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Integrated Management of Fire-Adapted Invasive Plants That Change Wildfire Regimes

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Introduction

Fire is an important process in North American landscapes and ecosystems and altering the pattern of fire can have important consequences for human health and ecosystem health and services. Invasive plants that are fire-adapted, such as cheatgrass (*Bromus tectorum*) and medusahead (*Elymus caput-medusae*), increase fire frequency and intensity by changing the amount, type, and distribution of fuel loads (Figure 1.1) and this change in fire frequency provides a competitive advantage to the fire-adapted invader. Invasive plants that are not fire-adapted, such as Brazilian peppertree (*Schinus terebinthifolia*), often decrease wildfire frequency by similarly affecting fuel loads and changing the competitive balance.

As invasive plants shape their environment, they change resource availability, species composition, and ecological processes (Figure 1.1). These changes affect the ecological services, like carbon sequestration and recreation, provided by these natural systems. The amount of carbon sequestered can be estimated by the carbon emissions during a wildfire and the amounts are sobering (Loehman et al. 2014, van der Velde et al. 2021, Bartowitz et al. 2022). The negative impacts of invasive weed species on recreational activities such as fishing, hunting, hiking, wildlife viewing, and water-based recreation are well documented and can cost millions in lost revenue and hundreds of jobs (Eiswerth et al 2005). Maintaining the ecological services provided by these natural systems is integral to the missions of several federal and state agencies and departments.

A thorough understanding of the role these invasive plants play in changing ecosystems and fire cycles is essential to a proper accounting of their impacts. For invasive plants that increase wildfire frequency and intensity, their human health and economic impacts are documented, in part, in the data collected on the impacts of wildfires (National Academies of Sciences, Engineering, and Medicine 2020). Capital losses associated with the 2018 wildfires in California totaled over $150 billion (Wang et al. 2021). It is estimated that federal firefighting costs in the U.S. were about $3 billion per year (Burke et al. 2021), and it is estimated that there are about 50 million homes in the wild-land-interface that could be affected by wildfires (Burke

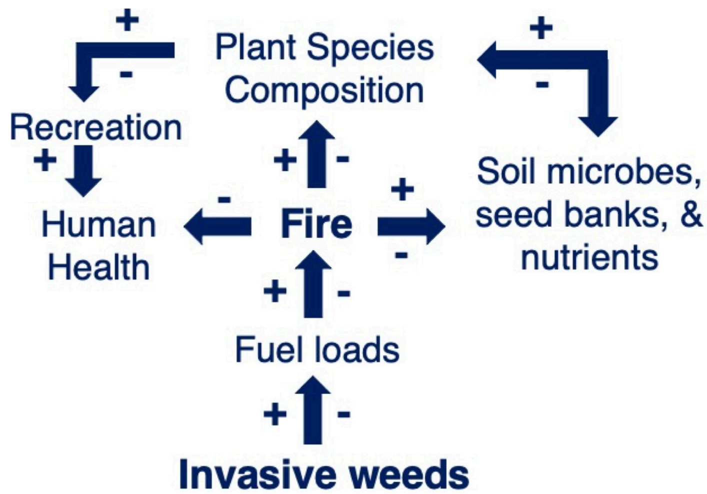


Figure 1.1. How invasive weeds affect wildfire regimes that in turn affect their environment and human health

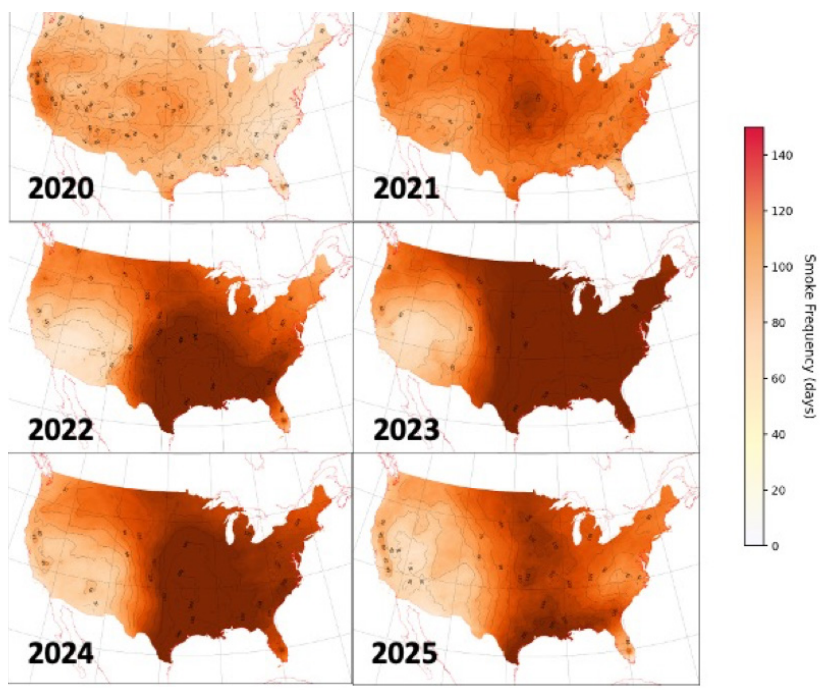


Figure 1.2. National Oceanographic and Atmospheric Administration's Hazard Mapping System showing days exposed to wildfire smoke in the contiguous U.S. from 2020-2025 (available at www.ospo.noaa.gov)

et al. 2021). In addition to death, wildfires have significant human health consequences. Exposure to smoke from wildfires can increase the risks of asthma, COPD (chronic obstructive pulmonary disease), heart failure, ischemic heart disease, and heart attack (Magzamen et al 2021, Rice et al. 2021, Sullivan et al. 2022) and the extent of smoke exposure across the country is alarming. Viewing the maps from the National Oceanographic and Atmospheric Administration's Hazard Mapping System (available at www.ospo.noaa.gov) to identify days exposed to wildfire smoke (Figure 1.2), we found that over the past six years, large swaths of the country were exposed to smoke plumes containing ambient particulate matter consisting of 2.5 micron particles known to exacerbate respiratory and cardiovascular problems. Although it is impossible to say that invasive plants are entirely responsible for these impacts, we contend that they play a significant role and therefore some of these economic and human health impacts can be attributed to their presence.

Because invasive plants can impact fire frequency and intensity in negative ways, it is also important to understand the impacts associated with plants that reduce wildfire frequencies and intensities. However, these data are not often recorded or tracked, so it is more difficult to estimate the human health, environmental, and economic costs associated with these invasive plant species. As such, this remains an area in need of further research.

An accurate accounting of the impacts of invasive plants on human and environmental health through their effects on wildfires should inform policy regarding the management of these invasive plants. Invasive plants with large negative impacts should be the priority for management. Currently, numerous federal, state, regional and local committees, councils, and organizations are involved in informing policy related to invasive plants involved in changing wildfire regimes. A sound policy, informed by the best information identified by committees, councils, and organizations, has the best chance of managing invasive plants that impact fire regimes in a manner consistent with federal, state, and local priorities and budgets.

Chapter 2. Invasive Plant Life Histories and Their Relationship with Fire

Fire regimes can be variably affected by plant invasions, and in turn, invasive plants can respond in different ways to fire regimes (D'Antonio and Vitousek 1992). The response of invasive plant species to fire may be determined by population characteristics, such as density, as well as the life history of the species (King and Grace, 2000, Crandall and Knight 2017). Some invasive plant species spread quickly following fires that remove aboveground biomass (Keeley 2001; Simmons et al. 2007). Others can be controlled by fires so long as the fires are frequent and occur at a time of year or life stage when the species is more sensitive to fires (Emery and Gross 2005, Simmons et al. 2007, Flory and Lewis 2009). Determining how life history characteristics direct a plant species' response to fire is a key step in developing a management plan that includes fire (Grace et al. 2001, Fill and Crandall 2019). If the goal is invasive plant control or elimination, it is essential to know if the species positively responds to fires by surviving and/or spreading through establishment of new individuals.

Invasive plants can survive fires by resisting or resprouting. Invasive trees can survive and resist fires once they have developed thick bark or grown above flame heights, escaping

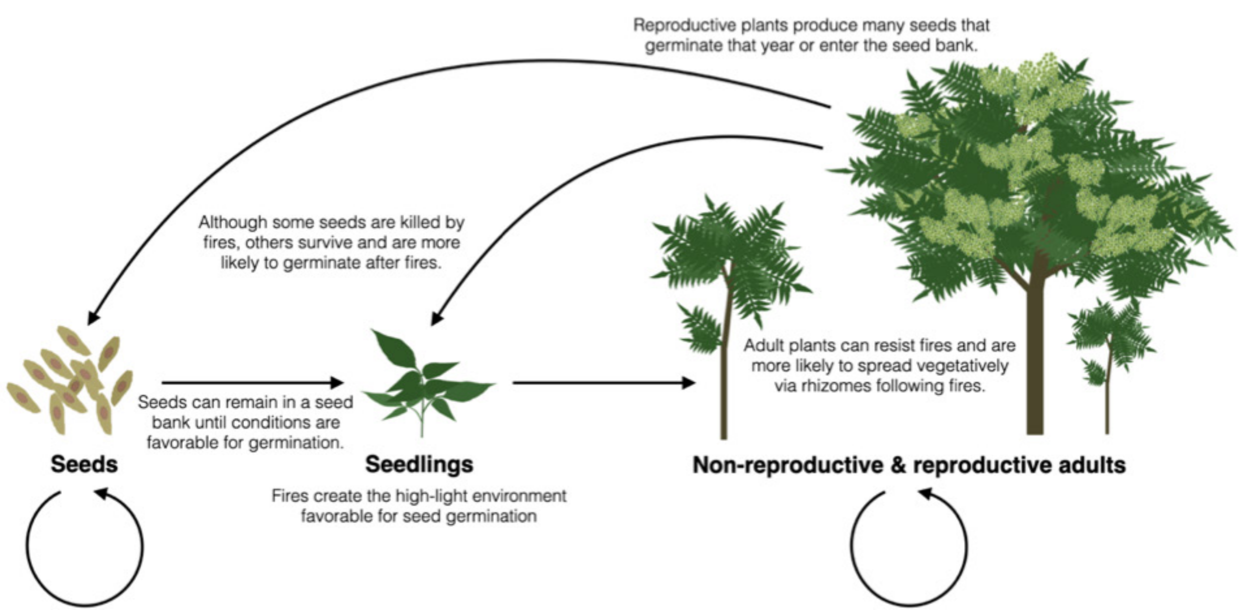


Figure 2.1. Life cycle diagram for Ailanthus altissima showing the life stages and transitions that should be considered for fire management. The circles with arrows represent stages in which plants can remain for more than a year. The other arrows indicate transitions between stages. Large trees can resist fires and spread vegetatively, producing new individuals. Seedings are also more abundant following fires that reduce the litter layer, exposing seeds to higher light levels. Thus, the spread of Ailanthus altissima is enhanced by fires via both increased seedlings and vegetative sprouts (Guthrie et al. 2016; Crandall and Knight 2015).

fires that would otherwise critically damage them (Pile et al. 2017). Some rhizomatous grasses, such as *Microstegium vimineum*, are known for increasing fire temperatures, which causes the mortality of neighboring species (Wagner and Fraterrigo 2015). This opens areas for *M. vimineum* to spread vegetatively and creates a feedback loop of more intense fires followed by invasion (Emery et al. 2013; Knapp et al. 2023). In contrast, some invasive species, such as torpedograss (*Panicum repens*), have a slightly lower abundance and spread more slowly via rhizomes after fires, providing a window during which herbicide control methods are more successful (Toth 2007). Although a species might primarily survive and spread vegetatively, there could be conditions under which it also re-establishes from seeds, increasing its potential to spread after fires (Guthrie et al. 2016; Knapp et al. 2023).

Invasive plant species that cannot survive fires must recruit in the post-fire environment and compete with any resprouting individuals. For populations to persist, invasive species that spread mostly or entirely via seed must reach reproductive maturity before the next fire. If this does not occur, seeds can be eliminated from the seedbank over time and with successive fires. For instance, yellow starthistle (*Centaurea solstitialis*) has been successfully controlled in California grasslands using multiple spring fires (Hastings and DiTomaso 1996, DiTomaso et al. 1999). This fire timing prevents seed production, reducing this annual species' ability to spread (Roche et al. 1997). Starthistle has been shown to have seeds that can persist in the seedbank for up to 10 years (Menke 1992), so annual fires would be needed, especially during early stages of control. Frequent fires could potentially be used to control annual invasives, especially during early stages of invasion (but see cheatgrass; Kerns and Day 2017). Before manipulating the fire regime to control annual invasive plants, however, the impact of such changes on co-occurring desirable species should be considered.

Some invasive plant species may persist and spread after fire using some combination of resprouting or recruiting from seeds. For instance, adult trees of tree-of-heaven (*Ailanthus altissima*) can resist fires and survive (Figure 2.1; Rebbeck et al. 2017; Crandall and Knight 2017). Although they spread vegetatively via below-ground rhizomes in the absence of fire, a higher abundance of new sprouts is observed after fire (Crandall and Knight 2015). In addition, germination from the long-lived seedbank is also enhanced where fires have removed understory litter, increasing the light availability required for germination (Guthrie et al. 2016; Rebbeck and Jolliff 2018). Understanding how plants persist and spread after fires can provide insights into effective management using fire coupled with other control methods.

Regardless of life history, some plants will be more sensitive to variation in the fire regime, particularly frequency and seasonality. For instance, burning when plants are in flower will likely reduce the seed crop for that year, slowing spread (Grace et al. 2001). In addition, some research has shown that burning invasive grasses, such as smooth brome (*Bromus inermis*) and Kentucky bluegrass (*Poa pratensis*), in the spring when they are tillering and have lower carbohydrate reserves can reduce abundances (Curtis and Partch 1948, Grilz and Romo 1994, Salesman and Thomsen 2011, Palit and DeKeyser 2022). Targeting invasive species with fire during a particular season often has only short-term effects and repeated fires or fires coupled with herbicides are often necessary for longer-term control (Bahm et al. 2011). For restoration purposes, managed fires should be ignited, when possible, during seasons and at frequencies similar to the natural fire regime for the region to avoid detrimental effects on co-occurring native species.

A more holistic approach is needed for determining the responses of invasive species following fire, including additional life history observations and information on responses to aspects of the fire regime, such as seasonal timing and frequency. Determining an invasive species' relationship with fire regimes is crucial for developing a fire management plan. Although a fire might reduce populations of an invasive species during the first year, land managers should be aware that there could be a population explosion during subsequent years or mediating factors that affect how invasive species respond to fires (Guthrie et al. 2016; Fill et al. 2019). Many studies examining fire's effects on invasive populations last only one to two years, providing little long-term data on population growth. In addition, the invasive species is often studied without accompanying data on the coexisting plant species that will also be affected by the fire regime (e.g., Sherrill et al. 2022). For current information on invasive species effects on and response to fire regimes, visit the USDA Forest Service Fire Ecology Information System (FEIS): https://www.fs.usda.gov/research/rmrs/products/dataandtools/tools/fire-effects-information-system-feis.

Chapter 3. Policy Goals and Implications

Effective policy governing invasive species management, or any ecological system, should include the following five goals: (1) Provide a manageable solution to a well-documented, well-understood problem. It should also be succinctly defined to address specific challenges with specified desired outcomes. (2) Incorporate best science. Policy is inevitably litigated, and adjudicators rely heavily on best science or the perception of best science for making decisions on subject matter with which they are unfamiliar. (3) The policy should be developed at the conclusion of an analysis phase where possible solutions have been thoroughly vetted with people who will be affected by the policy. Vetting should be performed before the policy is written. (4) The policy should be properly funded and supported before implementation, and then quickly implemented. (5) The policy should include plans for adaptive management and periodic updates over time, and the funding for additional research and monitoring.

To provide manageable policy solutions for invasive species challenges, solutions must be based in both ecological and economic realities, beginning with the reality of time. Perryman et al. (2003) coined the term pristine-management-paradigm to express the false, but commonly held, ideal that ecological systems are static entities that can be held in a static condition if only protected from burning, grazing, and other disturbances. Society has long desired a condition that resembled the landscape at the time of European settlement. This was and is an impossible goal. Setting ecosystem composition goals based on poorly understood or perhaps mere perceptions of past compositions leads only to failure. Maintaining options for future generations through ecosystem resilience is the highest and best plan for public land management. Plant communities are dynamic systems that evolve, going through plant community changes even in protected circumstances. For instance, if native plant systems could be cordoned off from invasive species, man-made or natural perturbations, and insect outbreaks at any moment in time, a half-century later they would not resemble the same ecosystems structurally or compositionally that were protected five decades prior. Through changes in climate and competitive interactions, plant densities and age classes would change over the landscape as plants mature and die while new stands develop. Even if protected from disturbances, areas that were excellent seasonal habitat for some wildlife species at the time of protection can become poor seasonal habitats over time.

Second, ecologically speaking there is no such thing as “intact or native landscapes.” It is an indefinable term that requires some entity to provide definition. For simplicity’s sake, assume that “intact or native” means that there are absolutely zero non-indigenous, invasive species present in the plant community. For example, circumscribe an area of sagebrush that contained within its boundary zero invasive annual grasses and on the other side of the boundary invasive annual grasses exist. If the non-invaded area was protected from livestock grazing, invasive species will expand into the pristine area and soon become the dominant lifeform (Gornish et al. 2023; Davies et al. 2021; Perryman et al. 2021; Stephenson et al. 2022; Perryman et al. 2020; Porensky et al. 2020; Perryman et al. 2018). Across biomes, invasive species will expand into the protected areas regardless of management efforts. They may not become immediately dominant but will become dominant if left unmanaged. Protection, and even active management efforts, will not prevent or eliminate the presence of invasive species. The key is preventing or reducing their dominance, not their presence.

If “intact, native landscapes” refer to landscape compositions and networks of the past, then things like Little Ice Age Climate features, interstate and intrastate linear disturbances such as roads and railroads, urban development, fencing, the addition of free-roaming bison herds and their unmanaged grazing behavior, intentional burning by Native Americans, and extensive naturally-ignited, unsuppressed fires would have to be included in the management parameters. We cannot go back in time attempting to produce what someone’s preferred perception of the landscape structure and composition might have been. It is an ecological impossibility.

Additionally, with the advent of invasive species in all landscapes since the 19th century, many native ecosystems in the western U.S. are now novel ecosystems with new species compositions that include invasive species and should be managed differently than they were even a few decades ago. Protection to maintain the current ecological state or phase forever is neither ecologically possible nor a thoughtful idea. Linear-thinking approaches to natural resource

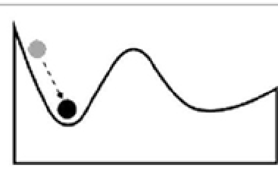
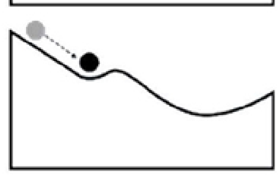
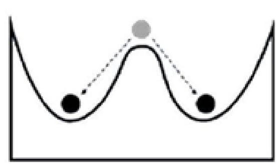
Resilience/Adaptive capacity	States and transitions	Contributing factors	Management options
High resilience		• Environment favorable • Species/functional groups stabilize system • Disturbances within HRV	• Strengthen capacity to adapt to environmental change • Prevent or minimize disturbances outside or the HRV
Low resilience		• Environment less favorable • Species/functional groups to stabilize system lacking • Disturbances outside ofHRV	• Prepare for anticipated change • Reduce risk of undesirable transition • Use opportunities to promote desired transitions
High adaptive capacity		• Changes in environment or magnitude of disturbance • Species/functional groups allow return to desired state • Disturbances within adaptive capacity	• Manage system for gradual adaptive response • Further strengthen the desired state

Figure 3.1. Ball and trough diagram showing responses of agroecosystems, watersheds, or management areas to stressors and disturbances. The agroecosystem, watershed, or management area (ball) can leave the current stable state (valley) with sufficient perturbation or change (arrows) to drop into an adjacent stable state (adjacent valley). HRV is historical range of variation that reflects landscape resilience to perturbation and is based on available empirical data, such as vegetation cover type area, disturbed area, or patch size distribution. (Adapted from Chambers et al. 2019a).

management that do not consider unintended negative consequences of a management action (including no action) can be devastating.

The rangeland ecology and management profession has developed and embraced non-linear management approaches for at least three decades. Non-equilibrium hysteresis models of plant community changes are an innovation of modern rangeland science (Chambers et al. 2019a). The concept that rangelands and forests can reach an always predictable Clementsian climax end-state that is entirely stable is incorrect. Ecological systems often exist in multiple stable states and may be moved out of the original configuration into new configurations from which it may be difficult or impossible for them to return to the original state (Figure 3.1). Hysteresis describes the phenomenon whereby a system might return to the original stable state by multiple pathways and not simply reverse the original path it took to move between the original state and the new state. The idea that these complex systems might enter a new stable state that is difficult or impossible to leave upon management perturbation is important for policy development and implementation. Once a plant community becomes a novel system, eradication of invasive species or restoration to an original ecological state may be an ecological or economic impossibility.

Some invasive plant species act as ecosystem engineers, changing the system processes, such as fire, in fundamental ways that provide the invader with a competitive advantage. Once a system has fundamentally changed and therefore become a novel ecosystem, returning it to a previous state may be impossible. For instance, soil microbiology may be difficult to restore to a pre-invaded state. It may be theoretically possible to restore the system entirely, but practically impossible. In some cases, there is a legislative directive or goal on the part of the land manager and managing agencies that requires a large investment. But in other cases, such an investment might not make economic or ecological sense.

An invasive plant management plan for local, regional, or national issues should have clear goals and consider costs and benefits. For example, complete eradication and restoration may be prohibitively expensive and not required to achieve the management goals. Additionally, local or state requirements may not align with management tactics required to achieve eradication. An example of areas that might be protected, regardless of cost, are National Park Service lands. This system boasts some of the most iconic natural areas in North America and these national treasures are legally mandated to be managed in an "unimpaired" or "untrammeled" condition for future generations. A significant investment is authorized to preserve them in their "original" stable state. However, other lands that have undergone invasive species colonization do not legally require similar investments in resources.

Three decades ago, Dewey et al. (1995) published a paradigm of weed management plans that considered the most practical way to address invasive plant species. That paradigm embodied the idea that large weed infestations can be controlled to allow important natural resource values to be conserved. And a logical corollary is that largely uninvaded natural areas should be protected from invasion. This idea has been adopted by the Western Governors Association in their 2020 "Toolkit for Invasive Grass Management in the West" (Anonymous 2020) and in the discussion of protecting the sagebrush steppe (Maestas et al. 2021).

An important control point in the effort to manage invasive plants is early detection and rapid response. It is increasingly important that citizens have the tools and training necessary to identify new plants in their surroundings. And both the tools and training are increasingly available. With widespread dissemination of information through live proceedings, webinars, blogs, social media, and other mass communication vehicles, Cooperative Extension at Land Grant Universities play a crucial role in delivery. It is critically important for municipal, county, state, and federal agencies to support both Cooperative Extension and other public and private efforts to forge an educated and aware public.

Chapter 4 – Committees and Councils with Invasive Plant Management Responsibilities

Numerous councils, associations, and committees are involved in informing and implementing policy focused on invasive plants that affect wildfires. The connection among all organizational levels (local, state, regional, national) is important to ensure that information flows among associations, committees, and workgroups. The Wildland Fire Leadership Council and the Western Weed Coordinating Committee are examples of committees that tie together these organizational levels. But in most cases the connection across the different levels of organization occurs through the network of councils, committees, and associations. It is therefore imperative to maintain these networks and connections.

National committees or associations with weed management responsibilities include the National Invasive Species Council (NISC), Federal IPM Coordinating Committee (FIPMCC), Federal Interagency Committee for the Management of Noxious and Exotic Weeds (FICMNEW), and the Wildland Fire Leadership Council (WFLC). These information-sharing committees consist of federal agencies and departments and foster collaboration and make recommendations for national and regional management activities. The North American Invasive Species Management Association (NAISMA) is a network of pest management professionals who implement management programs to prevent the detrimental impacts of invasive species. The Early Detection and Distribution Mapping System (EDDMapS) is a web-based mapping tool used to document the distribution of invasive species across the United States.

Examples of regional committees and associations focused on weed management include the Western Weed Coordinating Committee (WWCC), National Institute of Food and

Agriculture (NIFA) Multistate Research Projects, and Western Association of Fish and Wildlife Agencies (WAFWA). These committees and working groups function to facilitate communication and collaboration among federal, state, and local entities in both the public and private sectors. The University of Wyoming Institute for Managing Annual Grasses Invading Natural Ecosystems (IMAGINE) develops workshops and symposia to empower informed decision making by landowners and managers and demonstrate that long-term, landscape-scale management is possible. It also hosts the Northeast Wyoming Invasive Grass Working Group.

Several national and regional committees are focused on developing biological options for weed control. These include the Technical Advisory Group for Biological Control Agents of Weeds (TAG-BCAW), which provides guidance and recommendations to researchers and regulating agencies on the use of biological control organisms, and the Western State Classical Weed Biological Control Coordination Community (WSCWBCC), which comprises representatives from state and federal agencies and enables information sharing.

Nearly all states have State Invasive Species Councils (ISC), Invasive Plant Councils (IPCs), Exotic Pest Plant Councils (EPPC), Cooperative Weed Management Areas (CWMA), and Cooperative Invasive Species Management Areas (CISMA) that address invasive plants, including those that affect wildfires. A listing of state invasive plant councils can be found at the National Association of Invasive Plant Councils website (na-ipc.org). Many of these state and local councils are focused on education and eradication efforts.

Professional science societies, such as the Invasive Plants Committees for the Weed Science Society of America and the Western Society of Weed Science, form ad hoc committees to address invasive species issues. These ad hoc committees serve to enhance communication and collaboration among invasive species groups/committees and to develop timely invasive species symposia and other opportunities in invasive species awareness.

In addition to the standing committees and councils listed above, ad hoc committees and workgroups are often formed to focus on a specific target species. The composition of these committees varies but often includes researchers, educators, and practitioners from the university, state and federal agencies, local non-profit organizations, and tribes who can provide information about local habitat, ecology, use patterns, monitoring, and control methods. State invasive plant council members assist by identifying management goals. Ranchers, land managers, and industry representatives help to set priorities, leverage resources, and improve visibility. Officials from federal and state regulatory agencies should be included if there is a regulatory component. Members from other state and federal agencies leverage existing networks to synergize work and raise funds and provide technical assistance to ranchers and private land managers. It may also be useful to have a social scientist on the team to assist with evaluation of outcomes.

Chapter 5. Integrated Pest Management for Invasive Plants Affecting Wildfire

5.1. Prevention

Prevention remains the most economical way to deal with invasive plants (Farrar et al. 2015, National Invasive Species Council 2016). Early detection and rapid response is a fundamental strategy to prevent establishment of invasive plants (Research Group, 2014). The U.S. Department of Agriculture (USDA) Animal and Plant Health Inspection Service (APHIS) Plant Pest Quarantine (PPQ) determines the risk of specific invasive species and provides invasive species priority lists (see https://www.aphis.usda.gov/laws-regs/animal-plant-diseases-pests-concern). The utility of these lists depends on information-sharing networks, and compatible data systems to facilitate the sharing of invasive species information are a priority of the National Invasive Species Council.

5.2 Monitoring

Managing invasive plants at the landscape scale relies on the ability to characterize the level of invasion and habitat degradation, allowing managers to classify areas from pristine with minimal invasion and largely intact ecological services to highly invaded with degraded ecological services. Monitoring the efficacy of control efforts is also critical to measuring management success.

Two mapping platforms for monitoring invasive annual grasses are the U.S. Department of Agriculture-Agriculture Research Service (USDA-ARS) Rangelands Analysis Platform (rangelands.app) and the U.S. Geological Survey Invasive Species Habitat Tool (gis.usgs.gov/inhabit). The mapping products available at the Rangelands Analysis Platform separate areas dominated by annual forbs and grasses from areas dominated by perennial forbs, grasses, and shrubs and therefore can delineate areas with largely intact ecological services. The Invasive Species Habitat Tool, or INHABIT, delivers species distribution model outputs to inform invasive plant species management at multiple spatial scales. Species distribution models generate spatial predictions of habitat suitability for species presence.

Current remotely-sensed products can be rapidly acquired but appear to provide less accuracy than ground surveys (Baur et al. 2022). There is a trade-off between the accuracy of the information and the cost and time required to collect it and this trade-off has important implications for rapid and wide-scale assessment.

5.3. Management Tools

The management tools discussed below are best deployed in concert to shift the competitive balance towards desirable plant species. Effective management of invasive plants might include targeted grazing, mowing, prescribed burning, biological control, herbicides, and competitive planting. Land managers and pest control practitioners need to understand how these tools can best be used together. In addition, management methods have tradeoffs in terms of efficacy, cost, environmental impacts such as negative impacts on non-target or desirable plant species, and acceptance by local communities.

Biological Controls

Classical biological control of weeds is the introduction of host-specific natural enemies or biocontrol agents, like arthropods, mites, and fungal and bacterial pathogens, to reduce and permanently stabilize the density of the target invasive plant below a damaging threshold. The biocontrol agents multiply and disperse by themselves and undergo thorough testing to ensure they will not attack non-target plant species. For landscape-level suppression or prevention of damage from an expanding invasion, classical biological control should be considered. If successful, it brings about desired ecological change over large areas without repeated costs or treatment of the entire infested area (Van Driesche et al. 2010).

Biocontrol can suppress targeted weed density associated with increased wildfire risk and has been particularly useful in suppressing targeted weed species that invade following wildfires. For instance, a population of dalmatian toadflax (*Linaria dalmatica*) that dominated a site following wildfires in Southern California experienced a 99% reduction in percent cover after the introduction of the biocontrol agent *Mecinus janthiniformis*. Moreover, annual and perennial grasses increased, annual forbs remained abundant, and perennial forbs and shrubs remained rare (Smith et al. 2021). Similarly, rush skeletonweed (*Chondrilla juncea*) that invaded after a wildfire near Mountain Home, Idaho, was suppressed on all but approximately 10% of the original infestation due to the combined impact of a rust fungus (Puccinia chondrillina) and a gall mite (*Aceria chondrillae*) (Milan 2022).

Biocontrol has the potential to play an important role in the management of weeds that are changing wildfire regimes, but a lack of investment in weed biocontrol research and development has resulted in limited progress for new weed biological control programs. Despite funding challenges, surveys by Dr. Brian Rector (USDA-ARS) and collaborators in the Balkan nations and adjacent regions have yielded four candidate biocontrol agents for invasive annual grasses known to impact wildfire regimes (Rector, 2020).

Cultural Control

Livestock like cattle, sheep, and goats can provide an ecological service by reducing the biomass of undesirable plant species and thereby reducing fine fuels and shifting the competitive balance towards desirable plant species (Briske 2012, Schachtschneider et al. 2024, Xu et al. 2026). But for grazing to be an effective strategy for weed management, it must be the primary aim of a grazing program. It is not clear that grazing allotments focused on production and weight gain, using standard stocking rates and grazing windows, achieve weed management goals (Derner et al. 2022). For weed management, high-density grazing with sheep or cattle for short durations in the spring is an effective strategy for managing medusahead, but variations in spring weather can limit the impact of grazing (Kyser et al. 2014). Fall cattle grazing on cheatgrass can significantly reduce the presence of cheatgrass seed in the seed bank in the Great Basin by removing fall cheatgrass standing biomass that serves as a protected site for cheatgrass germination and growth (Perryman et al. 2020).

Virtual fencing could facilitate the use of grazing to manage invasive plants by reducing the costs and increasing the flexibility of livestock placement. Virtual fencing has been in use for more than two decades for companion animals and appears to work with many different types of grazers, including cattle, sheep, goats, and others. The fencing appears to have limited impact on the environment and can provide additional information on animal performance and behavior. Although the cost of the technology is still relatively high – $10,000 to $12,000 for the base unit and collars sold on a subscription basis for $40 to $55 (Johnson 2024, Hoag et al. 2024) – these costs will likely drop substantially over time. However, recent work has shown that virtual fencing may not be the best option where predation is a concern. O'Connor (2024) demonstrated that predation on livestock can be minimized by physical fencing, removing and composting carcasses, and range-riding on horseback.

There are benefits and drawbacks to the use of grazing. Grazing can be incorporated with chemical controls to achieve better control of target weeds, as was seen by Lenhoff et al. (2019) where intensive sheep grazing in May and herbicide applications in fall (glyphosate, imazapic, and rimsulfuron) and spring (glyphosate and imazapic) effectively reduced cover of cheatgrass. However, changing environmental conditions impact the logistics of having the right number of animals available at the right time. Additionally, the lack of suitable water sources on many rangelands can limit the ability to distribute livestock effectively. And grazing in threatened or endangered species habitat may not align with management goals.

Prescribed burning can be used effectively to control plant species that tend to suppress fire (Briske 2012; see Chapters 2 and 8). Brazilian peppertree is not adapted to frequent fires and can be managed using fire if stands are not too dense. Prescribed fire, integrated with other control tactics such as grazing or chemical control, can be used to manage fire-adapted species. Also, chemical applications following a prescribed fire, when invasive plant species sprout before desirable species, can effectively reduce the seed bed loading of non-desirable plant species.

Mechanical Control

Mechanical controls are often used for making fuel breaks, including those along roads. Additionally, mowing can be used to suppress invasive annual grass species and shift the competitive advantage to desirable annual grass species. For instance, mowing medusahead late in the growing season – when many desirable annual grasses like wild oat and soft chess have already set seed but medusahead has not – can reduce the seed bed loading and have multiyear effects (Kyser et al. 2014). Tillage can be a useful management tool in areas of non-desirable monoculture. Tillage can kill existing undesirable vegetation while preparing a suitable seedbed for revegetation as part of an integrated approach utilizing competitive seeding. However, in rangeland ecosystems dominated by desirable perennial species like woody shrubs, mowing or tillage are not effective tools. In addition, mowing and tillage can be challenging in areas with steep slopes because they can increase soil erosion, or in areas where rocks are present and striking rocks with steel mowing blades can cause fires. A cultural resource survey may be required for these types of activities.

Chemical Control

Chemical control can be used to reduce the spread of invasive plants that change wildfire regimes and thereby reduce the buildup of hazardous fuels, rehabilitate disturbed sites, and restore native and desirable plant communities (Mealor et al. 2013, Kyser et al. 2014, Davis and Mangold 2019, Bennion et al. 2025, Lazarus and Germino 2025). The choice of herbicide is based on target vegetation, efficacy, and specificity, and site characteristics, including the presence of desirable vegetation, management objectives and level of risk of the herbicidal formulations to human and environmental health.

Herbicides can be used on landscapes where mechanical or cultural methods cannot be deployed or where control is needed over large areas. Federal agencies responsible for plant management over large areas use herbicides for management, but these agencies are subject to additional requirements, including National Environmental Policy Act (NEPA), the Federal Land Policy and Management Act (FLPMA), and cultural resource survey requirements for ground applications.

Most state Cooperative Extension Services provide information on managing weeds including those in wildland areas using chemical controls. In California, information is available at weedcut-new.ipm.ucanr.edu, wric.ucdavis.edu, and ucanr.ipm.edu. In Montana, the Monthly Weed Posts (available at montana.edu/extension) provide chemical control options. The search tool used to pull information from various state Cooperative Extension sources and publications (Ask-the-expert) has been added to many Cooperative Extension sites such as the University of Florida Cooperative Extension site (available at edis.ifas.ufl.edu) and the Extension Foundation website (available at extension.org). Several handbooks published by state Extension Services are excellent resources for chemical control options (Mealor et al. 2013, Kyser et al. 2014).

Chapter 6. Restoring the Landscape Following Invasion and Wildfire: Techniques, Tools, and Best Management Practices

On landscapes where invasive plant abundance is high and desired vegetation is scarce, intense restoration actions may be necessary. These efforts often include revegetation – the intentional planting of desired vegetation. The goal of revegetation is to reduce the risk of soil erosion post-wildfire and re-establish vegetation that will compete with re-invading weeds and meet land management objectives. Without revegetation on degraded landscapes, invasive plants are likely to grow again and dominate the site, which then perpetuates the change in wildfire regime.

While revegetation sounds ideal in theory, it is difficult to execute and more often results in failure than success (Davis and Mangold 2019, Baur et al. 2022). Failure may be due to a variety of issues, including poor planning and execution by practitioners, but even if executed perfectly, seeded species are subject to harsh environmental conditions. These harsh conditions include, but are not limited to, unpredictable precipitation; seasonal drought; predation by rodents, insects, and birds; soil pathogens; and competition with invasive plants that grow from seeds in the soil seed bank. Additionally, revegetation can be very expensive; it often requires multiple passes over a landscape to control invasive plants, prepare the soil for planting, and finally plant seeds of desired species. Seeds can be expensive and hard to source, especially if the end goal is re-establishment of native plants whose seed can cost over a hundred dollars per pound when supplies are severely limited (National Academies of Sciences 2023).

Given the importance of revegetation on degraded landscapes and the high risk of failure, executing revegetation with the best techniques, tools, and practices is extremely important to improve the likelihood of success. Successful revegetation begins with careful planning. Site assessment, determining goals and objectives, controlling invasive plants, planning the seeding, site preparation and implementation, monitoring, and long-term management are all critical steps (Orloff et al. 2022). Site assessment includes knowing the climate, soils, landscape position (elevation, slope, aspect), existing vegetation, and accessibility of the area for equipment to carry out the seeding. Conducting the site assessment is crucial for choosing appropriate species for seeding and best methods of seeding them. Short-, mid- and long-term goals and objectives should be considered and match what a site can support based on the site assessment. For example, landscapes with low precipitation and shallow soils will have low productivity and limited livestock grazing capacity. They may also be more susceptible to re-invasion by fire-adapted invasive plants and require more aggressive control measures and seeding with desired, introduced species versus native species.

After a perturbation (natural or man-made), regeneration of invasive plants from the underground propagules (seeds, stolons, rhizomes, etc.) is likely. Therefore, controlling invasive plants is an important step in revegetation, and herbicide application is the most common method to do so. An advantage of chemical control is its efficiency in that it can treat a large area quickly with a high degree of efficacy. The type of herbicide to use and the timing of application depends on which invasive plants are present, but application within weeks to months post-fire is common with repeated applications often required. While chemical control is efficient, seeded species and existing vegetation can be injured by the herbicide. Herbicides vary in how long they persist in the soil, from no persistence for a herbicide like glyphosate to several years for a herbicide like indaziflam. While persistence can be beneficial for invasive plant control, persistence may require a delay in seeding or creative application of multiple herbicides to avoid injuring seeded species. For example, indaziflam is highly effective at controlling invasive annual grasses by killing seeds as they germinate in the soil, and its persistence in the soil results in multiple years of annual grass control. However, areas treated with indaziflam cannot be seeded for at least 24 months because the herbicide is not specific to seeds of annual grasses. Sequential seeding may be a way to integrate seeding with herbicides of varying persistence. In sequential seeding, an effective but less persistent herbicide like imazapic or glyphosate is used prior to and shortly after seeding of desired species; once desired species have grown one or two years and have acquired a root system deep enough to be tolerant of indaziflam, indaziflam can be applied to provide long-term control of re-invading annual grasses.

Technologies aimed at protecting seeded species from persistent, soil-active herbicides are also emerging. For example, herbicide protection pods (HPPs) encase seeds in activated carbon that protects the seed from a herbicide's toxic effect. This allows for simultaneous application of herbicide and seeding instead of delaying seeding for months to years, during which time invasive plants may become dominant once again. This technology has been tested in greenhouse studies and in limited field studies with variable outcomes. Further testing and development are needed.

Planning a revegetation seeding includes determining what species to include in a seed mix, a seeding rate, and a seeding method and timing. Species selection will be site-specific and based upon the site assessment and land management goals and objectives. Species that establish quickly reduce soil erosion and compete with invasive plants. Field observations suggest including five to 10 species in the mix; mixes with more species often result in only the five to 10 most competitive species establishing and persisting. The species in the mix should include both short- and long-lived species and species with variable growth forms (both deep and shallow-rooted species, and species that grow early and late in the growing season). Introduced species are generally cheaper and easier to establish than native species, and introduced species are generally more competitive against re-invading invasive plants. However, introduced species may not meet long-term management objectives. Seeding is best accomplished with a no-till drill seeder, but broadcast seeding either by ground or aerially is often employed, especially if the landscape is hilly or rocky. As for timing, seeds are often planted during the autumn immediately after a wildfire. Seeds remain dormant in the soil over winter and emerge the following spring. Spring seeding is more difficult to implement due to less predictable weather and soil conditions, but when possible, it has sometimes resulted in higher success than fall seeding. Planning a seeding can be complex with many factors to consider that vary from site to site. Field offices of the USDA-Natural Resources Conservation Service have resources to assist with planning a seeding at the local level.

Site preparation and implementation of seeding are also important steps during revegetation. An overarching goal of this step is to prepare the site so that there is good contact between seeds in the seed mix and soil at the site. Most burned areas require little to no site preparation because the fire has cleared the site of debris, and ash from the fire helps cover and retain seeds.

Monitoring a landscape that has been revegetated involves periodically assessing vegetation to see if it is on track to meet management goals and objectives. Seeded species may take several years to establish, and short-term results are not always indicative of long-term outcomes. In a revegetation study in a wildlife management area in southwestern Montana, seeded grass biomass increased from about 30 kg/ha two years after seeding to nearly 2,000 kg/ha 15 years after seeding; at the same time, the invasive plant spotted knapweed (*Centaurea stoebe*) decreased by 86% (Rinella et al. 2012). Similar trends were seen at a site in Utah where revegetation after wildfire occurred 16 years earlier (Ott et al. 2019).

After a successful revegetation plan has been executed, long-term management is important to maintain seeded species and prevent or reduce re-invasion by invasive plants. Invasive plants are often a symptom of poor land management. Returning to the same management post-revegetation may degrade the site once more. If livestock have access to a site that has been revegetated, grazing management is particularly important. Grazing typically needs to be deferred until seeded species are well-established, usually after two growing seasons. After that, grazing management should be designed to encourage desired species and deter invasive plants.

In addition to careful planning and then using the best techniques, tools, and practices, being strategic about where to revegetate is of utmost importance. Historically, indiscriminate revegetation following wildfire was common practice. However, following wildfires some portions of a landscape are more disturbed than others and some are more resilient than others. For example, fuel breaks and fire lines are often the most disturbed parts of the landscape. Research and observation have shown that these areas are particularly prone to invasion and soil erosion. Fuel breaks and fire lines are ideal for revegetation because the level of soil disturbance is high, and invasive plant seeds may have been moved around or introduced into the area on construction equipment. For example, research from rangeland in Montana showed that the invasive annual cheatgrass was three to four times more abundant on a fire line created by a bulldozer than it was in the burned area or adjacent non-burned area three years after the fire (Seipel et al. 2018), and this effect was still evident nine years after the fire. Fuel breaks and fire lines should be prioritized for revegetation to slow establishment and spread of invasive plants and reduce soil erosion. If time and money are limited, revegetate fuel breaks and fire lines first.

Chapter 7. Case Study: Invasive Plants Add Fuel to California's Fires

California is floristically and ecologically diverse. It supports the greatest number of unique native plant taxa of any state and encompasses a respectable 11% of the "Level IV" ecoregions that occur in the continental U.S. (Griffith et al. 2016). Most of California's ecoregions have a Mediterranean-type climate that is characterized by winter rains and warm, dry summers. With a few exceptions (primarily the Mojave, Sonoran, and Colorado deserts), these regions are also fire-adapted. Although debate continues over regional historic fire patterns and the effects of fire suppression on wildfire intensity across the state, current fire regimes are clearly more impactful to both humans and ecosystems than they were in the past. Regime changes are associated with increased human ignitions, frequent and extreme drought events, altered vegetation, altered disturbance patterns, and importantly, invasive plant species (D'Antonio and Vitousek 1992, Brooks et al. 2004, Syphard et al. 2017, Weise et al. 2023).

The costs associated with wildfires can be staggering. In 2018, wildfires in California cost nearly $150 billion, or the equivalent of about 1.5% of the state's annual gross domestic product (Wang et al. 2021). Not captured in this estimate are the severe impacts to native plant communities and wildlife habitat. Shrubland habitats in the dryer regions of California can undergo "type conversion" to invasive plant-dominated landscapes favored by – and prone to – wildfire, creating in some cases a highly stable vegetation community dominated by exotic annual grasses (D'Antonio and Vitousek 1992). In desert environments where fire was historically absent but is now present due to fuel from annual grasses, type conversion can be even faster (Underwood et al. 2019). The recent Dome Fire in the Mojave Desert provides a stark example of the devastating ecological impacts that even a single fire, carried by invasive annual grasses, can have to a habitat that is intolerant to fire (Boxall 2020, Los Angeles Times).

California's legislature, agencies, parks, and people are in a heightened state of awareness of the risks of wildfires and the need to manage them. The Legislature's 2021 "Blueprint for a Fire Safe California" proposed 11 bills that address varying aspects of wildfire prevention and management, clearly underlining the level of urgency that wildfire management has reached in the state. California's Wildfire and Forest Resilience Action Plan (California Forest Management Task Force 2021) lays out several actions, including vegetation management, to address fire spread and defensible space, improve wildfire preparedness, and reduce risk. Although several types of invasive plants are known by both researchers and land managers to foster fire in California, they are not yet formally highlighted as a specific target for wildfire management funding.

Wildfire management is especially complicated in California in part because of the diversity of habitats and natural fire regimes that occur within the state. There are distinct differences in fire regimes and fire-fostering invasive plants in wetter northern California climates and dryer southern California climates.

Northern California

In both lowland rangelands and in high-elevation semiarid systems, medusahead is considered to be an ecosystem transformer that acts as a fire promoter by creating fuel corridors that facilitate fire spread. Medusahead has invaded the high desert sagebrush steppe in the northeast corner of California, the western foothills of the Sierra Nevada range, and the coastal ranges north of San Francisco Bay (Calflora 2023). Medusahead has become dominant in some areas where perennial grasslands and naturalized annual grasslands serve as important rangeland for California livestock production (Nafus and Davies 2017). It now occupies more than a million acres of annual-dominated grassland, perennial grassland, oak woodland, and chaparral communities, where low-intensity, spring surface fires were once common as a component of indigenous land management practices. Management options in annual grasslands include prescribed and cultural burning, while pre-emergent herbicides and establishment of shaded overstories may be more effective in some perennial grasslands and woody plant communities (Nafus and Davies 2014).

In forested systems, the role of invasive plants as drivers of change is less clear and is interlinked with tree die-offs, tree pests, and drought. Invasive perennial shrubs such as gorse (*Ulex species*), French broom (*Genista monspessulana*), and Scotch broom (*Cytisus scoparius*) establish and spread quickly in open areas after a fire, reproducing from the soil seed bank and resprouting from root crowns. A study from shrubland habitat in Chile found that French broom produced more biomass, more fine fuels, and was more flammable than two co-occurring invasive and native shrubs (Garcia et al., 2015). The authors concluded that while brooms may not in themselves increase fire frequency, they do likely exacerbate wildfire's negative ecological effects. Because they burn easily, mature broom plants can also be controlled in some situations with well-timed late season burning.

Southern California

Invasion by highly flammable invasive plant species, combined with increases in human-caused ignitions and physical disturbance, have altered fire regimes across much of the shrubland and desert habitat of southern California (Keeley and Fotheringham 2001); Syphard et al. 2017). They have created an "invasive plant fire regime cycle" that can lead to type conversion to annual grasslands, a more rapid accumulation of fuels after fire, and consequently, a shorter fire return interval (Brooks et al. 2004; Figure 7.1). Increases in dry nitrogen deposition from smog in otherwise nutrient-poor systems have further favored the spread of invasive annual grasses that can germinate and grow more quickly under nutrient-rich conditions, literally adding fuel to the fire (Padgett and Allen 1999).

Recent remote-sensing efforts have estimated that non-native annual grasses now occupy more than 10% of the land area across large sections of the Western United States and have concluded that cover greater than 10% poses a significant wildfire threat to dryland shrub systems (Pastick et al. 2021; Dahal et al. 2022). Importantly, annual grasses, which provide significantly more fine, dry, and continuous fuels than do perennial bunch grasses and forbs, are presumed to have been a very minor component to southern California plant communities prior to European settlement. Exotic annual grass cover estimates from long-term monitoring plots in coastal sage scrub habitat in Orange County show high average cover of invasive annual grasses, ranging from 25% to 60% across coastal to inland sites over the last decade (Kimball et al. 2018, e.g., Figure 7.3). Ripgut brome (*Bromus diandrus*) and red brome (*Bromus madritensis*) are two of the most common invasive annual grasses in coastal sage scrub and chaparral systems. These systems are also highly impacted by several other invasive species that favor fire-prone environments, such as black mustard (*Brassica nigra*), Russian thistle (*Salsola species*), and fountaingrass (*Pennisetum setaceum*) in uplands, and giant reed (*Arundo donax*) in riparian areas.

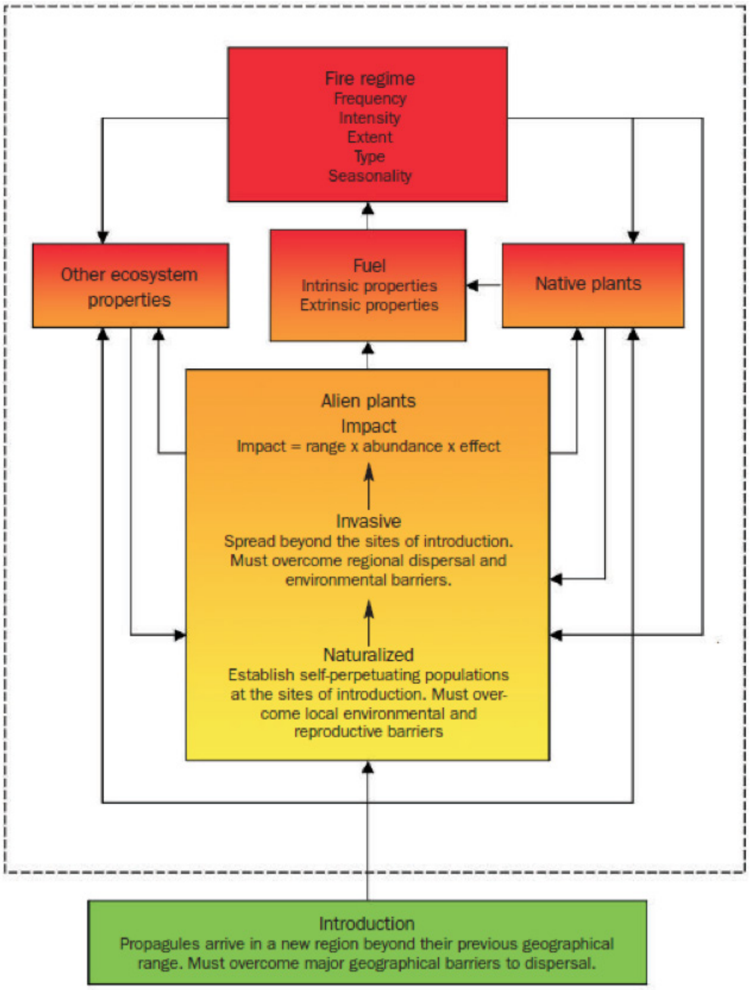


Figure 7.1. The invasive plant fire regime cycle, adapted from Brooks et al. 2004.

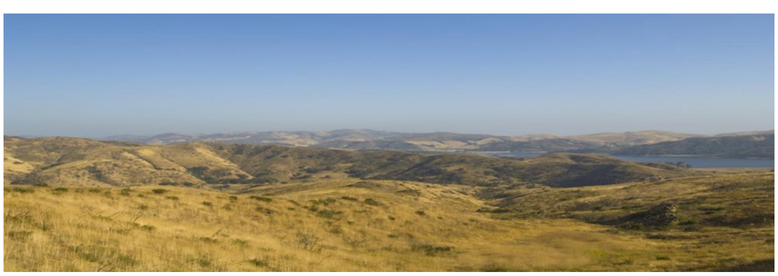


Figure 7.2. A degraded landscape in Southern California. Picture courtesy of Stephen Francis.

Coastal sage scrub and chaparral shrubland systems are experiencing higher ignition rates and, where fire is frequent, are converting from shrub-dominated vegetation communities to annual grasslands. These fire-mediated changes in vegetation lead to higher rates of erosion, increased exotic species invasion and higher fire hazard as perennially flammable fine grass fuels replace shrubs. Continued high fire frequencies in southern California also threaten the viability of plant and animal species that require longer fire-free periods. High-profile examples of such species include the federally listed California gnatcatcher (*Polioptila californica*) and Tecate and Cuyamaca cypress (*Hesperocyparis forbesii, H. stephensonii*).

The Mojave and Sonoran deserts further inland are not adapted to any frequency of fire and consequently have suffered especially serious impacts when fires do occur. Red brome, cheatgrass, and *Schismus* spp. and their persistent thatch have filled gaps between existing vegetation and created a flammable environment between perennial succulents and shrubs where historically only ephemeral wildflowers occurred.

Saharan mustard (*Brassica tournefortii*), stinknet (*Oncosiphon pilulifer*), and stinkwort (*Dittrichia graveolens*) are three newer broadleaf annual invaders to southern California that also leave behind fine, persistent thatch that serves as fuels and proliferate with frequent fire (Hedrick and McDonald 2020). Their direct effects are still largely unknown, but they are likely to only further add to habitat degradation, flammability, and habitat type conversation.

Management Actions

The key to addressing fire-promoting invasive plants and the complex mosaic of problems associated with them is strong partnership and coordination, both in planning and on-the-ground implementation. Both fire and invasive plants are landscape-level phenomena that know no property boundaries. Various working groups, such as Fire Safe Councils, have been established across the state to address wildfire preparedness and response (Everett and Fuller 2011). Increasingly, they are also confronting the exacerbating issues that invasive plants pose to wildfires, though most actions associated with them are generalized as "vegetation management." As yet, annual grasses have only rarely been addressed, in part due to a lack of available tools to manage them effectively at scale.

The National Park Service is working region-wide with the California Invasive Plant Council and resource and operations managers to develop best management practices for wildfire response preparedness. These best management practices will specifically help to reduce the spread of invasive plants during response operations. They include identifying clean sites for Incident Command Posts, incorporating invasive plant occurrence data into a national online Wildland Fire Decision Support System (WFDSS) tool, adding weed wash stations for wildland fire responders, and incorporating standard language regarding invasive plants into WFDSS for fire responders. The East Bay Regional Parks District in the San Francisco Bay area is implementing fuels reduction efforts that target invasive plant removal rather than all vegetation irrespective of origin. The Santa Monica Mountains Recreation Area in Los Angeles County is working with researchers and fire authorities to test different methods to reduce invasive annual grass thatch in fuel breaks (Wells 2022). The Natural Communities Coalition, Orange County Fire Authority, and Irvine Ranch Conservancy in Orange County are implementing roadside hardening projects that include selective vegetation treatments and planting native succulent vegetation that serves as a living fuel break and reduces invasive plant cover (Fry, unpublished data). California Invasive Plant Council has organized workshops with the National Park Service and other agencies on wildfire preparedness and invasive plants.

California faces the challenge of having both highly flammable wildlands and an extensive wildland-urban interface. These factors make anthropogenic ignitions more likely and increase costs associated with wildfires due to structural damage. Legislation was recently passed to facilitate controlled burns that reduce fuel loads and to reduce wildfire insurance rates for homeowners and businesses in high-fire-risk communities who take proactive fire-risk reduction measures. Ongoing state-led support for ecologically sensitive vegetation management strategies focusing on invasive plants will be key to mitigating the human, economic, and environmental impacts of wildfires across California's diverse habitats.

Chapter 8. Case Studies from the Southeastern U.S.

Cogongrass (*Imperata cylindrica*)

Cogongrass has been touted as one of the world's worst invasive species because of its interaction with fire. Native to southeastern Asia and tropical eastern Africa, cogongrass prefers warm, wet-weather ecosystems (Terry et al. 1996; EDDSMap 2021). This species is a primary concern in seasonally wet pine savannas of the southeastern U.S., where it reduces native species diversity and ecosystem function (Daneshgar and Jose 2009). As a rhizomatous, perennial grass that resprouts quickly after being top-killed by fire (Dozier et al. 1998), it maintains 60% of its biomass belowground, including fire-resistant roots and rhizomes (O'Brien et al. 2010). Seeds have a high germination rate but do not persist in the seedbank. Seedlings need competition-free areas to establish successfully (Dozier et al. 1998). Thus, cogongrass vegetatively spreads furthest immediately after fire, especially in frequently burned habitats (Lippincott 2000). Furthermore, Kato-Noguchi (2022) found that cogongrass might reduce the abundance of mycorrhizal colonization in the roots of neighboring plants as well as suppress the germination and growth of some native species. As a superior competitor that is allelopathic and benefits from fire (Estrada and Flory 2015), cogongrass is a nemesis to native plants and land managers alike.

Researchers have also discovered that wildfire occurrence is higher where cogongrass has invaded, and the fires spread over larger areas (Fusco et al. 2019). Flory et al. (2022) found that cogongrass invasions resulted in 65% greater fuel loads, increasing flame lengths and fire intensity, which caused high tree mortality, especially under drought conditions. Several studies have shown that cogongrass increases fire intensity via two mechanisms. First, it produces more biomass and greater fine fuel loads than co-occurring natives (Platt and Gottschalk 2001; Tomkat-Kelly et al. 2021). Cogongrass is taller than many of the bunchgrasses typically found in pine savannas, increasing the horizontal and vertical continuity of fuels (Stocker and Hupp 2008). In fact, cogongrass has been documented to increase fire temperatures by 140% or more (Lippincott 2000). Second, it has flammability traits that result in higher heat of combustion and shorter time to ignition (Cardoso et al. 2018). This results in higher fire temperatures and longer fire resident times. Cogongrass is not negatively affected by the higher fire intensity. Mortality after fire has been demonstrated only when temperatures were abnormally high and the fire resident time was long (Bryson et al. 2007).

Cogongrass benefits from the post-fire environment, which makes it even more critical for managers to begin controlling populations before burning. The higher fire intensity causes greater mortality of neighboring species and opens a competition-free area for cogongrass to proliferate and spread. Moreover, low N:P ratios are common in soils after wildfires and are conducive to cogongrass growth (Butler et al. 2021). In the field, cogongrass can be hard to distinguish from the many native grass species found in the open grassy understories of frequently burned habitats (Kirkman and Jack 2017). When populations are small or not in flower, they are less likely to be recognized and treated (Stone and Andreu 2022). One approach to cogongrass control is maintaining high cover and biomass of native species, mainly native grass species (Daneshgar and Jose 2009). Although cogongrass proliferates excessively belowground via a rhizome network and competes strongly for nutrients (Terry et al. 1996), functionally similar species can resist invasion (Daneshgar and Jose 2009; Just et al. 2017). In addition, cogongrass does not perform well in closed canopy systems but thrives under the partial canopies of pine savannas (Brook 1989).

Successful management of cogongrass requires killing belowground rhizomes. Even a tiny fragment of a rhizome can establish in a new area and spread quickly. This requires an integrated management plan that combines deep discing and herbicide application, as discing alone will not control cogongrass (MacDonald 2004). If timed properly, these treatments can also be combined with prescribed burning. For instance, Jose et al. (2002) documented a decrease in cogongrass after applying herbicides post-fire in the fall and then discing the following spring. Discing combined with herbicide is the most effective long-term control of cogongrass (Dozier et al. 1998). Still, repeated herbicide application for two or more years is typically necessary for eradication (Jose et al. 2002). Because young cogongrass shoots are sensitive to imazapyr and glyphosate, these herbicides are often the first line of defense against invasion. Other researchers have found that coupling glyphosate and prescribed fire was slightly more successful than glyphosate alone (Enloe et al. 2013). Planting overstory trees might also benefit control as closed canopies, regardless of patch size, decrease cogongrass spread and establishment (King and Grace 2000; Ramsey et al. 2003). Whatever the treatment combination, it is likely that revegetation will be necessary following cogongrass eradication (Jose et al. 2002).

Brazilian peppertree (*Schinus terebinthifolia*)

Fire suppression increases the likelihood that certain ecosystems are invaded by fire-inhibiting species (Mandle et al. 2011; Just et al. 2017). When fire frequency decreases, pine rocklands, coastal dunes, and salt marshes in the southeastern United States become more susceptible to invasion by non-native plants that suppress fires, such as Brazilian peppertree (Loope and Dunevitz 1981; Stocker and Hupp 2008). This invasive shrub is an aggressive spreader in the southeast and also in Texas, California, and Hawaii (EDDSMap 2021). Although peppertree is currently restricted to southern regions because of its low cold tolerance, we can expect it to spread northward to much of the Gulf Coast as our climate warms (Osland and Feher 2020). It can form dense monospecific stands, outcompeting native grasses and vegetation, and negatively affect natural areas as well as agriculture and cattle grazing (Ewel 1986). Because its leaves are relatively less flammable than those of other shrubs, it has been planted as fuel breaks in its native South American range (Kovalsyki et al. 2016).

Peppertree modifies its environment, facilitating its proliferation while suppressing native vegetation. This invasive shrub grows to about 10 m (33 ft) tall and has a short, woody stem hidden under its thick, intertwined branches. It outcompetes native plant species establishing near it, and crushed fruits below peppertree have been shown to decrease the germination of native seeds (Nickerson and Flory 2014). Its own seeds are dispersed by birds and mammals, which increase germination rates by removing the pulp around seeds as they pass through their digestive tracts (Ewel 1986; Dlamini et al. 2018). This mode of dispersal likely explains peppertree's widespread colonization. Furthermore, peppertree causes soil accretion in low-lying areas, increasing elevation in pine rocklands and marshes (Gordon 1998; Ewel 2013). Peppertree tends to spread on elevated, well-drained sites and does not survive well in low-lying areas with prolonged inundation (Dalrymple et al. 2003).

Peppertree also initiates a fire suppression feedback loop that enhances its invasion. Once established, this shrub is fire-tolerant thanks to the high moisture content of its leaves, branches, and litter (Meyer 2015; Stevens and Beckage 2010). In addition, peppertree litter has a high decomposition rate, leaving little fuel to carry fires (Doren et al. 1991; Mack and D'Antonio 1998; Grace et al. 2001). When adult peppertrees are girdled by fire, they have high survival, profusely resprouting from aboveground stems and root crowns (Woodall 1979; Grady and Hoffman 2012). Although crown fires can kill adult peppertrees, this fire type is unusual in dense invasions (Smith 1985; Doren et al. 1990). Fires do not typically carry well in peppertree stands (Meyer 2015), where fire temperatures have been recorded up to 200°C lower than in adjacent uninvaded areas (Stevens and Beckage 2009). Fire suppression by peppertree also protects its seeds, which are killed by high temperatures (>70°C/158°F) for prolonged periods (>1 hour; Nilsen and Muller 1980; Loope and Dunevitz 1981). Peppertree is sensitive to fires when young, with seedlings more susceptible to fire than adults (Florida Exotic Pest Plant Council 1997).

Because established, high-density peppertree stands suppress fires, prescribed burning is not an effective control method. When this invasive plant is at low density, fire mortality has been recorded as high as 30% to 45% (Stevens and Beckage 2009). Once established, even frequent fires will not stop its proliferation (Doren and Whiteaker 1990). Some researchers have posited that repeated fires with a fire return interval of four to five years could control low-density populations of peppertree, but intervals of more than eight years are unlikely to be successful, especially in the short term (Stevens and Beckage 2010). Furthermore, high fire frequency in invaded areas is likely to negatively affect some native species, including pines, while positively affecting peppertree individuals over 1 m (3.3 ft) tall (Loope and Dunevitz 1981).

Although repeated burning might slow new invasions, chemical and mechanical methods are frequently needed to control stands of peppertree, especially once they become dense (Doren and Whiteaker 1990; Ferriter 1997; Dalrymple et al. 2003). One option is to apply a herbicide containing triclopyr or glyphosate directly to the foliage (Panetta and Anderson 2001; Cuda et al. 2019). Managers may also cut down adult peppertrees and paint their stumps with herbicide, especially in wetlands where non-target species might be particularly vulnerable or water pollution is a concern (Gioeli et al. 2021). These conventional control methods are time-intensive and variably successful, resulting in control costs of over $3 million each year in Florida alone (Hiatt et al. 2019). Biological controls may also be a future method of reducing peppertree invasions. For instance, the release of thrips (*Pseudophilothrips ichini*) in Florida in 2019 had some success (Wheeler et al. 2022), and others, including a leaf galler (*Calophya latiforceps*), are still being tested (Cuda et al. 2023). Thus, there is hope that control of peppertree will become less intensive in the future.

Chapter 9: Wildfires and Plant Invasions in the Sagebrush Steppe-Great Plains Transition Zone of North America.

Historic Wildfire Regimes

The sagebrush steppe and prairie ecosystems of the North American Great Plains span the western and central United States and parts of Canada, exhibiting a marked decrease in precipitation from east to west (Lauenroth et al. 1999). This climate gradient reflects the rain shadow effect of multiple mountain ranges and is one of the main drivers affecting the distribution of plant communities (Figure 9.1; Epstein et al. 1996). The overall effect is a transition from the wetter eastern Great Plains to the drier western sagebrush steppe. Attempts to classify vegetation types across this region have been fraught with challenges, as the strong influence of gradual precipitation gradients and other climatic influences result in these ecosystems being highly transitionary (Coupland 1961; Omernik and Griffith 2014). Nonetheless, characterizing historic grass communities is especially relevant for understanding wildfire regimes, as grasses provide fine fuels that facilitate ignition and often regenerate quickly following a wildfire (Prior et al. 2017). The wettest and easternmost part of this region is characterized by dense stands of tall grasses like big bluestem (*Andropogon gerardii*) and switchgrass (*Panicum virgatum*) that thrive in the moist, fertile soils of the eastern Great Plains (Changnon et al. 2002). As precipitation declines westward, the vegetation shifts from a mix of short- and medium-height grasses, such as western wheatgrass, to a drier shortgrass prairie with sparser stands of low-growing grasses, including buffalo grass (*Bouteloua dactyloides*) and blue grama (*Bouteloua gracilis*), which are well adapted to arid conditions (Lura et al. 2019). Finally, in the driest and westernmost part of the region (just east of and interspersed amongst ridges of the Rocky Mountains and Sierra Nevada), the sagebrush steppe is dominated by sagebrush (*Artemisia tridentata*) and other drought-tolerant shrubs with bare ground and bunch grasses scattered throughout (Shinneman 2020).

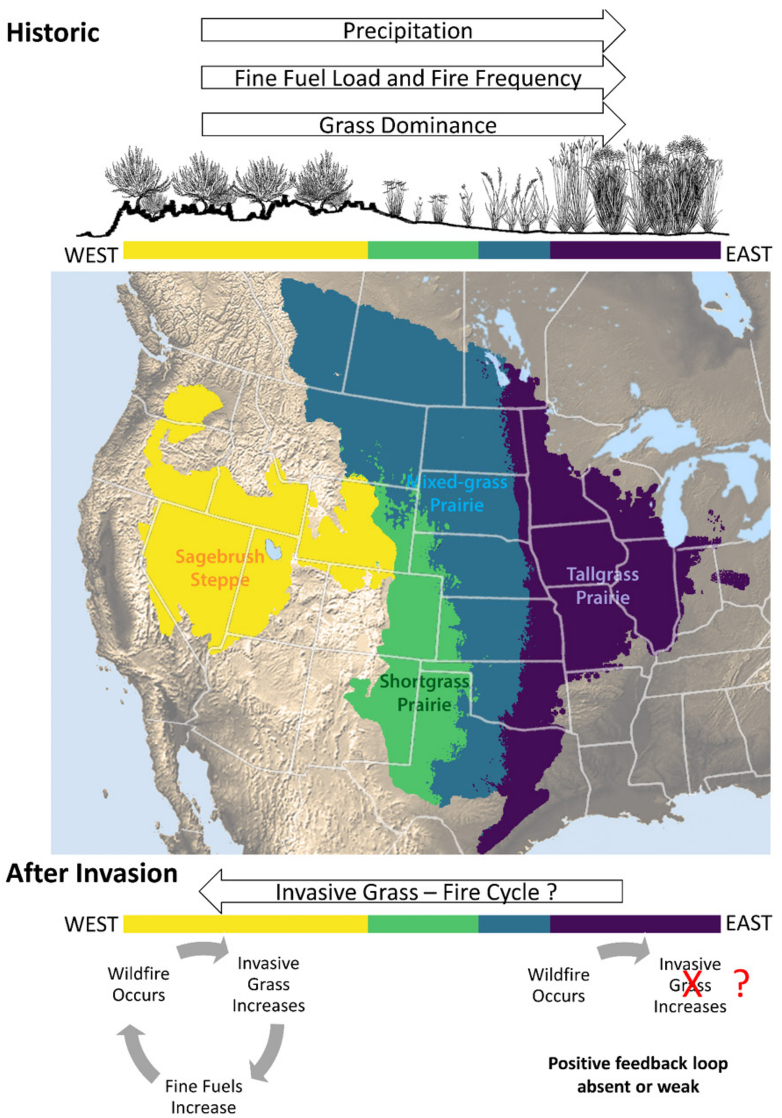


Figure 9.1: Map and hypothetical latitudinal cross section through the sagebrush steppe–Great Plains transition zone of western North America, with historical climate and vegetation gradients shown alongside their corresponding changes in fire frequency (top), and possible spatial differences in the strength of the invasive-grass fire cycle (bottom). Question marks denote uncertainty and the need for more research to confirm these trends.

Historic wildfire regimes were greatly influenced by these vegetation patterns, with wildfire frequency being positively correlated with fuel loads (above-ground biomass), with differing fuel loads imparted by different plant communities (Prior et al. 2017; Figure 9.1). Wildfires were historically most frequent in the eastern part of the region, despite its generally wetter climate, because periodic droughts were common and provided ideal conditions for ignition (Guyette et al. 2012). Throughout the Great Plains, the abundant fine fuels provided by grasses after a dry spell allowed fires to occur every one to 35 years, maintaining the dominance of fire-adapted grass species and preventing the establishment of woody plants (Epstein et al. 1996; Ratajczak et al. 2014). As the vegetation transitions to the drier, sparser plant communities in the west, fuel loads required to sustain wildfire take longer to accumulate, thus decreasing the frequency of fire. The sagebrush steppe, which has the sparsest vegetation, is dominated by woody shrubs at least partly because fire return intervals were historically quite long, with some areas enduring up to 350 years between wildfires (Bukowski and Baker 2013). Additionally, high shrub cover can suppress grass cover, therefore reducing the easily ignitable fine fuels that grasses provide. However, fires in the sagebrush steppe could be intense when they occurred, owing to the presence of woody biomass that allows fires to burn longer and hotter.

Wildfire Regimes After Invasion

The invasion of non-native grasses and alteration of wildfire regimes across the western United States have garnered much attention, especially within the sagebrush steppe biome (Smith et al. 2023). Numerous studies have confirmed a positive feedback loop between wildfire and non-native grass invasion (D'Antonio and Vitousek 1992; Fusco et al. 2019). Known as the "invasive grass-fire cycle," this process has led to the conversion of millions of acres in western North America, where the invasion of grasses increases the amount of readily ignitable fuels on the landscape, raising the likelihood of wildfires (Smith et al. 2023). After a fire, non-native grass populations are often thought to regenerate more rapidly than native species and become more abundant, driving a transition to landscapes dominated by invasive annual grasses and perpetuating a more flammable environment. Hundreds of thousands of acres of sagebrush are burned and replaced by invasive grasses each year, contributing to the quickly diminishing spatial extent of the sagebrush biome, which threatens over 300 species of conservation concern as well as livestock grazing areas (Doherty et al. 2022). Sagebrush is naturally slow growing with tenuous seedling survival, and attempts to restore these ecosystems post-fire have had a low success rate in many areas without costly interventions (Shriver et al. 2019).

Despite confirmation of the invasive grass-fire cycle in many parts of the world, evidence from the transition zone between the Great Plains and sagebrush steppe suggests that this cycle may not be ubiquitous across the region. Research in this region suggests that invasive grass abundance does not always increase after wildfire. In fact, prescribed fires may be an effective tool to suppress invasive grasses and promote native plant communities in areas where sagebrush populations are already lost or where their maintenance may not be a conservation goal (Symstad et al. 2021; Vermeire et al. 2011). For example, despite being the most infamous driver of the grass-fire cycle in more western ecosystems, cheatgrass was no more likely to be found within burned areas than outside of them in northeastern Wyoming (Porensky and Blumenthal 2016). Similarly, regular fire reduced the richness and abundance of invasive plants while promoting native grass dominance in a Kansas tallgrass prairie (Smith and Knapp 1999). However, it is important to note that cheatgrass cover in these study areas was generally low prior to fire (<10% in the Porensky and Blumenthal study), and Symstad et al. (2021) observed less predictable control outcomes after prescribed fire if cheatgrass cover was higher before treatment (>20%).

A possible explanation for this natural interruption to the invasive grass-fire cycle likely involves a complex interplay between climate, fire, and the evolutionary history of native species (Chambers et al. 2019b). Taylor et al. (2014) analyzed multiple sites across the western United States and found that cheatgrass responded most negatively to fire in areas with comparatively low temperatures and high summer precipitation. They further suggest that historical climate and fire regimes may have shaped native species compositions, favoring fire-adapted native grasses that can recover quickly after disturbances, thereby reducing the likelihood of cheatgrass responding positively to fire. Thus, fire-adapted native communities become increasingly common towards the east and may regenerate at least equally as well as invasive species after a fire (Figure 9.1). While this pattern may not result in plant communities completely resistant to annual grass invasion, it appears that the complete conversion to near-monoculture states after fire is less likely in the eastern prairies. This contrasts with drier areas further west, where native communities are less fire-adapted, allowing cheatgrass to increase in abundance and largely exclude native perennial vegetation.

Management Implications

The aforementioned research emphasizes the need to improve our understanding of how the invasive grass-fire cycle may be interrupted – both naturally and through management interventions. While management interventions to break the grass-fire cycle often concentrate on invasive species control or fire suppression, findings towards the central and east part of this region suggest that bolstering the native community should be given more focus and that dominance by invasive annual grasses after fire is not a foregone conclusion. Analyzing the potential tradeoffs of using invasive species control tools after fire has become especially important with the recent application of indaziflam, a pre-emergence herbicide that provides effective multi-year suppression of invasive annual grasses, in rangelands and natural areas. For example, Alba et al. (2024) showed that applying indaziflam successfully controlled invasive grasses after fire, but it also suppressed short-lived native species that primarily reproduce via seed. This is significant because short-lived, seed-dependent species naturally dominate early successional plant communities after large disturbances such as fire, and suppression may influence the successional trajectories of plant communities (Tilley et al. 2022). While invasive annuals may be suppressed, use of this tool after fire may also lead to depletion of native seed banks (Courkamp et al. 2022) so that plant community recovery relies on the post-fire persistence of mature perennial plants to promote re-vegetation. Thus, land managers must recognize and work within this set of tradeoffs. This presents a significant management challenge in this region and requires answers to questions we cannot yet answer, the most central being: will invasive annual plant abundance increase at this site after fire if we do nothing? Assuming the answer is "yes," based on our current understanding of the invasive-grass fire cycle, applying indaziflam may lead to a net negative effect due to impacts on the native seedbank.

Summary

Plants are ecosystem engineers that attempt to shift the competitive balance in their favor. One way the plants do this is by changing wildfire regimes. But changes in wildfire frequency and intensity impact human and environmental health, and therefore we must consider managing these species to minimize these impacts. Management goals, cost, ecosystem degradation, and likelihood for success must be considered when developing a management plan. There are numerous committees, councils, and organizations devoted to informing and implementing policies and management strategies focused on invasive plants, and the network of these committees and councils spans the different organizational levels (local, state, regional, national). Numerous management tactics exist and these should act in concert to shift the competitive balance towards desirable plant species. Restoration can be critical to long-term success, but this aspect remains a significant challenge. Long-term monitoring of restoration efforts is key to documenting success. Finally, different regions of the country face different challenges in managing their invasive plant problems and how these plants affect wildfires locally. It is clear that information from the local level on population and community ecology, including life histories, soil seed banks, level and frequency of disturbance, etc. can be a key to success at regional and national scales. Although not discussed at length here, strong and energetic local leadership is important in this effort. Informed local leadership is a critical partner in these efforts and support for these local efforts can be important for success.

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