Fire on Plant Communities

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The current state of information regarding the use of fire to manage invasive plants in wildlands largely resides in agency reports, some scientific publications, and the personal knowledge of individuals. The vast majority of this information is focused on the immediate effects of fire on the target invasive species, whereas there are very few examples of studies that have evaluated higher-order effects of these treatments on plant communities, soils, wildlife, or ecosystems in general. A major purpose of this chapter is to highlight the complexities of fire/plant community interactions, with the specific goal of providing guidance regarding the types of information that could be integrated into comprehensive monitoring plans for invasive plant control projects in the future.

Characteristics of Individual Fires

Any plans to use fire as a management tool must recognize that all fires are not the same. Fires can vary in their timing among seasons and within days. Similar to other invasive plant control methods, there may be an ideal time during the phenologic development of a target plant species when the application of fire will have its maximum effect. If this relationship is known, then fire can be precisely applied to achieve its desired effect.

Native plant communities can regenerate after fire. *Meadow barley* (Hordeum brachyantherum) *resprouting in early Spring, 1998 following a prescribed burn the summer before to control barb goatgrass. Two years of burning was found to provide good control of barb goatgrass. Hopland Research and Extension Center, Mendocino County, California. (Photo by Joe DiTomaso, University of California, Davis)*

The effectiveness of fire in controlling invasive plants can be greatly influenced by the amount of plant material that is consumed. Fires can vary in the way they move through fuelbeds. At one extreme, fire can burn all above-ground plant tissue in its path, creating what is known as a “complete burn.” At the other extreme, fire may only burn a subset of the total above-ground plant tissue and leave some areas only scorched or completely untouched. This irregular pattern is referred to as a “patchy burn.” Complete burns can only be reliably created where fuelbeds are highly flammable and continuous (e.g. in grasslands or Mediterranean shrublands). In most other cases, it can be difficult to consume all plant material within a burn perimeter.

Fire behavior of the flaming front can also vary significantly within and among fires. The rate of spread, residency time, depth (width), and height of the flaming front all relate to the intensity of fire, which is perhaps the single most significant variable related to fire’s ecological effects. The duration of smoldering combustion after the flaming front has passed can also have tremendous effects on soil heating, which can affect soil properties (see Chapter 5), and mortality rates of plant roots and soil seedbanks.

Characteristics of Fire Regimes

A clear distinction must also be made regarding the effects of a single fire versus the effects of a repeated pattern of burning over time, otherwise called a fire regime. Single fires can display significant temporal and spatial variability, as described. This variability is compounded in fire regimes involving multiple fires over time. Fire regimes can vary in the type, frequency, intensity, extent and spatial pattern, and seasonality of fire (Keeley 1977, Sando 1978, Heinselman 1981, Kilgore 1981). This variation can make it very difficult to predict the effects of fire treatments, especially the long-term effects of fire regimes on plant communities.

Effects of Fire on Individual Plants

High temperatures during fires can result in direct damage to plant tissue through combustion, and indirect damage to the physiological processes through radiant heating (Levitt 1972). Plant tissue that is metabolically inactive or dehydrated can withstand greater heating than tissue that is metabolically active or hydrated (Whelan 1995). Thus, burning during a plant's active growing season often results in the highest mortality rates.

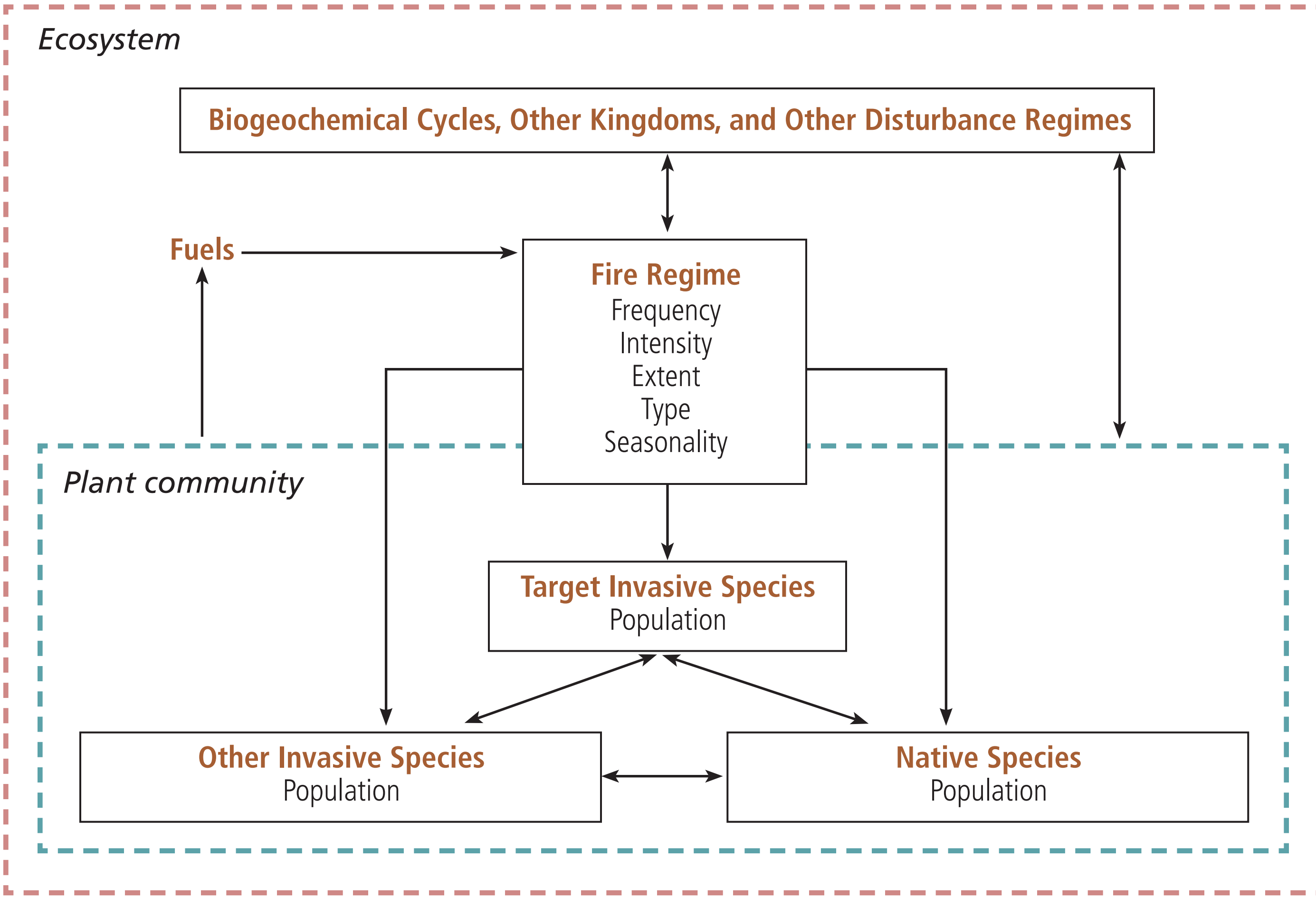
The effect of fire on individual plants depends on the degree to which perennating (surviving) tissues are protected from lethal temperatures (Whelan 1995). Raunkiaer (1934) developed a life form classification system that groups plants into categories based on the exposure of the perennating tissue to stressful growing conditions caused by summer drought or winter cold. This system is roughly based on the vertical positioning of the perennating tissue above or below the soil surface, which generally corresponds to areas of relatively high and low temperatures during fires (e.g. Brooks 2002). Pyke *et al.* (in prep.) recognized that this well-known classification system may be useful in predicting the general responses of different plant species to fire, and developed a decision support tool based on these responses. A summary of these predicted responses is provided in Table 2.

Post-burn monitoring establishes each fire's impact. *By recording species and other vegetation characteristics in burn plots, long-term trends can be established, allowing resource managers to determine whether or not prescribed fire objectives are being met. At Point Reyes National Seashore, vegetation data is collected pre-burn, and at one, two, five, and ten years after the fire. (Photo by Andrew Moyes, National Park Service)*

Table 2. Effects of fire on different Raunkiaer (1934) plant life forms (modified from Pyke *et al.* in prep).

Raunkiaer Life Form	Example	Perennating tissue	Exposure of perennating tissue to damage from fire
Therophytes	Annuals	Seeds that reside 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 that passes though the plant canopies, or by surface fire that girdles the trees.

Figure 1. Conceptual model displaying the interactions between plant communities and fire regimes within the context of the broader plant community and ecosystem.



Effects of Fire on Plant Populations and Communities

The collective responses of individual plants to fire produce population responses, which themselves collectively produce plant community responses (Figure 1). Predicting population and community responses to fire is difficult enough, considering all of the possible responses by individual species to fire and the multitude of interactions among plant species. However, the task becomes significantly more difficult when one takes into account the variable characteristics of fire and their implication for plant mortality, in addition to the effects of environmental conditions before and after burning, the effects of other taxa, and the effects of disturbance factors on the plant community. Accordingly, predictive fire effects models must include many variables to be reasonably accurate.

Use of Fire to Manage Invasive Plants

The management of invasive plants is typically done for two primary reasons: (1) to reduce the dominance (e.g. density, cover, biomass) of the target invasive species; and (2) to increase the dominance and diversity of native plants. This necessarily requires a plan to manage the invasive plant species at the population level, and the rest of the plant species at the community level. Even if the project goal is to simply reduce dominance of the invasive species, this may not be achieved if the effects of fire treatments on other interacting factors such as the plant community are not considered. In addition, treatments to control invasive plants can often improve site conditions for other invasive species, potentially creating new and possibly greater vegetation management challenges.

Effects of fire on target invasive plant species should minimally take into account the direct and indirect effects of the fire treatments on the target species, other invasive plant species, and the native species present at the site (Figure 1). Interactions among these factors are largely responsible for the characteristics of the resultant plant community. Because invasive plants generally thrive in disturbed environments, they often dominate post-fire landscape unless the native species are also fire-adapted. In some cases, active revegetation of native species may be necessary to both suppress the growth of invasives and promote the dominance of natives immediately after fire treatments are applied.

If the goal of treatments is to produce a self-sustaining fire regime, then the resultant plant community must create fuels appropriate for the desired regime characteristics (Figure 1). For example, where woody species have invaded herbaceous communities due to past fire suppression, one of the best ways to promote long-term control of the woody species is to restore a low-intensity frequent-fire regime. This is only possible if herbaceous fuels accumulate rapidly after they burn, thus promoting recurrent fire at an interval that prevents the seedlings of woody plants from surviving.

Experimental studies examine ecological effects of fire. *On Santa Cruz Island, CA, The Nature Conservancy used patchwork blocks to test how site topography and orientation, as well as burn frequency and seasonality, affect resulting plant community composition and structure. Data was collected before burning and for four years after burning. Results were complex, but in all cases non-native grasses regained dominance when burning stopped. (Photo by Rob Klinger, The Nature Conservancy)*

Research Needs

The multitudes of environmental conditions that occur pre- and post-fire ultimately influence the response of invasive plants to control treatments. Because research is not likely to test all scenarios, management burns for invasive plants should include effectiveness monitoring plans to take advantage of every opportunity to generate new data and improve predictive models. These plans should address weather, fire behavior, and post-fire treatments and responses to provide both science and management with useful information on the future potential for fire to control invasive plants.

Many studies describing the effects of fire on vegetation have been conducted after fires have occurred (i.e. post-hoc studies), where there was little or no information on the characteristics of the fire being studied. In many other cases where fire is applied as part of an experiment, the characteristics of fire are described in general terms, or summary values are presented describing the average characteristics of variables such as rate of spread, flame length, and residency time. Much of the intricate details of a fire that could be used to explain the responses by plant communities are often omitted (e.g. variations in fire behavior and patterns of fuel consumption, fire intensity, and soil heating). Future studies should attempt to include as much detail as possible about the fire treatment being evaluated. This will provide a stronger framework to build inferential models of the effects of fire on plant communities and other ecosystem properties.

CHAPTER 5: Effects of Fire on Chemical, Physical, and Biotic Properties of Soil

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A major concern of using controlled fires for management of invasive species is the potential unwanted negative effects of fire on soil chemical, physical, and biological properties. Furthermore, invasive species themselves may alter soil chemistry and biota in the absence of any fire. But fire also has beneficial effects on soils, and may be used to restore the negative effects that invasive plants may have caused. Changes caused by invasive species, unlike those caused by most fires, are considered permanent without restoration efforts.

This chapter compares the effects of fire and invasive plants on soil properties, and provides guidelines for managers who wish to consider fire as a way to restore soil altered by invasive species.

Effects of Fire on Soil Chemical and Physical Properties

Fire temperature. Negative effects of fire on soil are caused primarily by high temperatures that affect surface as well as deeper soils (Neary *et al.* 1999, Korb *et al.* 2004), while low- and moderate-temperature fires generally have long-term positive benefits for fire-adapted ecosystems (DeBano *et al.* 1998). High temperature fires are especially a problem where historic fire suppression has caused an increase in the fuel load, or in stand-replacing and slow-burning fires in forests and high-productivity shrublands that may burn at temperatures of 700°C or greater. Alternatively, ground fires in forests (with low ground fuel load) and grass fires burn at a range of 200-300°C (Rundel 1983). 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).

Other factors that affect fire temperature are rate of burn and soil moisture. Heat damage from fire is greater in a hot smoldering fire that travels slowly across the landscape than a rapidly-moving fire. A moist soil will conduct more heat downward than a dry soil. A soil with a low fuel load and a low severity of heating will attain temperatures of only 100°C, with temperatures of approximately 50°C at 5 cm depth. When soil surface temperatures attain 700 °C, they 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 controlled burns may be reduced by managing for a fast-moving fire when soils are relatively depleted in moisture.

NITROGEN AND ORGANIC MATTER

Fires have differential effects on different components of the soil system depending largely on the components' susceptibility to fire temperature. Comprehensive reviews of fire impacts are provided in Neary *et al.* (1999) and DeBano *et al.* (1998). (This synthesis of fire effects on chemical and physical properties is taken in large part from these two sources, but also additional and more recently published literature summarized in Table 3.) Organic material is always lost in fires, as the primary constituent, C, volatilizes at 180°C. Surface litter and dry plant material are lost in the lowest temperature fires, and moist, living plant material is consumed when flames or adjacent smoldering material first dries it out. Soil organic matter (SOM) is lost as the soil temperature becomes hotter than 180°C. In a survey of twenty semi-arid to sub-mesic ecosystems on impacts of fire, of those reporting SOM, nine lost OM, four did not change, and one particularly low temperature fire even gained SOM (Tables 3, 4).

Nitrogen (N) is the nutrient that causes the most concern for land managers using fire because it begins to volatilize at 200°C. Over one-half of the soil N can be lost when the temperature is raised to

500°C. However, in ecosystems where decomposition is slow because of limited precipitation, cold temperatures, or short growing season, fire is an important agent of N mineralization. Total ecosystem N losses and gain of mineralized N through fire are tradeoffs. Most semiarid ecosystems have only small amounts of total ecosystem N in litter ranging from 1 to 10% (Neary *et al.* 1999). Even if 50% of N is volatilized during a fire, the total ecosystem loss is small in a fire that consumes, for instance, grass litter during the dry season when grass leaves have senesced and much N has been resorbed by the below-ground, living plant parts.

Alternatively, a crown fire in shrublands will remove a higher proportion of N, but in fire-adapted systems the ecosystem N is capable of recovery under natural fire regimes, as explained below. The mineralization of N immobilized in soil organic matter and litter may off-set the total ecosystem loss in terms of increased plant productivity. This generalization assumes, of course, that the vegetation is adapted to fire and will be able to resprout or reseed rapidly to take up the mineralized N. A literature survey of 20 sub-mesic to semi-arid ecosystems shows that more than one-half of the sites where N was reported decreased in total soil N following fire, while the others showed no change (Tables 3, 4). At the same time, more than one-half of the sites had an increase in mineral N with others showing either a decrease or no change. Decreases in extractable N were generally caused by erosion. Several of the studies also reported increased productivity of the vegetation following the burn, presumably because of increased inorganic nutrients. This same general conclusion was summarized in earlier reviews (Neary *et al.* 1999, DeBano *et al.* 1998), which gave multiple examples of fires that decreased total ecosystem N but promoted increased productivity of vegetation following fire due to mineralization.

Fire affects all components of soil. *Chemical, biological and physical properties are all altered from heat and oxidation of fuels. Henninger Ranch, U.S. Sheep Experiment Station, Kilgore, ID. (Photo by Stephen Ausmus, USDA Agricultural Research Service)*

OTHER NUTRIENTS

In contrast to N, both potassium (K) and phosphorus (P) require >700 °C for volatilization and their loss is usually minimal unless the fire is followed by erosion. Other nutrients such as calcium (Ca), magnesium (Mg), and sodium (Na) require much higher temperatures for volatilization. These other nutrients showed either increases or decreases depending upon the type of ion (Tables 3, 4). Extractable P was higher after fire in one half of the studies that reported P, and decreased or showed no change in the others. The fire temperature was not reported in most of these fires, but was most likely below 700°C in all or most, as fires seldom burn hotter than this at ground level under natural levels of fuel buildup. The mineralization of organic forms of P by fire would increase extractable P, while post-fire erosion would reduce it. The other inorganic ions important to plant growth measured in these studies were primarily extractable K, Ca, and Mg, which increased in most cases due to fire-caused mineralization, and were lost due to wind or water erosion in others.

Table 3. Responses of soil chemical and physical properties to fire in semi-arid to sub-mesic, fire-prone ecosystems. "+" indicates an increase, "-" indicates a decrease, "NC" indicates no change in a property, "min" indicates minimal, although statistically significant, change. A response of "- then +" means the fire caused a reduction in that property, followed by an increase over time in multi-season studies. Two responses in one column indicate that different sites had different responses for that publication. Blanks indicate that property was not reported.

Publication	Location	Vegetation	# Yrs after fire	pH	Extr. N	Total N	Extr. P	Inorg. Ions	OM	Bulk Density
Anderson *et al.* 1997	Florida	Sandhill pine	> 1 yr.		+ then -	NC	+		-	
Badia and Marti 2003a,b	Spain	Medit. shrub	Lab	+		NC or -		-	-	+
Ballard 2000	British Columbia	Boreal forest	variable	+	+	-	-	+, NC	-	+
Bauhus *et al.* 1993	Australia	Forest	Lab	+	+		+			
Bennett *et al.* 2003	Australia	Grassland	> 1 yr.	+	NC					
Blank *et al.* 2003	Nevada	Riparian	3 yrs.		-	- (min)		+/-		
Castelli and Lazzari 2002	Argentina	Shrub	> 1 yr., first burn		+, NC		+, NC	+, NC		
Castelli and Lazzari 2002	Argentina	Shrub	3 yr., 2nd burn		-, NC		NC	-, NC		
Forgeard and Frenot 1996	France	Heath	Lab					NC, -	NC, -	
Franco and Sosa 1997	Baja Cal.	Medit. shrub	1 yr.	NC	+	NC	NC (total P)		NC	
Gimeno *et al.* 2000	Spain	Medit. shrub	< 1 yr.		-			-	-	
Kutiel and Naveh 1990	Israel	Pine forest	> 1 yr.		+ then NC	-	+ then NC		-	
Litton and Santelices 2003	Chile	Forest	2 yrs.	+				+/-		
Lynham *et al.* 1998	Ontario	Pine forest	10 yrs.	+ then -	+	-	+	+	- (min)	
Mills and Fey 2004	S. Africa	Grass/ savanna	Annual, 30+ yrs.	+	-	- (min)		-	NC	
Ojima *et al.* 1994	Kansas	Grassland	Annual, 10 yrs.			NC then-			+, then -	
Picone *et al.* 2003	Costa Rica	Grassland	1 yr.		+	-	+	+	-	
Seastedt and Ramundo 1990	Kansas	Grassland	Annual, 10 yrs.	NC		NC			NC	NC
Snyman 2002	S. Africa	Grassland	2 yrs.	+		-	- then NC	+ then NC	-	+
Snyman 2003	S. Africa	Grassland	2 yrs.	+		-	- then NC	+ then NC	-then NC	+

Table 4. Number of studies showing changes in soil chemical and physical properties. Based on Table 3, with bolded values indicating those responses that are most frequent for each property.

Property	increase	decrease	no change
Soil pH	**9**	0	2
Soil extractable N	**8**	4	3
Soil total N	0	**8**	5
Soil extractable P	**6**	3	3
Other inorganic ions	**8**	6	4
Soil organic matter	1	**9**	4
Bulk density	**4**	0	1

PH

Of the 11 studies that reported effects of fire on pH, nine had increased pH and two showed no change after fire (Table 4). Elevated pH is the usual situation following fire, at least in the few surface cm of soil (Neary *et al.* 1999). The ash left behind after a fire consists primarily of cations (e.g., Ca, Mg, Mn, K, Na) that were constituents of plant living tissue and OM. Plant tissue may contain up to about 10% dry mass of cations, depending upon the species and type of tissue. Upon wetting, the pH of pure plant ash material dissolved in the laboratory may be 9 to 11. Following a fire the base cation oxides are hydrolyzed with the next moisture input, creating a basic solution. The resultant soil pH depends upon the quantity and constituent ions of the ash, the buffering capacity of the soil, and leaching. Some of the ash may be lost by erosion, causing future nutrient depletion, while some ions are leached into the soil profile causing decreased pH below the surface. The pH will drop again rapidly as plants grow, taking up cations whose charges will again be balanced by negatively-charged plant organic acids.

PHYSICAL PROPERTIES

Fire also affects soil bulk density, erosion, hydrophobicity, and moisture. Several of the studies reported either an increase or no change in bulk density following fire (Table 4). The increase in bulk density is largely caused by the combustion of surface roots and decomposition of roots throughout the profile when shoots are fire-killed (Neary *et al.* 1999). With the change in soil structure due to loss of soil macropores, infiltration is reduced, and the subsequent drier soil may slow vegetation recovery, especially in semi-arid areas (Snyman 2002, 2003). Lower soil moisture may also be related to higher soil temperature in burned areas that have no litter or vegetation cover to ameliorate direct radiation, and to loss of litter that slows the surface movement of water and allows greater infiltration. Furthermore, soil crusting may occur after fire in soils high in clay, which also impedes water infiltration (Mills and Fey 2004).

Hydrophobicity, or water repellency, is caused when fire-bared soil surfaces seal under the impact of raindrops, resulting in increased surface runoff (Doerr *et al.* 1998, Neary *et al.* 1999, Ballard 2000). This occurs after hot fires or certain litter types that leave hydrophobic organic compounds in the soil surface, although the hydrophobic compounds are destroyed in extremely hot fires where soil temperatures are greater than 290°C (DeBano *et al.* 1976, Neary *et al.* 1999). Hydrophobicity may contribute to erosion on slopes and loss of nutrient-rich ash and topsoils, and is another long-term impact of fire.

Erosion is the most devastating impact of crown fires that remove all vegetation including SOM. The erosive impacts of these fires are well known in areas such as southern California, where at this writing the daily news brings stories of mudslides during the 2004-2005 El Niño rainfall season, following the widespread (>300,000 ha) fires of November 2003. The burn areas impacted by erosion are primarily steep slopes that were covered with coniferous forest, chaparral, and coastal sage scrub. Recovery of woody vegetation is slow and erosion continues for three years or more after a burn, even with efforts to slow erosion by seeding non-native grasses (Beyers 2004).

LONG-TERM RESILIENCE AND RECOVERY OF CHEMICAL AND PHYSICAL PROPERTIES

There are many examples of rapid recovery from fire in fire-adapted vegetation under normal fire regimes, with normal fuel loads, and normal fire temperatures for that ecosystem type (DeBano *et al.* 1998). Recovery can be variously defined, such as recovery of soil N or C capital to ecosystem N or

C capital (the latter includes soil, litter, and plant N and C). Herbaceous vegetation may recover quickly, while the aboveground C in forests may take decades or centuries. More revealing recovery and degradation patterns can be gleaned from more intensively burned sites. A modeling exercise by Ojima *et al.* (1994) showed that annual burning of Kansas tallgrass prairie for 10 years caused an initial increase in soil organic matter with no change in soil N, but after several years the soil N did decline. The model assumed erosion had occurred. The validation data set came from Seastedt and Ramundo (1990), who, contrary to the model, measured no change in either organic matter or N, but also did not observe erosion. According to the predictions of Neary *et al.* (1999), ecosystems with a high proportion of biomass and nutrient storage belowground, such as tallgrass prairie, are more buffered from the severe impacts of fire than those with a smaller proportion below ground.

Some remarkable long-term studies were done in South African grassland and savanna. Annual burns over more than 30 years showed no change in soil organic matter, but a minimal decrease in total soil N, and a decrease in extractable N (Mills and Fey 2004), again showing the resiliency of sub-mesic grassland. Fire temperatures tend to be somewhat lower in grasslands, so relatively little soil N will be volatilized.

On the other hand, chaparral is a fire-adapted shrubland that sometimes has extremely high surface temperatures of 700°C. Although the vegetation may be totally consumed, the impacts on soil N are mixed, with most fires showing increases or no change in mineral N (decreases occur primarily through erosion), and others showing decreases or no change in total N (Table 3, Neary *et al.* 1999, DeBano *et al.* 1998). Consumption of the vegetation means that a large fraction of total ecosystem N has been lost, but studies have shown the capacity of chaparral to recover its N capital during succession. Available N in fact declined during 80 years of succession following fire in chaparral, indicating the role of fire to release immobilized N and stimulate plant growth (Fenn *et al.* 1993). In addition, the early years of succession in chaparral are characterized by high colonization of the N-fixing leguminous shrub *Lotus scoparius*, which declines after only 5-7 years. Other sites have persistent stands of actinorhizal *Ceanothus* spp., which also fix N.

Soil samples and vegetation reflect changes. *Measurements before and after burns can document changes to soil nutrients, plant productivity, and other parameters. (Photo by Stephen Ausmus, USDA, Agricultural Research Service)*

Chaparral also recovers quickly because it has a large proportion of its biomass belowground, and because soil temperatures drop below 200°C at 2.5 cm, thus reducing the loss of soil N (DeBano *et al.* 1977, cited in Neary *et al.* 1999). The studies that have shown poor recovery from fire are those in slash piles (Korb *et al.* 2004) where both biotic and physical characteristics of soils are severely changed due to high soil temperature. Lack of recovery may also be due to the loss of the soil seed bank in hot, stand-replacing fires. Furthermore, any sites that are subject to erosion for the reasons described above will also show poor recovery. For the purposes of using fire to control invasive weed species, the obvious implications are that large burns on steep slopes should be avoided, and biomass reduction prior to burning may be necessary to avoid very hot burns.

Effects of Fire on Soil Microbiological Properties

Nutrient availability to plants is regulated by soil microorganisms, so their survival through the fire and their ability to recover after fire are essential to restoring burned sites. Survival is dependent upon soil temperature, with surface temperatures >100°C killing most microorganisms, at least in moist soils. However, soil is a good insulator, and the temperature at 2.5 cm depth may be a benign 50°C when the surface is 100°C (Neary *et al.* 1999). Thus the survival of subsurface organisms is guaranteed even in a moderately hot fire. However, when the soil surface temperature reaches 700°C, as under slash or very high fuel accumulations, the soil temperature may be as hot as 100°C down to 22cm (Neary *et al.* 1999).

A substantial number of studies have been done on saprotrophic bacteria and fungi and mycorrhizal fungi following fire, using a variety of measurements to detect abundance or activity. The measurements include soil respiration, microbial biomass, microbial C and N, nitrification, mineralization and direct microscopy. Some studies are based on short-term laboratory incubation studies, but many more include multi-year field observations.

Studies that measured microbial biomass after fire by measuring microbial C or respiration found an increase in microbial activity as often as a decrease (Acea 1996, Andersson *et al.* 2004, Fonturbel *et al.* 1995, Mabuhay and Nakagoshi 2003, Badia and Marti 2003a,b, Bauhus *et al.* 1993, Garcia-Oliva *et al.* 1998). Saprotrophic microbial activity after a fire depends upon how much of the soil organic matter was consumed by the fire. The fire may leave a large amount of dead, but not completely combusted organic material, thus providing a carbon source for saprotrophs.

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^- (Bauhus *et al.* 1993, Anderson *et al.* 2004, White and Zak 2004). However, after several months the microbial activity often declines to levels below the immediate post-fire level, and may not recover entirely until the soil organic matter biomass recovers to pre-burn levels (White and Zak 2004). A reduction in nitrification is not necessarily limiting to plant productivity, as most plants are capable of taking up NH_4^+ directly. In fact, most studies of wildland fires at "normal" fuel loads, including fast-moving chaparral fires which may have soil surface temperatures of 700°C (but decline to 100°C within a few cm, Neary *et al.* 1999), show an increased productivity of vegetation for one or multiple years post-fire, another indication that productivity has not been hindered (e.g., Seastedt and Ramundo 1990, Carreira and Niell 1992).

Mycorrhizal fungi are vital for nutrient uptake. *The* Russula *spp. pictured here (approximately 35x magnification) form ectomycorrhizal root tips on* Quercus agrifolia, *and help the tree cope with drought stress. Soil microorganisms like these residing 2-3 cm below the soil surface typically survive all but the hottest fires. (Photo by Edith Allen, University of California, Riverside)*

Mycorrhizal fungi are also of concern because they are important in procuring nutrients and water for plants, as well as other functions of drought stress tolerance and pathogen protection. Of the studies reviewed, only two showed a limitation of mycorrhizal inoculum for plant establishment following fire. One of these was a slash pile fire that burned for several days in one location, where soil biota were charred to 10 cm depth (Korb *et al.* 2004). Attempts to restore pines failed unless they were inoculated with mycorrhizal fungi. Another severe fire was the Yellowstone fire of 1988, which burned patchily across the landscape in fire-suppressed lodgepole pine forest. In some locations the fires were so hot that they burned large roots to 0.3m deep, and killed soil microorganisms. Observations of recolonizing pine seedlings showed that 50% died in the first growing season, likely because of lack of mycorrhizal fungi (Miller *et al.* 1998). These are ectomycorrhizal plants, which are normally obligately mycorrhizal, meaning they will die without inoculum. However, those that did survive became mycorrhizal by the end of the first growing season. It was not clear whether the inoculum came from deep buried living inoculum, or colonized by spores from a more distant source. Spores of ectomycorrhizal fungi are easily dispersed by wind (Allen 1991), so either mode of inoculation is plausible.

A demonstration of differential soil temperature effects on mycorrhizal fungi comes from studies in pinyon-juniper woodland (Klopatek *et al.* 1994). Fires burn hotter under trees because of accumulated litter, and the percentage of arbuscular mycorrhizal inoculum was reduced more under trees than in interspaces. With the exception of the two severe fires described above (Miller *et al.* 1998, Korb *et al.* 2004) most studies showed little or no reduction in root mycorrhizal infection of plants resprouting following fire (Anderson and Menges 1997, Rashid *et al.* 1997, Allen *et al.* 2003, Korb *et al.* 2003). A study of eucalyptus fires in Australia showed that some sites had reduced infection while others did not, possibly related to soil type (Launonen *et al.* 1999). Even though there is still considerable inoculum left in the soil after fire, other studies showed that the fungal species composition is changed by fire (Stendell *et al.* 1999, Baar *et al.* 1999). Recovery of some fungal species may take years after a fire (Allen *et al.* 2003), and in fact fungi and other microorganisms undergo a fire-induced succession just as do plants.

Table 5. Number of studies showing changes in ecosystem properties following invasions by exotic species (from Ehrenfeld 2003). Bolded values indicate highest number of responses for each parameter.

Parameter	increase	decrease	no change
Plant biomass	**16**	4	0
NPP	**10**	0	2
Growth rate	**10**	0	0
Litter mass	7	5	2
R/S	1	**5**	1
Soil C	6	6	1
C mineralization	**4**	2	2
Decomposition	**10**	2	0
Microbial C	**3**	1	2
Total soil N	**9**	6	6
Extractable soil N	**9**	4	4
N mineralization	**13**	3	2
N fixation	**8**	2	0
Biomass N	**11**	1	2
Soil C/N	2	1	1

Impacts of Exotic Species on Soils

Before considering the effects of fire on invasive plants, managers must first understand how the invasive plants have affected soils. An excellent review of impacts of invasive plants on soils and nutrient cycling was recently done (Ehrenfeld 2003) and will be the basis of the review reported here (Table 5). This was a review of 79 papers that studied impacts of invasives on the same properties discussed for fire, as well as N fixation and plant growth parameters. Considering first the vegetation responses to invasions, most invaded sites had greater biomass than the native vegetation that was replaced, with 16 of 20 sites that reported biomass having greater biomass, four having decreased biomass following invasion, and none reporting no change in biomass. The remaining studies of the 79 reviewed did not report plant biomass. Most invasions are caused by productive plants such as woody species or fast-growing grasses that replace native grasslands, while those that cause a decrease in stand biomass include annual grasses that replace native shrublands. Net primary productivity and growth rate similarly tended to increase in invaded stands. However, litter mass and soil C were reported in an equal number of studies where they increased or decreased following invasion, in spite of the increased biomass of invaders. The elevated plant biomass was likely off-set by an increased decomposition rate, as 10 of 12 studies reporting decomposition had higher rates following invasion. A majority of sites also reported increase mineralization and increased microbial C.

The changes in N following invasion were also dramatic, with a majority of studies reporting increased total N, extractable N, rates of N mineralization, N fixation, and biomass N (Table 3). Many invasive species are leguminous or actinorhizal shrubs that have N-fixing nodules, and these are especially problematic in promoting elevated ecosystem N and causing permanent alterations in the nutrient cycling of the system. Alternatively, those systems that are invaded by a single species of flammable annual grass, such as *Bromus tectorum*, have reduced soil N over time caused by frequent fires and erosion (Evans *et al.* 2001).

Sometimes cause and effect may not be easily differentiated, as invasive species richness and cover were greater in nutrient-rich than nutrient-poor soils (Bashkin *et al.* 2003). This study concluded that the invasive species selected nutrient-rich soils, but invasive species may also promote faster mineralization and higher extractable nutrients (Evans *et al.* 2001).

Finally, invasive plant species may also cause changes in the species composition of soil microorganisms. The invasive grass *Bromus madritensis* formed an association with a species of arbuscular mycorrhizal fungus termed the "fine endophyte," while native shrubs that it replaced formed associations with the normal "coarse endophyte" (Sigüenza *et al.* 2005). This had a functional response, in that fine endophyte promoted a greater growth response of the exotic grass than the coarse endophyte did for the native shrub. In another study, the shift in arbuscular mycorrhizal composition caused by another exotic grass, *Avena barbata*, was reversed after native grasses were planted again (Nelson and Allen 1993). Shifts in microbial species composition caused by invasive plants are not well studied, but in this case was alleviated by restoration.

Conclusions

Managers are sometimes reluctant to use fire as a tool because of the potential negative impacts on soil properties. However, these risks can be mitigated by careful burning and judicious selection of sites. The most severe long-term effects of fire will occur where erosion removes topsoils, but the risk of erosion can be reduced by burning only small areas when treating on steep hills, or avoiding burning altogether in steep terrain in favor of other weed control measures. Along with erosion, managers are concerned that hot fires will alter soil chemical, physical, and biological properties. Only the hottest fires, as under slash piles or fire-suppressed vegetation with high litter build-up, will cause long-term changes in soil chemical, physical, and biological properties. Hot fires can be avoided by careful fuel control in advance of the fire. This has been done, for instance, in high density ponderosa pine forests in northern Arizona, where pole-size trees were first removed before fire (Fulé *et al.* 2004).

Fire may be especially beneficial where invasive species have increased the litter layer and increased soil N. While there have been many studies on the effects of fire on N losses in natural vegetation, studies on use of fire to deplete elevated N in invaded systems are apparently still lacking. With the knowledge of fire temperatures and fuel loads needed to reduce surface N, fire seems like a viable method not only of controlling the invasive plants as described in other chapters in this volume, but also of reducing elevated soil N over time. Studies are needed to determine the extent to which fire will restore soils with elevated N and C caused by invasive species.

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