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Chapter 7

Applying Ecological Concepts to the Management of Widespread Grass Invasions

Carla M. D'Antonio, Jeanne C. Chambers, Rhonda Loh, and J. Tim Tunison

Abstract The management of plant invasions has typically focused on the removal of invading populations or control of existing widespread species to unspecified but lower levels. Invasive plant management typically has not involved active restoration of background vegetation to reduce the likelihood of invader reestablishment. Here, we argue that land managers could benefit from the ecological principles of *biotic resistance* and *ecological resilience* in their efforts to control invading plants and restore native species. We discuss two similar but contrasting case studies of grass invasion that demonstrate how these principles can be applied to control and management. In seasonally dry Hawaiian woodlands, management of invasive fire-promoting grasses has focused on seeding native species that are resilient to fire disturbance and can coexist with grasses. Resistance to grass invasions appears to be weak in unburned native habitats. Thus, the focus of management efforts has been to increase resilience of the native vegetation to inevitable disturbance. We contrast this with the Great Basin of the western USA where the annual Mediterranean grass, *Bromus tectorum*, also has promoted an increase in fire frequency in shrublands and woodlands. Here, a three-tiered approach has been employed in which preventative management in the form of fire or fire surrogates is used in the initial stages of invasion to increase the resilience and resistance of the native herbaceous vegetation. In transitional stages where *B. tectorum* is well established but not dominant, mechanical or herbicide treatments are used to open up dense and senescing shrub canopies, thereby increasing vigor of native perennial herbaceous species through competitive release. The released competitors (perennial grasses) are then assumed to provide resistance to *B. tectorum* invasion. Following complete *B. tectorum* dominance, the focus of management is intensive seeding of native species to create resistant plant communities that reduce the likelihood of reinvasion.

Keywords Biotic resistance • Cheatgrass; Ecological resilience • Ecological restoration • Exotic grasses • Fire • Grass/fire cycle • Great Basin • Hawaii • State and transition models

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7.1 Introduction

While the control of some plant invasions may be as simple as removing a small founding population, more often control is part of a larger vision for the ecological management and eventual restoration of a natural area (D'Antonio and Meyerson 2002; D'Antonio and Chambers 2006). Ideally, attributes of a "fully restored ecosystem" as described by the International Society for Ecological Restoration (http://www.ser.org/) are applicable to the management of widespread invasions in the context of ecosystem restoration rather than simple species removals. One of these attributes states that the restored ecosystem should be "self-sustaining and able to persist under existing environmental conditions." Using this as a guiding principle, long-term control of nonnative species should involve the creation of systems that are resilient to future disturbance and resistant to reinvasion. Thus, two ecological concepts that are particularly critical to long-term management of plant invasions are *biotic resistance* and *ecological resilience*.

In this chapter we explore and contrast two case studies in which the concepts of resistance and resilience play different roles in the management of specific plant invasions. We focus on two examples of invasion by fire-promoting grasses, one in Hawai'i and the other in the Great Basin region of the Western USA. These two systems were chosen because together they provide a range of insights into how the ecological principles of resistance and resilience can be applied to the management of persistent or recalcitrant plant invaders. We focus on grass invaders because they are widespread and difficult to control and because they often dramatically alter ecosystem structure and functioning. Thus, efforts to manage them or the communities of which they are now a part are critical to maintaining native biodiversity and ecosystem functioning in some regions.

7.1.1 Ecological Resistance and Resilience

Since the rise in interest in the field of biological invasions, community ecologists have explored the role of ecological resistance as an ecosystem property influencing invasion success (e.g., Rejmanek 1989; D'Antonio 1993; Maron and Vila 2001; Levine et al. 2004). Despite the extensive recent history of research in this area (reviewed by Levine et al. (2004) and Bruno et al. (2005)), land managers have been slow to embrace the concept into their approaches to *weed* control. Instead, weed invasions in natural areas have been controlled largely by chemical and mechanical means, or through prescribed burning and grazing. Such approaches do not explicitly engage ecological principles but focus instead on a *top-down* approach where the manager is acting as a predator removing the species from the system (McEvoy and Coombs 1999). This approach has been successful at eradication of invaders in many discrete isolated areas. Typically, however, little attention is given to whether the post-removal community has higher resistance to further weed invasion and increased resilience to future disturbance.

Resistance in an ecological sense refers to the ability of a community to withstand encroachment by nonnative species. An overriding conclusion from the literature on ecological resistance is that resistance is a probabilistic phenomenon (D'Antonio et al. 2001b), varying across community types for a given invader species (e.g., D'Antonio 1993) and among invader species within the same community type. Likewise, the relative importance of abiotic vs. biotic mechanisms of resistance varies across landscapes and among years within the same landscape. Thus, some communities are inherently more susceptible to invasion than others while communities on the resistant end of the spectrum can be highly susceptible to invasion after disturbance or a stress that opens up space in the community (Davis et al. 2000). Despite the variation in resistance over space and time, experimental studies have demonstrated that background vegetation plays an important role in suppressing reinvasion of target weeds after initial top–down control (McEvoy and Coombs 1999). D'Antonio and Thomsen (2004) have argued that ecological resistance should be a more important part of invasive plant management than it is currently.

In contrast to ecological resistance, *resilience* describes the ability of a community to return toward its predisturbance, and presumably preinvasion, state after a disturbance. It is a restoration goal because high resilience in a restored site means less hands-on input is required by managers to keep vegetation within target bounds. With relevance to invasive species, high resilience of native or desired species will potentially reduce the need for immediate (postdisturbance) and future control of invaders and for reseeding or replanting of native species after disturbances. Hence, high resilience increases the likelihood of resistance to invasion, and both should be considered as management goals.

7.1.2 Fire-Enhancing Grass Invasions

The invasion of natural areas by introduced grasses is a widespread phenomenon (e.g., D'Antonio and Vitousek 1992), and grasses are disproportionately represented on virtually all published lists of high-impact natural area invaders (Daehler 1998). Grasses can be difficult to control and manage for several reasons: (1) They frequently have excellent mechanisms of resilience reestablishing from buds, roots, or seedbanks after disturbance. (2) Many disperse readily across the landscape either through effective passive dispersal or through attachment to animals. (3) They are not easily controlled through classical biological control. (4) They can transform ecosystems through their interactions with fire regimes (D'Antonio and Vitousek 1992; D'Antonio 2000; Brooks et al. 2004). (5) They often play conflicting roles in landscapes because within the same region they can provide forage for livestock while also promoting fire and/or loss of native species. As a result consensus for control is difficult to achieve.

In the Hawaiian Islands, perennial grasses from several other continents became widely established by the mid 1900s (Smith 1985). Within the dry and mesic parts of the islands they have been found to enhance the occurrence of fire (Mueller-Dombois 1981; Tunison et al. 2001). This has occurred largely to the detriment of native species (Hughes et al. 1991), although there is some variation in the severity of fire impacts across environmental gradients (D'Antonio et al. 2000). Some of the perennial grasses are resilient to fire because of rapid resprouting from basal root crowns (Smith 1985; D'Antonio et al. 2001a). Others, however, are killed by fire but regenerate rapidly from seed (Tunison et al. 1994, 1995; D'Antonio et al. 2000, 2001a).

Nonnative, fire-enhancing grasses have also invaded xeric and semiarid portions of the mainland USA with large-scale ecological consequences. The most widespread of these is *Bromus tectorum* or cheatgrass, an annual species from the eastern Mediterranean region. *B. tectorum* was widespread throughout the intermountain western USA by the 1930s (Mack 1986) and was associated with widespread increases in fire frequency and size and declines in native species a few decades later (e.g., Whisenant 1990). This species increases fire frequency and size by increasing the homogeneity or horizontal continuity of fuels and the rate of fire spread across what is otherwise a patchy shrub-steppe ecosystem that experiences summer dry lightning (Link et al. 2006). Because of rapid spring growth and early maturation, *B. tectorum* plants typically produce seeds prior to the fire season. Populations recover rapidly after fire via dormant seeds not killed by fire and very high seed output due to increased resource availability in the years following fire. Many of the native shrub species in these ecosystems are killed by fire, and in areas with depleted herbaceous understories, *B. tectorum* can rapidly dominate the ecosystem (e.g., Whisenant 1990).

7.1.3 Chapter Overview

In this chapter, we use two case studies to emphasize a way to incorporate ecological concepts into the management of persistent plant invaders. The example of grass invasions in Hawaii that we describe is a case study of a situation in which ecological resistance plays little role in the planning and implementation of control and revegetation of grass-invaded, fire-prone ecosystems. In Hawaii Volcanoes National Park's drier lowlands and mid-elevation habitats, exotic grasses have invaded virtually every place they could have invaded, and degradation and invader seed sources are widespread. Management is focusing on creating resilient native plant assemblages that can coexist with the invasive grasses particularly after fire has already occurred. Technically this is a restorative activity rather than ecosystem restoration (sensu Jordan 2003). Nonetheless, the goal is to create communities with some of the desired attributes of a restored ecosystem (http://www.ser.org/). By contrast with the Hawai'i example, grass invasions in the Great Basin of the USA are not complete and some potentially resilient native communities still exist. In sites that have not converted to complete *B. tectorum* dominance, management is focusing on maintaining native resilience while increasing resistance to invasion. In vast areas of the Great Basin, however, native communities have been replaced with *B. tectorum* grasslands. In these areas management is focusing on restoration of native communities that will be resistant to further invasion by the introduced grasses.

7.2 Case Study I: Hawaiian Dry and Submontane Seasonal Environments

7.2.1 Study System

The Hawaiian Islands are characterized by large environmental gradients driven by the prevailing trade winds and the volcanic shield topography. Dry forests and shrublands exist on the leeward side of the high islands while wet forests occur on the windward sides of the high islands. In contrast to the strongly varying microclimates, soil chemistry across the islands is relatively constant with all soils ultimately deriving from basaltic lava or ash. The soils are typically nitrogen limited when young, colimited by nitrogen (N) and phosphorus (P) at intermediate ages (10,000–100,000 years), and P limited on the older surfaces (Vitousek 2004).

Grasses from other regions of the world have invaded virtually all of the many microclimates in Hawaii from dry coastal terraces receiving <20-cm precipitation to sites with >3 m of rainfall annually. A detailed list of grass invaders is not available but most species are listed in the flora of Hawaii (Wagner et al. 1999).

7.2.2 Grass Invasion Impacts

While introduced perennial grasses are widespread throughout the Hawaiian Islands, those that have invaded dry shrubland and open woodland habitats appear to be causing the most dramatic changes because they change the continuity and density of fuels (Mueller-Dombois 1981; Smith 1985; Smith and Tunison 1992). The most ecologically significant grass invaders on Hawai’i Island include species from both the new and old world. Three of these are (1) *Pennisetum setaceum* (fountain grass), a perennial bunchgrass from Africa, (2) *Melinis minutiflora* (molasses grass), a mat-forming grass from Africa, and (3) *Schizachyrium condensatum* (bushy beardgrass), a perennial bunchgrass from Central America. *P. setaceum* has invaded the leeward side of Hawaii Island from sea level up to 3,000 m. It competes with native vegetation (Cabin et al. 2000, 2002) and regrows after fire. In mid-elevation environments that have experienced fire, native species richness has declined due to the lack of ability of native species to compete with *P. setaceum* and their relatively slow regrowth after fire (Shaw et al. 1997). Although *P. setaceum* is an ecologically important invader and a large new project has been initiated to assess ways to reduce its impact (S. Cordell, Institute of Pacific Island Forestry, personal communications), we will not discuss that work here. We will instead focus on management of habitats invaded by *M. minutiflora* and *S. condensatum*.

M. minutiflora was introduced to the Hawaiian Islands as livestock forage (Parsons 1972). It invades mesic shrubland and open woodland ecosystems where it promotes an increase in fire occurrence and intensity (Tunison et al. 2001). Individual *M. minutiflora* are typically killed by fire, but the abundant seedbank is a source of resilience and high, rapid postfire seed production of first-year plants results in rapid recovery of *M. minutiflora* populations (D'Antonio et al. 2001a). *M. minutiflora* co-occurs with *S. condensatum* in many natural areas. This latter species is from Central and South America where it is not typically associated with fire-prone ecosystems. It readily invades submontane forests, competes with native vegetation before fire (D'Antonio et al. 1998), and resprouts rapidly after fire (D'Antonio et al. 2001b). Its relative, *Andropogon virginicus* (broomsedge), from the southeastern USA, also occurs across these environments and is also associated with increased fire frequency (Tunison et al. 2001).

Within Hawaii Volcanoes National Park, fires are associated primarily with volcanic activity and humans. Fire regimes began to change after the spread of *A. virginicus*, *S. condensatum*, and *M. minutiflora* within park boundaries in the 1960s. Tunison et al. (2001) documented that fire frequency has increased by threefold and fire size by sixtyfold since the establishment of these grasses. D'Antonio et al. (2000) analyzed impacts of grass-fueled fires on native species diversity and cover in 19 sites from the coastal lowlands to the upper submontane seasonal zone of the Park and found the strongest impacts in the submontane zone where the prefire dominants are intolerant of fire. These dominants include the native tree, *Metrosideros polymorpha*, and the native shrub, *Leptecophylla tameiameia* (formerly *Styphelia tameiameia*, Wagner et al. 1999). In the coastal lowlands *A. virginicus* and *S. condensatum*-fueled fires regenerate toward at least some native species although native diversity is reduced (Tunison et al. 1994). *M. minutiflora*-fueled fires in both the coastal lowlands and submontane seasonal zone greatly reduce native species.

By sampling burned vs. unburned forests across the same elevation and rainfall within the submontane zone, Friefelder et al. (1998) documented that the homogenous structure of the grass canopy in burned sites resulted in an approximately threefold increase in wind speeds than were found above the canopy of unburned forests (with grasses). This resulted in modeled fire spread rates that were 20 times higher than those in unburned forest with exotic grasses in the understory. Mack et al. (2001) and Mack and D'Antonio (2003a, b) documented extensive changes in productivity, microclimate, and nitrogen cycling in burned compared with unburned woodland. They found that by greatly altering species composition including the elimination of native woody species, fire created an ecosystem that has much lower primary production and is much leakier for nitrogen. For example, Mack et al. (2001) report that net primary production is reduced by 55% in burned sites compared with nearby unburned counterparts, but that annual net nitrogen mineralization rates are an order of magnitude higher in burned sites. The lack of primary production during periods of high soil N mineralization resulted in periods with high potential for loss of nitrogen through leaching or trace gas losses (Mack, unpublished), a result corroborated by lower recovery of an N15 tracer added to plots within each site (Mack et al. 2001).

In addition to these ecosystem changes, the elimination of native species due to fire changes the substrates upon which nitrogen fixation, and thus ecosystem N accretion, occurs. Native plant species with symbiotic nitrogen fixation are rare in these unburned woodlands. Instead nitrogen fixation occurs via asymbiotic fixation in association with the litter of native species (Ley and D'Antonio 1998; Mack et al. 2001). The primary substrates for fixation are leaf litter of the dominant unburned tree species *M. polymorpha* and the organic (O) layer of the soil in unburned sites. Because *M. polymorpha* is eliminated by fire and the O layer is greatly altered and no longer fixes N, the capacity of these systems to fix N decreases by two orders of magnitude. This could significantly alter ecosystem development since these sites are on young volcanic soils low in nitrogen (Mack and D'Antonio 2003a).

The almost complete replacement of native woody species with invasive grasses in the submontane seasonal zone should have a profound influence on wildlife composition. However, no studies have been done to document such impacts. Furthermore, native birds are rare in intact native forests below 1,200 m because of the prevalence of introduced avian malaria. Nonetheless, we have observed two species of native birds, the Amakihi (*Hemignathus virens*) and the Apapane (*Himatione sanguinea*) in our unburned sites. By contrast we have not observed any native bird species in the burned sites.

7.2.3 Difficulty of Control and Management

By the early 1970s, feral goats had extensively browsed the woody vegetation in the coastal lowlands and submontane woodlands of this region. After their removal in the 1970s, grasses invaded virtually all of the coastal lowland and submontane shrublands and forest with at least some soil above the lava bedrock (T. Tunison, personal observation). By the 1990s, roughly 80% of the submontane seasonal zone and lowland shrub/grasslands had experienced at least one grass-fueled fire (Tunison et al. 2001). The primary management strategy for the ecosystem was fire suppression achieved through restricting human access to sites during the dry season and immediate fire suppression when ignitions occurred. Chemical and mechanical control of grasses was only being done for *P. setaceum*, which is restricted in its distribution within the park. *M. minutiflora, S. condensatum*, and *A. virginicus* were considered too widespread for any sort of chemical or mechanical control. Use of grazing animals, none of which would be native to this environment, was not considered a viable management strategy.

Restoration of the prefire native community was not considered to be a viable management strategy because most of the prefire dominants are not resilient to fire. Surveys of sites that had burned once but not burned again for 20 years demonstrated that only one of the prefire native shrubs, *Dodonaea viscosa*, regenerates over time in these ecosystems (Hughes et al. 1991). A resurvey of these sites in October 2007, unburned since 1973, confirms the lack of recovery of native species (D'Antonio et al., unpublished). The prefire dominant tree, *M. polymorpha*, grows very slowly as a seedling and it along with *L. tameiameia*, the dominant shrub, cannot tolerate competition from the grasses (Hughes and Vitousek 1992). In addition, they would readily be killed by the next, inevitable fire. For these reasons, it was decided that any restoration efforts toward these species would be futile.

Surveys of burned sites in the coastal lowlands (Tunison et. al. 1994) and our own and other observations (Tunison et al. 1995) suggest that there are fire-tolerant species in the Hawaiian flora that can be found in these ecosystems. Their scarcity throughout these sites is considered in part to be a function of previously high goat browsing, which is known to have reduced many woody species (Mueller-Dombois and Spatz 1975; Loope and Scowcroft 1985). Only one of these native species, *D. viscosa*, was common in burned sites (Hughes et al. 1991; D'Antonio et al. 2000). However, along the roads leading to these sites, burned individuals of planted trees showed regeneration from seed and stumps (Tunison et al. 1995, and personal observation). Also, the occurrence of other native dry forest and shrubland species in fire-prone habitats higher on Mauna Loa and Mauna Kea volcanoes suggested that there are fire-tolerant species in the Hawaiian flora that could grow at these sites. We hypothesized that to persist successfully in these habitats, a native species would have to tolerate growing with the dense exotic grasses that form an almost continuous canopy within 2–3 years after fire, and they must have mechanisms of resilience to regenerate rapidly after the inevitable fire. The success of the introduced grasses in invading intact woodland (D'Antonio et al. 2001a) suggested that no matter what plant communities exist on the sites, they are likely to offer little resistance to grass invasion. Thus, we designed a revegetation program that focused on developing fire-resilient native species assemblages that can coexist with grasses and persist with the new disturbance regime. It was unlikely that resistance could be strong enough to significantly dampen grass invasion in the near term especially since *S. condensatum* regenerates rapidly from root crowns after fire and can reduce the growth of native species (D'Antonio et al. 1998).

Our approach to the management of these sites is appropriately termed ecosystem rehabilitation (Bradshaw 1997). Ecosystem restoration implies the return of the composition and functioning of a system to the predisturbance state (Jordan 2003). Efforts to manage plant invasions in these sites focus on the process of resilience to restore ecosystem functions to the sites, with little understanding of whether resilience was a feature of pre exotic-grass ecosystems. It is possible that we are restoring aspects of composition that occurred prior to the extensive goat grazing that occurred before botanical records of the area were kept, but we do not know this.

7.2.4 Current Approaches to Management

In January 1993, we conducted our first experimental burn within Hawaii Volcanoes National Park with a goal of reestablishing more native species by reburning an already burned area heavily dominated by *M. minutiflora* and *S. condensatum*. The site had burned originally in 1972 and had few native species (Hughes et al. 1991; D'Antonio, unpublished). We started by introducing seeds of two species that were fire-tolerant and perhaps fire-enhanced, *D. viscosa* and *Sophora chrysophylla*. The latter is a small, N-fixing tree known to support native bird species. We seeded plots both before fire and after fire to evaluate whether fire would stimulate or inhibit their germination. We also seeded adjacent plots of identical age (previous woodland) that we did not burn in order to evaluate whether restoration could be achieved in a previously burned, exotic-grass-dominated site without reburning it. We followed seedling emergence and survival over the next 4 years. This initial fire was a low-intensity fire consuming only 50% of the aboveground grass biomass (Mack et al. 2001). The grass layer regenerated very quickly both from seed (*M. minutiflora*) and from resprouting individuals (*S. condensatum*) greatly limiting the window of time for native species to establish. Nonetheless, we found that both *S. chrysophylla* and *D. viscosa* germinated and grew within the burned plots (Fig. 7.1). The former species also germinated and grew in the unburned plots but individuals did not reach as large of a size. In the burned plots *S. chrysophylla* individuals reached flowering age within 5 years. Fourteen years later some of these individuals were very large (Fig. 7.2) and had seedlings of their own species growing nearby, suggesting that further recruitment had occurred. *D. viscosa* established at a lower rate than *S. chrysophylla* perhaps because its seeds benefit from higher intensity fire events (Tunison et al. 1995). Despite this, *D. viscosa* establishment was higher in the burned than in the control plots (Fig. 7.1). Some individuals of this species persisted throughout the monitoring time and they too reached reproductive maturity and were still present after 14 years (Fig. 7.2).

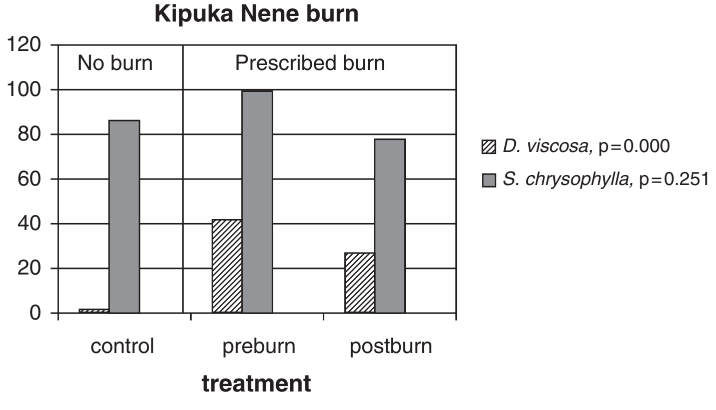

Fig. 7.1 Example of results of revegetation burns in Hawaii Volcanos National Park. Shown here are results of seeding in the first controlled burn conducted for revegetation of grass-invaded, burned seasonal submontane habitats conducted by Tunison, D'Antonio, and Loh in January 1993. Data are individuals of either *Dodonaea viscosa or Sophora chrysophylla* that were >10 cm in height after 4 years. N = 5 per treatment. Error bars not shown but P values represent significant treatment affects in a one-way ANOVA for each species. "Unburned" plots were grass dominated and were like the burned plots that had burned originally in 1973

Fig. 7.2 Example of a mature S. *chrysophylla* tree 14 years after being experimentally seeded into a grass-dominated burned Hawaiian submontane seasonal habitat. A fruiting individual of a second seeded species, *D. viscosa* (a shrub), is at the base immediately to the left of the tree. The matrix in which they are growing is the African grass *Melinis minutiflora* that has dominated these sites since invasion and the first fire in the early 1970s (photograph by C. D'Antonio)

Our next experimental burns occurred in July 1995 in two patches of vegetation burned previously in 1972 that were also dominated by *M. minutiflora, S. condensatum*, and *D. viscosa*. In contrast to the previous burn, these experimental burns were of very high intensity, consuming 98% of aboveground biomass and 20% of root and soil organic matter in the upper 20 cm (Mack et al. 2001). Plots were again seeded with *D. viscosa* and *S. chrysophylla* both before and after fire and in unburned control plots. Establishment was generally higher in postburn compared with preburn seeding and declined after initial germination due to periodic drought (Loh et al., unpublished report). Nonetheless, the successful germination and early growth of *D. viscosa* and *S. chrysophylla* in the burned plots compared with the unburned control plots suggested that high-intensity reburning of grass-dominated sites could promote native species establishment.

Over the next 5 years, the National Park Resources Management Division under direction of J.T. Tunison and R. Loh conducted five more experimental burns. Six additional species were planted in the submontane zone to evaluate regeneration and persistence with the ever-present invasive grasses; several of them showed potential for postfire revegetation and fire tolerance (Table 7.1). Because of the funding limitations the results have been monitored only sporadically but they suggest that several species of native plants are suitable for postfire rehabilitation and will persist under the exotic grass-dominated site conditions. In addition to these

Table 7.1 Hawaiian dry forest plants that were utilized in postfire revegetation prescribed burn experiments in Hawaii Volcanoes National Park, seasonal submontane woodland in and shrubland fires

Plant	Hawaiian name	Family	Life form
Bidens hawaiiensis	Ko'oko'olau	Asteraceae	Shrub
Dodonaea viscosa	A'ali'i	Sapotaceae	Shrub
Myoporum sandwicensis	Naio	Myoporaceae	Small tree
Osteomeles anthyllidifolia	U'lei	Rosaceae	Shrub
Santalum paniulatum	Iliahi	Santalaceae	Tree, hemiparasite
Scaevola Kilauea	Naupaka	Goodeniaceae	Shrub
Sophora chrysophylla	Mamane	Fabaceae	Small tree
Sida fallax	Ilima	Malvaceae	Sub shrub

experimental burns, an accidental fire within the zone dominated by *S. condensatum* and *A. virginicus* occurred in 2000. Resources Management staff at Hawaii Volcanoes National Park used the approach recommended by this experimental burning program and actively seeded the burn with most of the suggested native species and other available as yet untested species. They also transplanted seedlings of species evaluated in the pilot burns (Loh et al. 2007). Approximately 900 acres were seeded and replanted with a total of 30 native species. Many of these had been studied in our experimental burns or in laboratory heating trials (Loh et al. 2007). Over 2.7 million seeds and 18,000 individual plants were placed into the burn. By 2004 eleven native species had reached reproductive maturity. These are still present as of October 2007.

7.2.5 *Ongoing Challenges and Unanswered Questions*

At least three of the native species tested showed a strong ability to establish after fire, and seeds of these species have been collected and stockpiled for postfire seeding when further wildfires occur. Several of the remaining species that responded at least somewhat positively to fire are harder to collect native seed from. Germination of three additional species tested only in heat trials in the lab was heat-stimulated (Loh et al., unpublished), suggesting that they could be useful, but all three are uncommon making it difficult to collect and store seed. Stockpiling of seed for future postfire seeding has therefore been limited both in species composition and amount. While postfire rehabilitation via native seeding is used in other portions of the western USA such as the Great Basin, the supply of native seed tends to be limiting and rarely can more than a small percent of burned areas be reseeded. Seeding rates are typically low and contribute to poor success. This is likely to be an enduring challenge in Hawaii Volcanoes National Park where seed sources for some species are sparse and technology for native seed production is poorly developed. Thus, although initial trials suggest that postfire seeding can lead to the successful establishment of native species, in reality restoration will be limited by availability of appropriate seed and will therefore likely be of limited species composition. Since 2006, R. Loh and staff have been establishing native plant seed *orchards* in various portions of the park to serve as a source of material for future restoration efforts. Their efforts will provide critical information on native plant propagation and possibilities for large-scale seed production.

Are the communities that are being created actually resilient to fire? This was a critical element of the initial argument for this approach to restoration/revegetation. Testing this, however, will require reburning of the *revegetated* sites. This has only been done for two small sites and it was observed that *S. chrysophylla* individuals regrew from root sprouts and *D. viscosa* individuals regenerated from the seedbank. We do not yet know how the other species would respond or over what range of fuel conditions populations of desired native species will show resilience.

On the leeward side of Hawaii, S. Cordell and others (Institute of Pacific Island Forestry, Hilo Hawaii, personal communications) are experimenting with *green stripping* as part of large-scale ecosystem rehabilitation in fire-prone shrublands and woodlands. Their goal is to prevent the spread of *P. setaceum* fueled wildfires into remnant patches of native dry forest that harbor rare species. Their approach focuses on finding species with fuel characteristics that will suppress grass growth and reduce fire spread rates. Although such an approach seems unfeasible in the vast stretches of grass-invaded submontane forests and shrublands in Hawaii Volcanoes National Park, it is being considered in areas where fire could spread from grass-invaded portions of the Park into a nearby residential subdivision. In addition, active fire suppression will help to slow the further loss of the prefire forests and shrublands. Special ecological areas with rare or unique species have been identified by Park personnel. Protecting these areas from fire is a high priority. This may be done through fire suppression, manual fuel reduction, or potentially green stripping if it can be effectively done.

Could the reestablishment of native species such as we have tried eventually lead to the establishment of some biotic resistance to exotic grass invasion? This is possible for *M. minutiflora* because its growth is sensitive to shading (D'Antonio et al. 2001a). However, *S. condensatum* and *A. virginicus* both tolerate a high level of shade so it would take a very dense shrub or tree canopy to reduce their invasion rate. Nonetheless, high densities of native shrubs could decrease the fine fuel biomass that accumulates with exotic grass invasion, potentially reducing fire intensity. Despite this, the long-term persistence of native species in these areas will rely more on creating a community of species resilient to fire than any ecological resistance that these communities might provide.

Are we restoring important ecosystem functions to these sites? Reestablishment of a heterogeneous plant canopy is a potential benefit of managing these grass invasions by rehabilitating these ecosystems with native woody species. As mentioned previously, Freifelder et al. (1998) showed that the homogenous exotic grass canopy promoted high wind speeds and therefore faster fire spread rates. The breakup of this homogenous canopy through the establishment of taller woody species could therefore help to slow the spread of wind-driven fires through the region. An additional benefit of rehabilitation with woody species is the possible use of these species by native birds. No research has yet been done on this question.

7.3 Case Study II: *B. tectorum* in the Sagebrush Steppe

7.3.1 Study System

The intermountain area of the western USA is an arid to semiarid region in which most of the precipitation arrives during the winter or early spring. High spatial and temporal variability in precipitation both among years and within growing seasons is a defining characteristic. Nutrient availability, especially nitrogen, is typically low, but increases with elevation (Alexander et al. 1993; Dahlgren et al. 1997), and closely tracks moisture availability (Evans and Ehleringer 1994). High topographic variability results in strong gradients in both soil water and nutrient availability. These gradients determine the distribution of species and ecological types within the region. Sagebrush (*Artemisia* species) is the most abundant shrub with the subspecies *A. tridentata wyomingensis* dominating areas with effective moisture of 20–25 cm (8–12 in.) and *A. tridentata vaseyana* dominating areas with higher effective moisture of 30–41 cm (12–16 in.) (West 1983). For sites in moderate to high ecological condition the associated species are predominantly perennial grasses with lesser amounts of annual and perennial forbs.

Settlement of the region around 1860 by European Americans resulted in major changes in vegetation structure and composition of sagebrush communities and increased their susceptibility to invasion by exotic species (Mack 1986; Knapp 1996). Initially, widespread overgrazing by cattle and sheep led to decreases in native perennial grasses and forbs (Miller and Eddleman 2001). The decrease in the herbaceous species reduced the necessary fine fuels for carrying natural fires and altered competitive relations in favor of woody species. As a result, *Artemisia* species generally increased in dominance.

B. tectorum, cheatgrass, was accidentally introduced into the region at several different locations in the late 1800s (Mack 1986). The annual grass rapidly spread into the depleted sagebrush ecosystems, especially the warmer and drier *A. tridentata wyomingensis* shrubland types (Mack 1986). The fine, more continuous fuels contributed by *B. tectorum* resulted in more frequent and larger fires (Whisenant 1990; Knapp 1996). In many parts of the region an annual grass–fire cycle now exists in which fire return intervals have decreased from about 60–110 years to as little as 3–5 years (Whisenant 1990). It has been estimated that areas dominated by *B. tectorum* covered a minimum of 20,000 km^2 or 5% in the 1990s (Bradley and Mustard 2005) with an additional 150,000 km^2 at high risk of conversion (Suring et al. 2005).

7.3.2 Grass Invasion Impacts

Invasion by *B. tectorum* rapidly alters many ecosystem properties. *B. tectorum* dominance can alter nutrient cycling and soil microbial communities even in the absence of fire (Evans et al. 2001; Hawkes et al. 2006). Fire-induced community changes following invasion can lead to reduced soil water recharge and reduced soil moisture patchiness (Obrist et al. 2004). In addition, although invasive annual grasses can stabilize topsoil, loss of vegetative cover following fires or other disturbances increases overland flow and surface erosion resulting in the loss of soil nutrients, siltation of streams and rivers, and increased susceptibility to flooding (Knapp 1996). At regional scales repeated fire and progressive increases in annual grasses can result in conversion of shrublands and woodlands from carbon sinks to carbon sources (Bradley et al. 2006). Large-scale change in land cover from diverse shrublands to homogenous grasslands potentially can influence the region's albedo affecting evapotranspiration and, ultimately, moisture transfer, convective activity, and rainfall (Millenium Ecosystem Assessment 2005). The net effect could be an increase in aridity of the region.

Conversion of sagebrush communities to annual grasses results in increased landscape homogeneity and decreased patch diversity. A growing number of sagebrush-obligate species are at risk due to habitat loss (Knick et al. 2003), and approximately 20% of the ecosystem's native flora and fauna are already considered imperiled (Center for Science, Economics and Environment 2002). More frequent fires associated with cheatgrass invasion are resulting in increased costs for land management agencies due to increased fire suppression and land rehabilitation costs (US Department of Interior, BLM 1999, 2000). Local communities benefit from money spent for fire suppression, but can suffer from wildfires due to loss of livestock forage and property, health, and safety risks caused by smoke and particulate matter, and reduced recreational value and income.

7.3.3 Difficulty of Control and Management

Control of *B. tectorum* has been difficult because of its persistent seedbank, rapid response to disturbance, highly plastic seed production, and ability to compete with native species. Removal of herbaceous perennials in sagebrush communities can cause increases in both soil water and nitrate availability, conditions that promote *B. tectorum* growth (Chambers et al. 2007). Fire causes the death of fire-intolerant shrubs (Young and Evans 1978), and can result in greater soil water availability (Chambers and Linnerooth 2001) and dramatically increase soil mineral nitrogen (Stubbs and Pyke 2005; Rau et al. 2007), which can be up to 12 times higher in the postburn compared with preburn community (Blank et al. 1994, 1996). *B. tectorum* has the capacity for high growth rates (Arredondo et al. 1998) and can rapidly respond to increased availability of nitrogen (Lowe et al. 2002, Monaco et al. 2003) and soil water (Link et al. 1990, 1995). Biomass and seed production of *B. tectorum* can increase 2–3 times following removal of perennial grasses and forbs, 2–6 times after fire, but 10–30 times following both removal and burning (Chambers et al. 2007). Field and modeling studies show that *B. tectorum* populations have an 80–90% risk of exploding to densities near 10,000 plants m^{-2} within 10 years after fire (Young and Evans 1978; Pyke 1995).

7.3.4 Determinants of Resistance and Resilience

Evidence exists that the resilience of sagebrush communities and their resistance to *B. tectorum* are greatest on sites with relative high percentages of native perennial grasses and forbs. Long-term observations show that an inverse relationship exists between *B. tectorum* and total perennial herbaceous cover of sagebrush-steppe recovering from livestock grazing (Anderson and Inouye 2001) and of sagebrush semidesert responding to wildfire and livestock grazing (West and York 2002). Mechanistic research indicates that following overgrazing or fire, susceptibility to invasion by *B. tectorum* is lowest on sites with relatively high cover of perennial herbaceous species (Chambers et al. 2007). Under these conditions native perennials typically increase following fire, thus limiting growth and reproduction of *B. tectorum* (Chambers et al. 2007). Native species with similar growth forms and phenology, like *Elymus elymoides* (squirreltail), have the capacity to preclude or limit the establishment and reproduction of *B. tectorum* (Stevens 1997; Booth et al. 2003; Humphrey and Schupp 2004). Sites with low abundances of perennial grasses and forbs typically have reduced resilience following perturbations and, thus, are less resistant to invasion or increases in *B. tectorum*. The seedbanks of perennial herbaceous species, especially grasses, are typically small (Hassan and West 1986). Also, the seedlings of native perennial grasses are generally poor competitors with *B. tectorum* because the annual grass can germinate earlier in the fall and under colder winter temperatures (Aguirre and Johnson 1991). *B. tectorum* exhibits greater root elongation at low soil temperatures (Harris 1967) and is capable of competitive displacement of the root systems of native plants (Melgoza and Nowak 1991).

The ability to control *B. tectorum* or increase the resistance of sagebrush communities to its invasion varies in these topographically diverse ecosystems. The current distribution of *B. tectorum* indicates that while the species is abundant and widespread at lower elevations, invasion of high elevation *A. tridentata* systems has been minimal (Suring et al. 2005). *B. tectorum* exhibits relatively high germination at cold temperatures (Evans and Young 1972) and has considerable ecotypic variation in optimal night/day germination temperatures (Meyer et al. 1997; Bair et al. 2006). However, ecophysiological limitations due to cold temperatures can restrict its growth and, consequently, reproduction within *A. tridentata vaseyana* communities during short and cool growing seasons and in higher elevation mountain brush communities in general (Chambers et al. 2007). Precipitation, via its effects on available soil water, appears to be the primary control on *B. tectorum* invasibility when temperature is not a factor. High variability in available soil water at lower elevations may result in lower average native perennial cover and increased windows of opportunity for growth and reproduction of *B. tectorum* when available soil water is above a certain level (Chambers et al. 2007).

7.3.5 Current Management Approaches

The type of management approach or restoration activity used depends on the stage of invasion and the environmental characteristics of the affected communities (see Whisenant 1999; Chambers 2005; D'Antonio and Chambers 2006). Site prioritization depends on management goals and the need to maintain or improve habitat for a growing number of animal species obligate to sagebrush ecosystems, such as sage grouse (*Centrocerus urophasianus*) (Hemstrom et al. 2002). For the purposes of discussing management approaches for areas exhibiting *B. tectorum* invasion, we describe three ecological states (Figs. 7.3 and 7.4). In the first state, resilience is high for most *Artemisia* community types. The existing vegetation is managed to maintain or increase resilience to disturbance and resistance to *B. tectorum* invasion. Shrubs or trees may be increasing in abundance, but native herbaceous perennials are still a significant component of the community. *B. tectorum* may be present, but has relatively low abundance. Preventative management can be used to increase resistance by reintroducing disturbance in the form of fire or fire surrogate treatments (Wright and Chambers 2002; Chambers 2005; Miller et al. 2005).

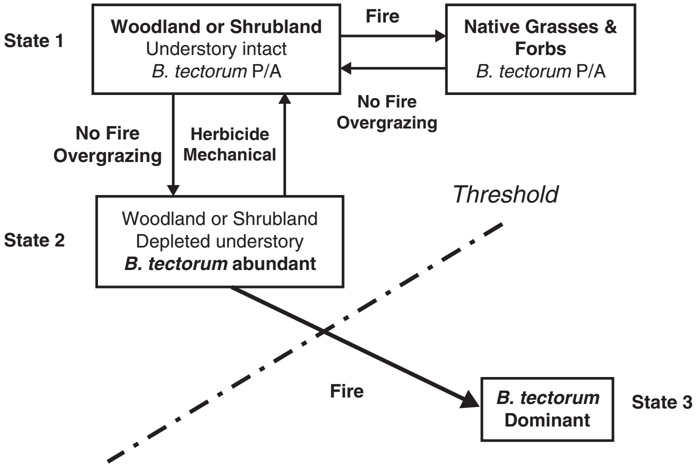

Fig. 7.3 State and transition model for Great Basin sagebrush steppe in western USA. Boxes represent ecosystem states. P/A indicates present or absent but not abundant. Arrows represent processes promoting transitions among states

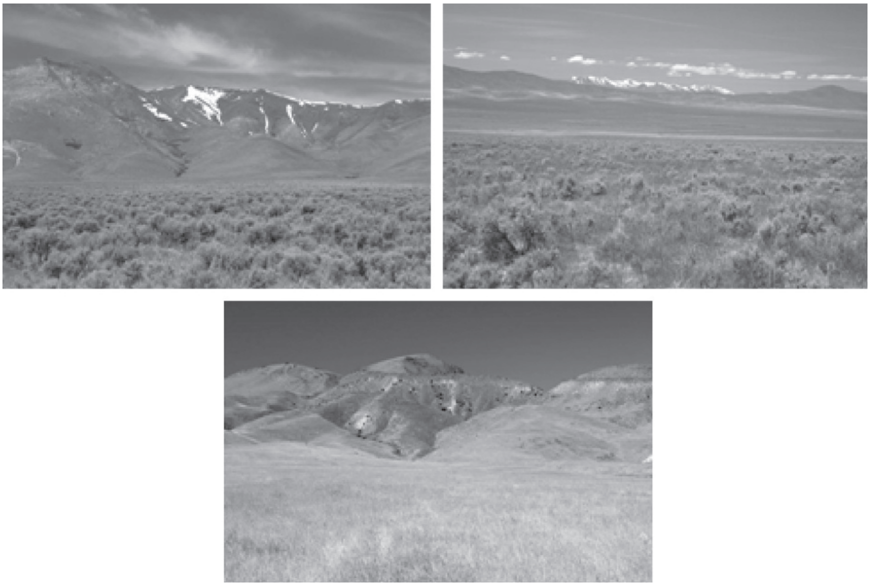

Fig. 7.4 Examples of sites in different stages of *B. tectorum* invasion in Great Basin sagebrush steppe. Left panel = first state where A. *tridentata* shrubland has understory of native perennial grasses and minimal invasion. Center panel = transitional site with A. *tridentata* canopy but understory of *B. tectorum*. Right panel = state where shrubland has been fully converted to B. *tectorum* dominance (photographs: courtesy of Bob Blank, USDA-ARS, Reno Nevada)

Management objectives include increasing native grasses and forbs through competitive release from shrubs and trees, and reducing woody fuel loads to minimize the risk of high-severity fires. Treatments target *A. tridentata* or pinyon and juniper trees and may include prescribed fire or application of the herbicide tebuthiuron (Monson et al. 2004). A separate objective may be to rejuvenate shrub stands characterized by old age individuals and a lack of seedling recruitment. In this case, treatment often involves brush beating or mowing of *A. tridentata* to decrease shrub density, promote shrub seedling recruitment, and increase native herbaceous species through competitive release (Monson et al. 2004). Ideally, areas selected for these treatments have sufficient native perennial herbaceous species that reseeding is not required.

In the next state which we refer to as *transitional*, the community has low resilience and is at risk of crossing a biotic threshold following fire or other disturbance that could result in a new ecological state dominated by *B. tectorum*. Herbaceous perennials may be distributed throughout the community, but are present in relatively low percentages. Native shrubs are a significant part of the community, but *B. tectorum* is present and moderately abundant (Fig. 7.4, center). Management objectives are to decrease woody fuel loads, rejuvenate *A. tridentata* stands, and increase perennial herbaceous species through competitive release. Treatments to decrease woody vegetation typically do not involve fire because of high risk of *B. tectorum* dominance following fire. Instead, hand or mechanical treatments or application of the herbicide tebuthiuron may be used to decrease woody species abundance (Monson et al. 2004). In addition, preemergent herbicides may be used to decrease *B. tectorum* germination (Vallentine et al. 2004). Sites in this state are often revegetated immediately after wildfire because they lack sufficient perennial herbaceous species to provide the resilience for regeneration prior to increases by *B. tectorum*.

In the final state, a biotic threshold has been crossed due to fire and invasion. This new state is dominated by *B. tectorum* (Fig. 7.4, right). An increase in contiguous fine fuels due to *B. tectorum* dominance often results in higher fire frequencies (Whisenant 1990; Link et al. 2006). Low to very low percentages of herbaceous perennial species are present, and fire-intolerant native shrubs, including *Artemisia* species, are largely absent. This is the current condition of thousands of hectares of land in the Great Basin. Control of *B. tectorum* and aggressive revegetation are necessary to restore the native community. Integrated management strategies are being tried in which pretreatments are used to reduce the seedbank of *B. tectorum* followed by revegetation to establish the desired community (Sheley and Krueger-Mangold 2003; Vallentine 2004). Collaborative research and management projects are being implemented across the region to develop solutions for restoring these ecosystems. These are described in the next section.

7.3.6 Ongoing Challenges and Unanswered Questions

Managers face several challenges in defining and implementing the appropriate restoration treatments for sagebrush ecosystems exhibiting *B. tectorum* invasion. The first of these is accurately defining the state of invasion and the potential for recovery following the different types of available treatments. Developing state and transition models that illustrate the vegetation states and successional stages for the various sagebrush ecological types (community types) by the US Natural Resources Conservation Service (NRCS) and Bureau of Land Management (BLM) has helped managers to understand the possible trajectories for these systems. However, the specific conditions (site characteristics and vegetation structure and composition) that result in transitions or threshold crossings following disturbance or management actions have rarely been examined (but see Wright and Chambers 2002; Chambers et al. 2004). Currently, a regional, multiagency project funded by the Joint Fire Sciences Program, "Sage Step," is investigating the thresholds of recovery for sagebrush communities threatened by *B. tectorum* invasion and *Pinus monophylla* (single-needle pinyon pine) and *Juniperus osteosperma* and *J. occidentalis* (Utah and western juniper) encroachment (sagestep.org). The project is examining use of fire and mechanical removal in areas exhibiting *P. monophylla* and *Juniperus* encroachment, and fire, brush mowing, and herbicides in *B. tectorum*-invaded sagebrush sites. Study locations are positioned across the Great Basin to evaluate the generality of these treatments. At each location a gradient of *B. tectorum* invasion or woody plant dominance is being evaluated to determine the thresholds beyond which different treatments are not effective in promoting a resistant understory. Results will help managers select management tools and areas for treatment.

A significant challenge that managers face is the revegetation of sites that are transitional or already dominated by *B. tectorum*. This requires control of *B. tectorum* populations and establishment of native species communities that are resistant to *B. tectorum*. A proven method of controlling *B. tectorum* is the use of herbicides such as glyphosate, which result in high levels of *B. tectorum* mortality when properly applied (Vallentine 2004), but this can be expensive over large areas. Grazing by livestock has been suggested as a means of controlling *B. tectorum* seed production, but field trials show that the annual grass has highly plastic growth and produces seeds even after repeated short clipping (Hempy-Mayer and Pyke 2008). Herbage removal is therefore not effective in eliminating *B. tectorum*, and repeated removal of *B. tectorum* biomass by livestock can harm resident natives. The use of a head smut pathogen (*Ustilago bullata*), which often causes epidemic levels of head smut disease in Intermountain populations of *B. tectorum* is being explored as a biocontrol agent (Meyer et al. 2000, 2005). The pathogen also, however, infects native grasses. Research is underway to determine conditions under which it might be useful as a control agent.

Because invaders are highly responsive to nitrogen, a restoration approach that has been tried in many locations is to utilize methods that decrease N availability. Carbon amendments have been shown to decrease the growth, reproduction, and cover of some invasive species (e.g., Reever-Morgan and Seastedt 1999; Alpert and Maron 2000; Paschke et al. 2000), but they can also affect growth of native species (Monaco et al. 2003; Corbin and D'Antonio 2004). Nonetheless, lower competitive pressure from exotics may compensate for reduced nutrients. Despite some successes, this approach has not been shown to have long-term efficacy for weed control because carbon amendments can be expensive, difficult to use over large areas, and often have only short-term effects (Mazzola et al. 2008).

A recently completed USDA Initiative for Future Agriculture and Food Systems (IFAFS) project on "Integrated Strategies Toward Weed Control on Western Rangelands" evaluated several different approaches for restoring *B. tectorum*-dominated *A. tridentata wyomingensis* communities at eight locations across Oregon, Idaho, Utah, and Nevada (Nowak et al. 2006). The first approach involved identifying native grass and forb species with high probabilities of establishment and strong competitive abilities. Introduced grasses, especially crested wheatgrass (*Agropyron cristatum*, *A. desertorum*, *A. fragile*), are used extensively in the western USA to increase forage production of degraded rangelands and revegetate post-fire landscapes (Lesica and DeLuca 1996; Richards et al. 1998). However, the rapid loss of native sagebrush ecosystems including sagebrush-obligate wildlife species has emphasized the need to focus revegetation efforts on recreating native communities (Wisdom et al. 2005). Also, recent research shows that areas seeded with introduced *Agropyron* species are no more resistant to *B. tectorum* following fire than intact native communities (Chambers et al. 2007). The IFAFS project compared the performance of almost 20 native grass and forb species and two *Agropyron* accessions (Nowak et al. 2006). One accession, *A. desertorum* "CD II," outperformed native grasses in 30% of comparisons, while the other (*A. desertorum* "Valvilov") did not perform better. Several native accessions performed well at multiple locations, are commercially available, and provide viable alternatives to introduced species.

An approach for building community resistance is active seeding of functionally diverse species that will maximize resource uptake by the entire community once established. Revegetation mixtures that include grasses, forbs, and shrubs with varying life forms and rooting depths should facilitate resource extraction through the soil profile, while species with different phenologies should maximize use of available soil resources throughout the growing season. Thus, resistance would be maximized both in the short and longer terms.

The challenges of integrating these approaches are illustrated by the IFAFS project. It evaluated the effectiveness of control of *B. tectorum* using glyphosate, a short-lived herbicide, to reduce population abundance, followed by immobilization of soil nitrogen through sugar (sucrose) applications (Nowak et al. 2006) simultaneous with seeding a diverse mix of native grasses, forbs, and shrubs. Although sucrose addition decreased available N and initial *B. tectorum* biomass and seed production, by the second growing season the effect had disappeared (Mazzola et al. 2008). Sucrose addition also reduced growth of native plants and may have resulted in increased overwinter mortality of seeded natives (Mazzola 2008). Competitive effects of the seeded native species on *B. tectorum* reinvasion were slight although expected to increase as the native species mature. The native species were seed limited relative to *B. tectorum*. Higher seeding rates (600 vs. 150 or 300 plants m^{-2}) resulted in higher establishment of natives as long as *B. tectorum* densities were relatively low (<300–500 plants m^{-2}) (Mazzola 2008).

Recreating sagebrush communities with the functional diversity necessary to support sagebrush-obligate wildlife species and to resist *B. tectorum* invasion is therefore a challenging management goal. Although establishment of several native grass accessions was relatively high in the IFAFS project, establishment of native forbs and shrubs was low and seed availability was limited. For systems dominated by *B. tectorum*, it may be possible first to seed with competitive native grass accessions and then seed with a more diverse species mixture. Although native grasses with broad amplitudes are commercially available, the volume of seed needed to reseed burned areas at a reasonable rate is very high. Native species are generally seed limited relative to *B. tectorum*, and typical seeding rates for native species are probably inadequate (Mazzola 2008). Also seed increase programs and seed zones for native forb and shrub species are just beginning to be developed and for most species, seed supplies are far too limited for large-scale restoration efforts. These limitations are being addressed, in part, by the Great Basin Native Plant Selection and Increase Project, a collaborative effort of the BLM, Great Basin Restoration Initiative US Forest Service, Grasslands, Shrublands and Deserts Project, and other regional agencies and universities (http://www.fs.fed.us/rm/boise/research/shrub/greatbasin.shtml).

The project seeks to increase seed supplies of native plant species, particularly forbs. Its components include plant selection (source-identified seed sources, methods of propagation), seed and seeding technology, and seed production (federal and state nurseries, NRCS, private growers).

Sagebrush ecosystems are changing at a rapid rate. The 2005, 2006, and 2007 fire seasons had among the largest areas burned on record with 437,060, 542,683, and 360,170 ha (1,080,000, 1,341,000, and 890,000 acres), respectively, burned in the state of Nevada alone (http://www.forestry.nv.gov/docs/2007_accomplishment_report.pdf). A high percentage of sagebrush communities that burned will be invaded by or converted to *B. tectorum* (Hemstrom et al. 2002). It has been suggested that managers use a triage process involving “sorting through the sagebrush communities to allocate resources to maximize the number, size, type, and distribution of communities that survive” (Wisdom et al. 2005). The process includes (1) determining which communities are resilient and which are not, i.e., determining their ecological state, (2)developing a systematic process of prioritizing sites, across the entire region, for management activities, (3) utilizing appropriate management techniques to maintain sagebrush communities with a high degree of resilience, and (4) restoring some transitional or converted communities to serve goals of enhancing intact communities that will be resistant to *B. tectorum* invasion.

7.4 Concluding Thoughts

The two examples discussed here provide alternative views on how to promote native species in the face of grass invasions. In both cases the invaders are widespread and persistent, but managers are exploring ways to promote native species by focusing on maintaining or restoring resilience to the native assemblage, or establishing resistant plant assemblages that will reduce the intensity of reinvasion. The Hawaiian example is unusual in that it advocates promotion of a different type of plant community than is known to have existed on the invaded sites – a form of rehabilitation. Such an approach may be the only means for promoting more desirable species in the face of persistent, disturbance-promoting invaders where the new disturbance, fire, is not part of the historical successional framework of the sites and has resulted in an alternative persistent state. In the Great Basin, by contrast, fire was part of the preinvasion (pre-European) disturbance regime and successional framework (Fig. 7.3) and if perennial herbaceous species are still present in the native community, it is often possible to restore resilience and resistance prior to degradation by the invader. In the Great Basin, native grasses and forbs that respond favorably to increased resources following disturbance or management treatments can decrease invasion by *B. tectorum* and, thus, it is also possible to promote resistance. However, in both case studies the supply of native seed for enhancing resistance or resilience (or both) is an important factor limiting management options. The concepts of resistance and resilience are fundamental ecological processes that can assist greatly in "weed control" to help managers work toward goals such as creating "self-sustaining ecosystems able to persist under existing environmental conditions" (SER Primer, http://www.ser.org/). While practicalities may limit implementation of practices based on these concepts, they nonetheless can provide a scientific framework for the development of programs to guide future management efforts.

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