Chapter 13
Invasive Plant Species and Novel Rangeland Systems

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Abstract Rangelands around the world provide economic benefits, and ecological services are critical to the cultural and social fabric of societies. However, the proliferation of invasive non-native plants have altered rangelands and led to numerous economic impacts on livestock production, quality, and health. They have resulted in broad-scale changes in plant and animal communities and alter the abiotic conditions of systems. The most significant of these invasive plants can lead to ecosystem instability, and sometimes irreversible transformational changes. However, in many situations invasive plants provide benefits to the ecosystem. Such changes can result in novel ecosystems where the focus of restoration efforts has shifted from preserving the historic species assemblages to conserving and maintaining a resilient, functional system that provides diverse ecosystem service, while supporting human livelihoods. Thus, the concept of novel ecosystems should consider other tools, such as state-and-transition models and adaptive management, which provide holistic and flexible approaches for controlling invasive plants, favor more desirable plant species, and lead to ecosystem resilience. Explicitly defining reclamation, rehabilitation, and restoration goals is an important consideration regarding novel ecosystems and it allows for better identification of simple, realistic targets and goals. Over the past two decades invasive plant management in rangelands has adopted an ecosystem

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perspective that focuses on identification, management, and monitoring ecological processes that lead to invasion, and to incorporating proactive prevention programs and integrated management strategies that broaden the ecosystem perspective. Such programs often include rehabilitation concepts that increase the success of long-term management, ecosystem function, and greater invasion resistance.

Keywords Ecosystem resilience • Successional management • Novel ecosystems • Management • Rehabilitation • Resilience

13.1 Introduction

The economic impact of invasive plants on livestock production includes interfering with grazing practices, lowering yield and quality of forage, increasing costs of managing and producing livestock, slowing animal weight gain, reducing the quality of meat, milk, wool, and hides, and poisoning livestock (DiTomaso 2000). In rangelands, these noxious invasive plants were estimated to cause $2 billion (USD) in annual losses in the USA (Bovey 1987), which is more than all other pests combined (Quimby et al. 1991).

In the USA, the most prevalent invasive plants have been estimated to occupy between 41 and 51 million hectares of public and private land (Duncan et al. 2004) and they continue to spread at a rate of about 14 % per year (Westbrooks 1998) with no expectation that this rate will decline. Moreover, invasive plants have invaded over half of the non-Federal rangelands and they comprise more than 50 % of the plant cover in 6.6 % of these lands (USDA 2010). In Australia, more than 600 exotic plant species are recorded within rangelands, with 160 of these identified as threats to biodiversity (Grice and Martin 2006; Firn and Buckley 2010), and 20 considered Weeds of National Significance. Therefore, the challenges posed by invasive plants in rangelands are of serious concern and it is expected to increase in the next several decades.

Although it is difficult to assign a monetary value to the adverse consequences of invasive plants, they also adversely impact all categories of ecosystem services that are provided by healthy functional rangelands. Healthy rangelands not only provide economic importance around the world, but also multiple ecosystem services that benefit millions of people in both rural and urban areas. These services include food, fiber, clean water, recreational opportunities, and open space, minerals, religious sites, aesthetics, and natural medicines (Havstad et al. 2007; Rudzitis 1999). Furthermore, rangelands provide important nontraditional ecological services, including biodiversity, wildlife habitat, and carbon sequestration (Havstad et al. 2007).

Many invasive plant-infested areas have experienced drastic changes in vegetative structure and function, including plant community composition and forage quantity and quality. Plant invasion can reduce biological diversity, threaten rare and endangered species, reduce wildlife habitat and forage, alter fire frequency, increase erosion, and deplete soil moisture and nutrient levels (DiTomaso 2000). In various quantitative assessments, invasive plants have been estimated to decrease range productivity by 23–75 % (Eviner et al. 2010), native plant diversity by 44 %, and abundance of animal species by 18 % (Vilà et al. 2010). In addition, they have increased fire frequency and intensity and the amount of area burned, as well as the prevalence of other invasive species (Mack et al. 2000). For example, the dominance of *Bromus tectorum* dramatically shortened the fire frequency intervals throughout the Great Basin region of the USA, leading to near elimination of much of the native shrub vegetation (Whisenant 1990). Even in tropical areas of Hawaii the invasion of non-native warm- and cool-season grasses has provided an abundance of fine fuels, which have increased fire frequencies (D'Antonio and Vitousek 1992). This has subsequently led to dominance by more fire-tolerant non-native species (Fig. 13.1).

The management of invasive plants on rangelands can be more complicated and difficult than weed control in agricultural systems. While control often refers to population reduction of a target weed or invasive plant species, the term management is more inclusive and encompasses control efforts within the crop or rangeland ecosystem. In agricultural areas, for example, the goal of weed management is to eliminate all vegetation to enhance the yield of the desired crop. In contrast, on rangeland systems the goal of a management program is to preserve or enhance all desired species, yet remove one or a few undesirable species. In addition, unlike agricultural

Fig. 13.1 *Bromus tectorum* (downy brome or cheatgrass) infested area within the western United States. The invasive European grass has converted millions of hectare from sagebrush steppe to annual grasslands

production that occurs on private lands, approximately 50 % of rangelands in the 14 western US states are in public ownership that are managed by several federal, state, or local government agencies (Havstad et al. 2007). However, both public and privately owned rangelands confront similar challenges regarding invasive plant management. In many other areas of the world, particularly in less-developed countries, the motivation to manage invasive plants is more a function of human subsistence and survival, than it is about increased profitability or a return to a traditional historic community (Hobbs et al. 2009). In these situations, it would be expected that invasive plant management would be seldom attempted. These considerations can impact decisions or approaches to managing invasive plants on rangelands.

The overarching goal of this chapter is to summarize proactive strategies, and their corresponding conceptual frameworks that offer the greatest success in achieving desired outcomes in invasive plant management programs on rangelands. Our specific objectives are threefold. First, our goal is to clearly define relationships between invasive plants and ecosystem services, and to identify management systems that yield the greatest overall return of ecosystem services. To achieve this goal requires greater emphasis on important ecosystem services that can be realized through a more integrated management program, as well as to recognize under what conditions invasive species removal is possible. The concept of novel ecosystems may be more realistic for severely invaded and modified ecosystems. Second, our objective is to discuss the societal implications of novel ecosystems, the services they provide, and the consequences of short-, medium-, and long-term management activities. Integrated within a management program is the need for prevention strategies, including predictive models for assessing invasion risk and understanding the biological causes of succession and invasion, and flexible restoration strategies to maximize the success of converting degraded communities into functional systems. Our final objective is to provide a theoretical framework for recovery of degraded communities that contrasts reclamation, restoration, and rehabilitation and the expected outcomes and costs for each approach.

13.2 Scope of the Invasive Plant Problem

The widespread invasion and undesirable impacts of rangeland invasive plants have been recognized for well over 100 years. Much of the private rangeland in the western USA is now occupied by a variety of invasive plant species (Fig. 13.2). In the USA alone, it is estimated that there are over 3000 non-native plant species that have become naturalized and are able to maintain self-sustaining populations within rangelands (Kartesz 2010). However, only 37–60 non-native species are considered of major economic and ecologic importance (DiTomaso 2000). Many of these invasive plants were introduced with the genuine intention to improve ecosystems for a specific land use objective. Some of these introductions have proven successful, but many have not (Cook and Dias 2006).

In the western USA, several annual grasses (*Bromus hordeaceus* (soft brome), *Avena barbata* (slender oat), and *Lolium perenne* ssp. *multiflorum* (Italian ryegrass))

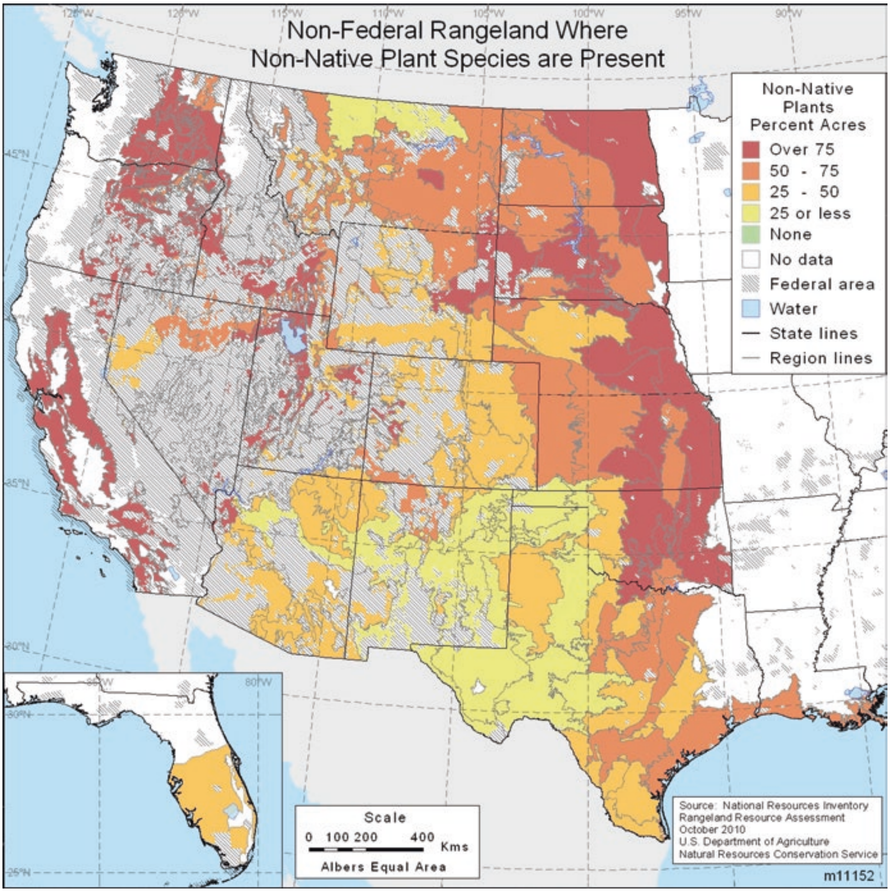

Fig. 13.2 Non-federal rangeland were invasive plants are present (USDA 2010)

and perennial grasses (*Phalaris aquatica* (hardinggrass), *Pennisetum ciliare* (buffelgrass), and *Eragrostis lehmanniana* (Lehmann lovegrass)) were intentionally introduced as forage species or as potential crops (e.g., *Isatis tinctoria* (dyer's woad)). However, the majority of invasive species, particularly thistles, were accidentally introduced as contaminants in seed, transported on equipment and vehicles, or in fur and clothing (DiTomaso 2000). By comparison, the Australian Commonwealth Plant Introduction Scheme was initiated in 1929, and over-time introduced more than 5000 species of grasses, legumes, and other forage and browse plants, including woody species (Cook and Dias 2006). In tropical Australia, 13 % of introductions have become a problem, with only 5 % being considered useful for agriculture (Lonsdale 1994). Low (1997) suggested that 5 out of 18 of Australia's worst tropical environmental invasive plants were intentionally introduced as pasture grasses. The degradation of these natural ecosystems, including rangelands, has generally occurred despite the best intentions of improving an ecosystem to provide ecosystem services people desire (Fig. 13.3).

Fig. 13.3 Mitchell grassland normally dominated by native perennial grass (*Astrebla* spp.) invaded by the African shrub *Acacia nilotica* (prickly acacia). Unlike the western United States, this invasive shrub has converted large expanses of the Mitchell grasslands to scrubland

Among the invasive rangeland plants in the USA, annual bromes (*Bromus* spp.) are the most pervasive and problematic. Of the annual bromes species, *Bromus tectorum* (cheatgrass, downy brome) is by far the most problematic and now infests 23 million hectares (Duncan et al. 2004) and 28 % of all non-Federal rangelands (USDA 2010). Downy brome was first introduced to the western USA in 1861, and by the early 1900s was widely distributed in many rangelands, particularly *Artemisia* spp.-dominated (sagebrush) ecosystems that were overgrazed (Billings 1994). In these areas, it altered the natural fire regime to replace native, perennial species as the dominant vegetation (Whisenant 1990). Similarly, *Centaurea solstitialis* (yellow starthistle) was introduced from Chile to California around 1850 and to other South American countries even earlier (Gerlach 1997). By the early 1900s it was a common invasive plant of rangelands, roadsides, grain fields, and alfalfa fields in northern California and Argentina, where it has outcompeted most native annual species. Today it is estimated to infest nearly six million hectares of rangeland in the USA (Duncan et al. 2004). Some of the other widely distributed and problematic rangeland invasive species in the western USA include medusahead (*Taeniatherum caput-medusae*), other *Centaurea* species, especially diffuse knapweed (*C. diffusa*) and spotted knapweed (*C. stoebe*), musk thistle (*Carduus nutans*), Canada thistle (*Cirsium arvense*), and leafy spurge (*Euphorbia esula*). Combined they are estimated to infest about 16 million hectares in the USA (Duncan et al. 2004). One of the most important rangeland invasive species in Australia, *Echium plantagineum* (Paterson's curse, salvation Jane) was introduced from the Mediterranean region in 1843, and by 1900 it was well established in rangelands of southeast Australia (Parsons and Cuthbertson 2001). Like *Centaurea solstitialis*, it has impacted native plant diversity, as well as pasture legumes, in Australian grazing lands (Cullen and Delfosse 1985). While invasive plants can cause broad-scale changes in plant communities, historical cultural practices, particularly overgrazing, can increase invasive plant establishment and proliferation. For example, by 1895 overgrazing of rangelands in several Canadian provinces and 16 western states of the United States (US) led to dense infestations of *Salsola tragus* (Russian-thistle) (Young and Evans 1979).

13.2.1 Ecosystem Services

While the detrimental effects of invasive plants in rangelands and other plant communities are well documented, there are many instances where they provide benefits to the ecosystem (Eviner et al. 2012). Typically, however, there are trade-offs between positive and negative impacts of invasive plants. This is most apparent in highly altered and degraded landscapes where abiotic conditions are so degraded that native species are unable to naturally recover and recovery may not be possible even when mediated by restoration efforts. In these systems, invasive plants may provide a number of beneficial services, including reduced soil erosion, regulation of pests and disturbance regimes, purification of air and water, increasing habitat for pollinators and other species, providing nurse sites for native plant establishment, and facilitating phytoremediation (Diaz et al. 2007; Richardson and Gaertner 2013).

Invasive plant species were even intentionally introduced to restore key ecosystem services in some degraded systems (Eviner et al. 2012). These services included livestock or wildlife forage, wildlife habitat, erosion control, honey source plants, and medicinal or ornamental value (Duncan et al. 2004). For example, in California the largely unintentional introduction of non-native European winter annual grasses, such as *Bromus hordeaceus* (soft brome), *Avena barbata* (slender oat), and *Lolium perenne* ssp. *multiflorum* (Italian ryegrass) have greatly altered the survival of grazing-intolerant native perennial grassland communities to only a fraction of their original composition (Murphy and Ehrlich 1989). Today, however, these annual grasses are considered desirable and productive forage species, particularly in the Central Valley and foothill grasslands of California. In other parts of the world, including the USA and Australia, non-native perennial grasses were also intentionally introduced for increased forage production, drought tolerance, and soil stabilization (D'Antonio and Vitousek 1992; Cook and Dias 2006; Lonsdale 1994). Among the more widely planted perennial grasses include, *Agropyron cristatum* (crested wheatgrass), *Pennisetum ciliare* (buffelgrass), *Eragrostis lehmanniana* (Lehmann lovegrass), and *Eragrostis curvula* (African lovegrass; Australia). All these species present significant trade-offs, depending on the region in which they had been introduced.

Despite the use of some invasive plants to provide ecosystem services, there is a general lack of understanding of how to predict and manage, or even measure, the effects of invasive species on ecosystem services (Eviner et al. 2012; Jeschke et al. 2014). This can limit the decision-making ability of land managers, yet ecosystem services are increasingly being used as criteria for prioritizing efforts to remove or manage invasive plants. In many situations, the focus has shifted from preserving the historic species assemblages within a particular site, to conserving the functionality and services provided by the existing plant community (Hobbs et al. 2011).

13.2.2 Novel Ecosystems and Restoration

While there are a number of factors that contribute to the severity of impacts of non-native plants on ecosystems, ecologists have recently recognized that these impacts may be a symptom of shifting environmental conditions that will no longer support the native community (Eviner et al. 2012; Hobbs et al. 2009). These "novel ecosystems" occur because the species composition and function of greatly altered ecosystems have been completely transformed from the historic system (Hobbs et al. 2009) (Fig. 13.4). In these situations, invasive species may not be dramatically disrupting ecological processes, but rather, they may be sustaining or restoring important ecosystem services under a different set of environment conditions.

Novel ecosystems can be the consequence of abiotic changes brought about through impacts of climate, land use, pollution, CO_2 and atmospheric nitrogen

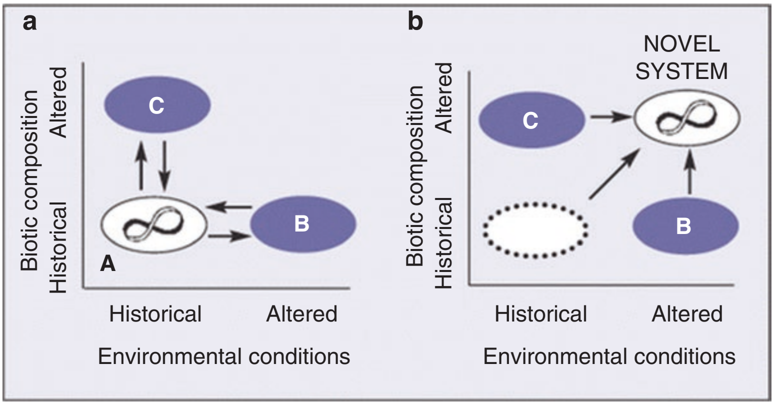

Fig. 13.4 Creation of novel ecosystems via biotic or abiotic change (modified from Suding et al. 2004). The "range of variability" and the adaptive four-phase cycle of a natural ecosystem are collapsed into the range of values found in zone A. (**a**) An ecosystem is altered by directional environmental drivers (A → B) or the addition or loss of an important species (A → C). (**b**) Once in the new state (either B or C), internal restructuring due to new biotic and abiotic interactions further alters community composition through changes in abundances or species losses, and through changes in biogeochemical interactions (from Seastedt et al. 2008)

enrichment, altered disturbance regimes, and urbanization (Steffen et al. 2004). These abiotic changes will almost certainly lead to subsequent changes in species composition and biogeochemical cycling that can irreversibly alter the system from its historic condition (Seastedt et al. 2008). In addition to abiotic changes, biotic modifications may also contribute to the development of novel ecosystems. These biotic changes may include new species invasions (plants or animals, including livestock), local extinction of keystone species or ecosystem engineers.

While some ecosystems remain intact, many, if not most, are novel and have an altered structure and function with unprecedented combinations of species under new abiotic conditions, compared to historic systems. This is an important starting point for the development and prioritization of invasive species management programs (Seastedt et al. 2008; Perring et al. 2013). However, it is likely that these novel systems are capable of further transformation and may not necessarily represent a resilient community. This is an important consideration in identifying desired outcomes and long-term management strategies that are designed to maximize ecosystem function and services, yet build and maintain ecological resilience (Eviner et al. 2012). Innovation, adaptation, and social flexibility will be required to attain these goals (Seastedt et al. 2008).

Many restoration efforts have the objective of recreating the historic landscape, despite the uncertainty associated with what is actually the "natural" ecosystem (Hobbs et al. 2009; Jackson and Hobbs 2009). In most cases outside of Europe and Asia, for example, the historic or "natural" ecosystem is defined as what was present before Europeans exerted their widespread influence on landscapes (Black et al. 2006; Bowman 1998). The goal of restoring ecosystems back to their "natural" state can only be effective when the historic range of variability in abiotic and biotic feedback mechanisms are still present (Seastedt et al. 2008). This may be feasible in some systems, for example, high elevation meadows, forests and riparian areas or remote and isolated wildlands, where invasive species have had only a limited impact. Alternatively, if the definition of a historic system is broadened to include a certain amount of modification and addition of new species, it may not be possible to conserve or restore the site to near-historic conditions, yet it may represent a functionally similar system (Hobbs et al. 2009). With an even broader definition of a historic system, it is possible to restore the key features and functions of the ecosystem, without the constraint of eliminating all nonindigenous species. Land managers need to take these considerations into account when assessing the feasibility of success, economic realities, and even intrinsic cultural values.

While restoration to a historic or near-historic ecosystem is possible in some cases, the abiotic and biotic feedbacks may have been so dramatically altered for many rangelands that they now represent novel ecosystems with unique assemblages of species and functions that have no analog to historic systems (Hobbs et al. 2009). Novel ecosystem recovery to conditions resembling historic conditions through restoration is considered very unlikely or impossible (Jackson and Hobbs 2009; Seastedt et al. 2008). Consequently, restoration programs should focus on managing for future change that emphasizes ecosystem function, goods, and services (Hobbs et al. 2011), maintaining genetic and species diversity, and encouraging biogeochemical processes that favor desirable species (Seastedt et al. 2008). The challenge for land managers in the future will be to determine what extent or type of changes are considered beneficial, while avoiding actions that create further ecosystem degradation (Jackson and Hobbs 2009). Restoration options that remove the requirement of attaining a historic ecosystem may reduce both effort and costs, yet may still achieve a desired outcome (Hobbs et al. 2009).

13.3 Major Conceptual Advances

In the current era of anthropogenic change, native species losses have been exceeded by gains in exotic species (Ellis et al. 2012). It is now clear that few native plant communities remain undisturbed and that most are accompanied by exotic species (Ellis and Ramankutty 2008). While novel ecosystems are not ubiquitous (Murcia et al. 2014), the global pattern of species reshuffling due to human disturbances creates the need to improve our understanding of proactive management possibilities. Proactive approaches are warranted because by definition, novel ecosystems have experienced simultaneous biotic and abiotic changes with no historical analog system or clear understanding of how to restore them (Hobbs et al. 2006; Williams and Jackson 2007). Given this high level of uncertainty, we sought to identify some of the major conceptual advances in invasive species management that have occurred during the past 25 years and explore ways to proactively apply them to rangelands that meet the criteria of novel ecosystems.

13.3.1 Integrated Invasive Plant Management

Although more prevalent in past years, invasive plant management has often focused on the control of a single species without regard to the unintended consequences of the control method. This approach typically relied on a single control technology, such as grazing, herbicides, or prescribed burning, and it has generally proven inadequate to keep pace with ecological threats emerging worldwide (Hobbs and Humphries 1995). This strategy has proven unsuccessful in the long-term (Masters and Sheley 2001), but frequently the resident native species or even desirable non-native species do not benefit from the management strategy and can actually deteriorate further (Seastedt et al. 2008). This occurs because removal of the invasive species does not necessarily restore the ecosystem to a functional system, but may lower the abundance of important desirable plants, and cause further losses in ecological functioning (Pokorny et al. 2005). Instead of recovering rangeland function, this control method may degrade the abiotic environment or open niches for reinvasion or invasion by other undesirable species (Masters and Sheley 2001). For example, the use of the herbicide aminopyralid to control *Centaurea solstitialis* in California can lead to the subsequent invasion and proliferation of even less palatable and equally noxious long-awned annual grasses, such as *Aegilops triuncialis* (barb goatgrass) or *Taeniatherum caput-medusae* (medusahead), that are not susceptible to the herbicide (DiTomaso et al. 2006).

13.3.1.1 Identifying Causes of Invasion

Greater emphasis needs to be focused on the management of invaded systems and identification of the underlying causal factors responsible for the invasion (Hobbs and Humphries 1995). This would contribute to a more appropriate approach to invasive plant management and to development of a broader ecological understanding of the mechanisms and processes that contribute to invasive plant success and develop management strategies that promote functional systems, provision of ecosystem services, and resilience to reinvasion (Hulme 2006; Seastedt et al. 2008; Sheley and Krueger-Mangold 2003). In some cases, this may require a compromise that is logistically practical and cost effective. Consequently, a broader view of invasive plant management emerged as integrated invasive plant management was combined with other aspects of ecosystem function.

13.3.1.2 Applying Multiple Control Tactics

Integrated pest management (IPM) of invasive plants stemmed from the realization that the dominance and spread of invasive plants indicates an underlying management problem that should be addressed before control can be successful. The basic elements of invasive plant management include the use of multiple control tactics and the careful integration of knowledge regarding the invasive species into the management effort (Buhler et al. 2000). For example, invasive plant management emphasized that recovery of degraded rangelands require more than control of the invasive plant. It is founded on a systematic, sequential application of multiple, combined tactics such as chemical, biological, cultural, and mechanical control measures to remediate ecosystem functions and reduce the negative impacts of invasive species below an economic threshold (Masters and Sheley 2001). This new management approach signified a change in inquiry from “what is that invasive plant and how do I remove it?” to “why is that invasive plant present and how can I manage the system to suppress it, prevent its spread, and remediate its impacts?” The adoption of invasive plant management was also spurred by the need to broaden typical control efforts that relied too heavily on herbicides and tillage (Holt 1994) and lessen the occurrence of herbicide-resistant weeds due to repetitive use of herbicides (Beckie and Reboud 2009). It is also important to recognize that an invasive plant management program is very often closely tied to restoration, mitigation, and rehabilitation efforts. As will be discussed in more detail, the goal of rehabilitation emphasizes both the short- and long-term effects on biodiversity and socioeconomic values.

Recognizing some of the key challenges associated with IPM can enhance the application of invasive plant management on rangelands. For example, biological, implementation, and research challenges must be addressed when developing invasive plant management systems (Buhler et al. 2000). Some of these challenges include failure to account for fecundity and survival of invasive species and excessive emphasis on individual populations in a single year as opposed to adopting a holistic approach based on analysis, theory, and implementation within an ecosystem. Research needs to develop practices capable of directly impacting propagule production, plant survival, and the transition from propagules to seedlings (Buhler et al. 2000). Another challenge of invasive plant management may be the most obvious; that is, prioritizing one control practice at the expense of an overall invasive plant management strategy that is environmentally and economically viable (Buhler 2002).

13.3.1.3 Successional and Process-Based Management

Integrated plant management has also benefited from the adoption of successional theory to understand the causes of succession, and adapting this theory to manage rangeland invasive plants. For example, Sheley et al. (1996) suggested that rangeland invasive plant managers need principles and concepts to guide their decisions as opposed to prescriptions for invasive plant control. They outlined a theoretical framework based on a successional model that emphasized influencing the primary causes of succession (i.e., disturbance, colonization, and the performance of species) to alter the plant community from an undesired state to a desired state (e.g., Pickett et al. 1987). This model was also closely aligned with specific ecological processes that should be influenced in order to affect underlying causes of succession, such as modifying *disturbance* to address site availability, *propagule dispersal* and *reproduction* to influence species availability, and altering *resource availability* or applying *stress* to impact performance of both invasive and desirable species (Sheley and Krueger-Mangold 2003; James et al. 2010). For example, the coordinated control of two invasive species—*Centaurea stoebe* (spotted knapweed or formerly *Centaurea maculosa*) and *Potentilla recta* (sulphur cinquefoil)—and perennial grass habitat restoration was accomplished by successively modifying invasive plant performance with herbicides, disturbance with variable seeding techniques, colonization with different seeding rates, and soil resource availability with cover crops (Sheley et al. 2006). The link between invasive plant management and successional theory has also been augmented by the realization that plant communities exist in alternative ecological states that may shift in nonlinear ways in response to disturbance (Westoby et al. 1989).

Process-based management was not only a core aspect of successional invasive plant management, but it also became a central theme of the emerging field of applied ecological restoration. Akin to invasive plant management, ecological restoration emphasized the importance of developing methodologies for landscape application while recognizing the need to target the specific processes responsible for degradation and recovery (Hobbs and Norton 1996). This process-oriented approach, based on the goals of renewing and maintaining ecosystem health, became an early working definition for the Society for Ecological Restoration (Higgs 1997). As the framework developed, further emphasis was placed on process-oriented restoration principles and practices to repair damaged landscapes (Whisenant 1999). This concept of managing processes at large scales expanded in the 1990s with the emergence of ecosystem-based management in federal US agencies (Koontz and Bodine 2008). Although the four largest land-management agencies in the USA (i.e., Forest Service, Fish and Wildlife Service, National Park Service, and the Bureau of Land Management) had formally adopted ecosystem management by 1994, implementing the preservation of ecological processes was identified as a primary challenge (Koontz and Bodine 2008). In recent years, process-based management has also become a central component of resilience-based management, which continues to explore how the role of ecological variables and processes influence rangeland dynamics at various temporal and spatial scales (Briske et al. 2008a; Bestelmeyer and Briske 2012).

13.3.1.4 Ecosystem Resilience

Resilience-based management should provide physical and ecological conditions that allow the system to be self-sustainable and return to pre-disturbance conditions, or reasonably close, within a fairly short time frame following removal, stress, or disturbance (Walker et al. 2002). In addition, a resilient system should be able to resist successful establishment, spread, and ecosystem change from invasive plants following the introduction of propagules (D'Antonio and Chambers 2006). The "whole-ecosystem approach" is now considered an essential aspect of managing invasive species. This is important because the secondary effects of invasive species removal can result in unexpected changes in other ecosystem components, such as (1) trophic cascades on food-web interactions among producers, consumers, and predators, (2) plant-herbivore interactions, and (3) native species reliance on exotic-species habitats (Zavaleta et al. 2001). An ecosystem perspective also provides linkages between the four common management responses of prevention, rapid response and eradication, control/containment, and restoration/mitigation to mirror the invasion processes of introduction, establishment, spread and impact, respectively (Table 13.1). Linking stages of invasion to specific management actions has since

Table 13.1 Relationships among stages of invasion, management strategy, management efficiency, and management costs (from Hulme (2006) and Simberloff et al. (2013))

Invasion stage	Management strategy	Management efficiency	Management cost
Introduction	Prevention	High	Low
Establishment	Rapid response and prevention	Moderate	Moderate
Spread	Control	Low	High
Impact	Restoration/mitigation	Very low	Very high

Invasion stage refers to sequential degradation of rangeland ecosystems over time from introduction of the invasive species to when its presence impacts ecological processes been proposed as a unified framework for biological invasions, wherein barriers to individual plants, populations, key processes, or entire species must be overcome in order for invasive species to pass to the next stage (Blackburn et al. 2011). Another theoretical framework similarly built upon invasion stages or processes reduced the redundancy among 29 leading invasion hypotheses. This was accomplished by documenting how propagule pressure, abiotic site characteristics, and biotic characteristics of the invasive species can be utilized to narrow the number of mechanisms and processes involved in invasion, as well as identify sequential steps needed to improve invasive plant management (Catford et al. 2009). One such framework for predicting the suite of traits that confer invasion success identified three primary factors, namely, prevailing environmental conditions, traits of the resident species, and traits of the invading species (Moles et al. 2008).

13.3.2 Managing for Ecosystem Function, Functional Species Groups, and Functional Species Traits

Because novel ecosystems have experienced extreme species reshuffling (e.g., Ellis et al. 2012), irreversible restoration thresholds become a defining characteristic due to species extinctions, invasion by exotic species, and highly modified ecological composition and structure. When abiotic and biotic characteristics are severely modified such that irreversible restoration thresholds are recognized, managing the novel ecosystem pursuant of desired functioning and ecological services may take precedence over futile endeavors to reconstruct historical biotic and abiotic composition and functioning (Hallett et al. 2013). This paradigm shifts attention away from managing for species composition and toward identification of key obstacles to maximizing ecosystem functioning and provisioning of ecosystem services.

Ecosystem function has been studied at multiple levels, including plant communities, species functional groups, species, and species traits. For example, as the importance of biodiversity gained prominence in the early 1990s (e.g., Wilson 1992), its functional role within plant communities became a theoretical arena to explore alternative hypotheses regarding its importance (Johnson et al. 1996). One such hypothesis—the redundancy hypothesis—was introduced by Walker (1992), who proposed that when several species regulate ecosystem processes in similar ways they can be considered a functional group and redundancy among species performing similar function enables the ecosystem to compensate for the loss of one or more species. This new interpretation drew attention away from individual species and emphasized identification of functional groups and their role in sustaining ecosystem processes and functions, including invasion resistance. Accordingly, it is now recognized that species and groups of species can have strong effects on their environment and on specific ecosystem functions. The presence of specific functional groups may be more important than species richness or a specific species, as was shown for novel forests that maintained basic ecosystem processes after widespread loss of native species and replacement by introduced species (Mascaro et al. 2012).

The research topic of functional species traits was suggested as a means to make comparisons across regions and scales and allow researchers to assess relationships between traits and ecological processes (Craine et al. 2002). The importance of functional species traits is also based on strong evidence that key ecosystem processes can be predicted by structural or functional traits (Diaz et al. 2007). Species functional traits have also been used to match traits of invasive species to those of native species in an effort to assemble plant communities with greater invasion resistance (Funk et al. 2008; Drenovsky et al. 2012). When plant communities are composed of, or are created using native species that have functional traits similar to invaders, greater invasion resistance is theoretically possible (Funk et al. 2008). This is particularly true when native and invasive species are functionally similar in phenology, which has been shown to strengthen invasion filters (Cleland et al. 2013). A flipside of this theory is that functionally dissimilar invasive species may be more likely to invade and become abundant due to limited competitive exclusion provided by the resident community (Strayer et al. 2006). Given that invasion resistance has been linked to functional species traits, using these traits as a restoration tool is a promising research field with far-reaching management potential. In particular, as the merits of functional traits and species selection for restoration are pursued in the future, it will be important to improve our understanding of which traits are needed to overcome abiotic and biotic thresholds and promote restoration (Jones et al. 2010).

13.3.3 Rationale for Preventive Measures

Accumulation of non-native species in many regions of the global is accelerating in response to human activities, necessitating the need for predictive models to assess invasion risk (Lockwood et al. 2005). Because novel systems are often infested with exotic species, some of which may be highly invasive, preventative measures to assess risk is needed to monitor the status of invasive species. Although, exotic species arrival to a new region is not necessarily a basis for invasive species status (Valéry et al. 2008), preventative measures must still be pursued because multiple invasive species concurrently exist within novel ecosystems. When dealing with multiple invasive species, which vary in their potential to invade, it becomes important to devise strategies to screen species (Pyšek et al. 2004) and set priorities for control efforts (Hobbs and Humphries 1995).

It is widely established that invasive species prevention is far more cost effective than allocating limited resources to control efforts (Finnoff et al. 2007; Panetta 2009) (Table 13.1). Two potential prevention approaches with very different consequences have been identified: (1) prevention of invasion in the present with low target invasive plant specificity and no damage costs in the future and (2) no costs for prevention in the present but possibly high costs for specific target invasive plant control in the future (Naylor 2000). Although both options come with risk and trade-offs between expected benefits and cumulative damage, early detection via proactive assessment of invasive plant flora within novel ecosystems makes good sense to detect problematic invasive plants that may exist in low densities before they proliferate and require large investments to control them (Hobbs and Humphries 1995; Simberloff et al. 2013). This is particularly true for invasive plants that possess a lag phase following initial invasion, but are predicted to rapidly spread and cause significant damage when barriers to invasion are removed (Cunningham et al. 2004). Preventative measures and control priority should also be given to invasive species that dramatically alter the ecosystem—often referred to as engineer and transformer species—and that are known to impact ecosystem processes (Hastings et al. 2007).

13.3.4 Negative Impacts of Invasive Species Management

Control of invasive species is often followed by increases in native species abundance (e.g., Flory and Clay 2009). However, this is most often observed when invasive species are recognized as ecosystem "drivers," and their removal leads to ecosystem recovery (Bauer 2012). In contrast, when an invasive species is considered an ecosystem "passenger" or "back-seat driver," models suggest the invasive species removal will not promote recovery of the native plant community or will require both removal and ecosystem restoration to promote recovery (Bauer 2012). This establishes that careful consideration should be given to how control measures impact entire ecosystems.

In some cases invasive plant management can have negative impacts on ecosystem properties, such that control efforts exacerbate invasion or damage desirable species (Kettenring and Adams 2011). For example, when control practices disturb soils or release resources, invasive species can often gain a competitive advantage over native species (Davis et al. 2000; Cleland et al. 2013). In some cases, invasive plant treatment can be worse than the cure, as is the case when nontarget species are injured following herbicide treatment (Rinella et al. 2009) or when eradication of target species contributes to invasion by other invasive species (Courchamp et al. 2011). According to this scenario, disturbance created by invasive plant management may cause an open site where the target invasive plant or another undesired species from the community readily reinvades (Buckley et al. 2007). Invasive species control can also exacerbate problems if negative abiotic effects persist after their removal and additional restoration steps are not taken to remediate these effects (Corbin and D'Antonio 2012). Lastly, while invasive species are primarily attributed to negative impacts on ecosystem processes and services, in some cases they may enhance rangeland functioning (Eviner et al. 2012). Consequently, the decision to control an invasive plant or restore a novel ecosystem to a natural system, with corresponding ecological services, is not always clear. Only in certain cases is this decision straightforward and certain, for instance, when the goods and services provided by attaining the natural system outweigh the costs of control and when the value of the restored systems is low, but restoration is inexpensive and easy (Belnap et al. 2012). Policy-makers and managers should rank invasive species and whether to pursue eradication or prevention measures based on two criteria: (1) species impact and (2) feasibility of removal or restoration (Parker et al. 1999). High emphasis should be given to eradication when both impact and feasibility of removal is high. In contrast, high priority should be given to research efforts preceding the prevention of species when their potential future impact is high and feasibility of removal is limited by lack of clear management strategies.

13.4 Implications of Conceptual Advances

Over the last two decades as invasive plant management has shifted away from a focus on tools and technology for short-term invasive plant control and toward an emphasis on identifying, managing, and monitoring ecological processes that drive invasion, an array of conceptual advances have developed. These advances are detailed in the previous section and include a refined understanding of invasive plant impact on ecosystem services as well as insight on potential negative impacts of invasive plant control efforts on ecosystems. These advances also included development of invasive plant management tools and strategies, prevention programs, and a better understanding of plant functional trait attributes and how they can be used effectively to design site-specific invasive plant control and desired plant restoration programs. These conceptual advances have major implications for understanding constraints and opportunities for rangeland invasive plant management now and in the future. Broadly, implications of these key conceptual advances can be organized under four themes, including (1) supporting incentives for ecosystem services, (2) deploying long-term invasive plant management programs, (3) addressing socio-economic dynamics of invasive plant management, and (4) refining how management strategies and goals are identified and developed. Each of these implications is discussed below.

13.4.1 Incentives for Ecosystem Services

One of the most salient implications of recent conceptual advances is centered on the links between rangelands, invasive species, and ecosystem services. While on an area basis rangelands represent one of the largest land cover type in the world, marginal rates of return range from low to subsistence level (Tanaka et al. 2011). However, these systems provide a large array of nonmarket ecosystem services to society across the globe including carbon sequestration, biodiversity conservation and water capture and storage and invasive plants, through various mechanisms, can seriously impact these critical services (Eviner et al. 2012; Plieninger et al. 2012). Broad recognition of ecosystem services has driven development of markets and incentives to maintain or enhance rangeland ecosystem services (Chap. 14, this volume). Development of markets and incentives vary substantially globally, and can include payments for services (e.g., carbon sequestration, targeted grazing for invasive species, or fire protection), cost-share programs, technical assistance, tax incentives, and conservation easements (Lubell et al. 2013). In the majority of situations, costs for invasive plant control on rangeland greatly exceed market benefits and incentive programs have traditionally been central to maintaining invasive plant management programs on private rangeland. A recent assessment initiated by the USDA Natural Resources Conservation Services (NRCS) (Briske 2011) recognized that the conservation benefits of invasive plant control are poorly described (Sheley et al. 2011a), making it difficult to identify the return on taxpayer support for these management practices . This assessment also found that invasive plant control can have pronounced negative effects on ecosystem services (Sheley et al. 2011a). On private rangelands, there is generally less support available from incentive programs than there is demand for incentive support (Aslan et al. 2009). The minimal efficacy of incentive programs question the long-term support that could be allocated to these programs. A similar situation exists on public lands where current management resources for invasive species only cover a fraction of the potential area where management is needed (US Bureau of Land Management 1999). A critical emerging opportunity is to more strongly define the relationship between invasive rangeland plants and ecosystem services (e.g., Vilà et al. 2010), and identify species and management scenarios where management inputs yield the greatest aggregate return on ecosystem services.

13.4.2 Deploying Long-Term Invasive Plant Management Programs

Developing integrated invasive plant management principles and decision tools that provide the foundation for long-term and sustainable invasive plant management programs is essential as society begins to closely examine costs and benefits of invasive plant management on private and public rangeland.

As highlighted in the NRCS assessment (Sheley et al. 2011a), decision support systems are a central component of invasive species management because costs of control are high, risk of practice failure and nontarget effects are substantial, and the response of ecosystems to control efforts are often difficult to predict (Epanchin-Niell and Hastings 2010; Januchowski-Hartley et al. 2011). Stakeholders largely agree that invasive plant management is a complex, iterative, and a long-term process (Aslan et al. 2009; Brunson and Tanaka 2011) and there are a number of examples describing how general principles and decision tools can be applied to guide long-term invasive plant management efforts (e.g., James et al. 2010; Sheley et al. 2011b). Despite these conceptual advances there has been little evaluation of the adoption or impact of this information in management decisions and almost no data available on the long-term efficacy of these alternative strategies (Sheley et al. 2011a). Variables associated with individual rangeland enterprises often constrain deployment of invasive plant management tools and strategies below the optimum scenario. How these constraints influence the efficacy of long-term invasive plant management efforts and the conservation or enhancement of ecosystem services is generally not known. The evaluation of adoption barriers and assessment of decision tools in actual management scenarios provides a major opportunity to bridge the science and practice of invasive plant management. Greater management-science linkages may improve the ability of managers to maintain or enhance ecosystem services with conservation incentives programs and publicly funded invasive plant management programs.

13.4.3 Socioeconomic Dynamics of Invasive Plant Management

A number of key socioeconomic factors that influence development and deployment of long-term integrated invasive plant management programs have been identified over the past two decades. The human dimensions of invasive plant management contribute as much as, or more to the success or failure of invasive plant management as do the heterogeneous and stochastic environmental conditions in which management decisions are made (Chap. 8, this volume). Therefore, identification and mitigation of some of these factors represents key opportunities to advance the adoption and impact of invasive plant management programs on rangeland (Sutherland et al. 2004; Briske et al. 2008b). While invasive plant control failures have commonly been linked to factors such as insufficient policy, funding, or scientific knowledge, the influence of diverse and complex socioeconomic conditions is becoming increasingly recognized (Hershdorfer et al. 2007; Epanchin-Niell et al. 2010). The concept of management mosaics has been used to describe the increasing extent of rangeland fragmentation in the western United States and the increased diversity of land use objectives within these fragmented landscapes (Epanchin-Niell et al. 2010). As fragmentation and diversity of land use goals increases, invasive species control becomes more difficult because it alters the costs and incentives for invasive plant control and prevention. These fragmented management mosaics pose a collective action problem because there is little economic incentive for a landowner to control invasive plants unless surrounding neighbors control invasive plants as well (Hershdorfer et al. 2007; Epanchin-Niell et al. 2010). As fragmentation increases, the number of adjacent landowners also increases. As each manager becomes responsible for managing a smaller portion of the landscape, their optimal invasive plant management actions are increasingly dependent on the invasive plant management decisions of neighbors. A greater number of landowners also mean a greater likelihood that different landowners will have different incentives, policies, or practices for invasive plant management. Landowners with higher control incentives than adjacent neighbors bare a larger proportion of control costs than landowners with neighbors having similar or lower control incentives (Wilen 2007). Land owner interaction can be addressed in one of three possible approaches:

top down regulation by centralized government, bottom-up self-governing efforts, and middle out civic environmentalism efforts (DeWitt et al. 2006). Each approach can have a critical role in determining the amount and impact of invasive species management. However, these approaches do not produce the same outcomes in all management situations, and in some cases certain approaches can have negative effects on invasive plant management adoption, coordination, and impact (Hershdorfer et al. 2007; Tanaka et al. 2011).

While numerous opportunities exist to evaluate how these different socioeconomic approaches may enhance long-term progress of integrated invasive plant management programs, actual and perceived costs-benefit ratios and how these change with changing land ownership are not the only socioeconomic dimensions influencing adoption. Adoption is also highly tied to belief systems, perceived ease and risk of implementation, compatibility with the agricultural enterprise and time period in which results can be observed (Didier and Brunson 2004; Tanaka et al. 2011). In many cases producers adopt non-cost effective conservation practices because they have strong lifestyle and conservation values (Didier and Brunson 2004; Brunson and Huntsinger 2008). In other cases, adoption occurs because producers or managers believe the practice has economical or natural resource value, even if the bulk of existing data is to the contrary (Sutherland et al. 2004; Briske et al. 2008b). In some instances, adoption is less influenced by perceived practice outcomes than the ease in which the practice can be learned or applied and the impacts of the practice observed (Didier and Brunson 2004; Tanaka et al. 2011). Therefore, as the impacts of increasingly complex socioeconomic landscapes on rangeland invasive plant management are quantified and major adoption barriers are identified, there are large opportunities to link research and extension efforts that overcome these socioeconomic barriers to deploying invasive plant management programs.

13.4.4 Identifying and Developing Management Strategies and Goals

Ecological restoration is predominantly focused on the recovery of functional plant communities as plants have a controlling influence on energy flows, hydrology, soil stability, and habitat quality (Young et al. 2005; Kulmatiski et al. 2006; Pocock et al. 2012). Consequently, invasive plant management is often tied directly to broader goals and paradigms of ecosystem restoration. Recent conceptual advances have argued for a more deliberate thought process to determine how ecosystem restoration targets, including invasive plant management efforts, are identified (Hobbs et al. 2011; Monaco et al. 2012; Jones 2013). Systems targeted for invasive plant management often are highly modified and a number of ecological and socioeconomic variables constrain realistic and practical restoration targets. Collectively, this line of thinking has argued that restoration ecology, invasive plant management, and conservation biology are all subsets of the broader field of intervention ecology (Hobbs et al. 2011). One major implication of this inclusive perspective is that it allows managers to select among a number of potential reference states and outcomes on which to base their management goals and strategies (Monaco et al. 2012).

State-and-transition models (STMs) have emerged as the leading conceptual framework to describe vegetation dynamics and to assess management scenarios on rangelands (Quetier et al. 2007). STMs are qualitative flowcharts that describe potential alternative stable vegetation states on individual ecological sites established by different combinations of soil and climate (Chap. 9, this volume). These models also identify potential thresholds between states and the existence of restoration pathways that may potentially reverse transitions between states. In most cases, alternative states, transitions, and restoration pathways are based on management experience and expert opinion. STM models provide managers a general framework to identify potential management goals and some basic understanding and tools to pursue those goals. This provides an opportunity for managers to consider ecosystem states that may represent more realistic goals than the historical reference site given current socioeconomic and ecological constraints. On public lands, a broad set of stakeholders are vested in invasive plant management and restoration decision-making and different stakeholder groups can have different values and goals (Brunson and Huntsinger 2008; Brunson and Tanaka 2011). Framing management decision-making under the general concepts of intervention ecology and multiple alternative states, as well as emphasizing ecosystem function and services instead of historical benchmarks, has provided managers a greatly improved foundation for setting management goals and selecting appropriate tools to achieve identified goals.

A major practical implication of recognizing more than one reference state to set management targets and select appropriate tools is an opportunity to incorporate plant material genetics and selection of desired species into invasive plant management strategies. In many management situations controlling invasive plants does not result in long-term reduction in invasive plant abundance unless desired plant species are sown following invasive plant control (Sheley et al. 2011a). Historically, two competing paradigms have emerged which included a "local is best paradigm" that argued for use of plant material from local provenances to preserve genetic, cultural, or social values. This paradigm assumes that locally collected material would be best adapted to local conditions (Jones 2013). The alternative paradigm argues for the development and use of plant material that could excel in a particular function (typically productivity) across a range of environmental conditions. Over the last two decades the plant materials emphasis has shifted from a taxonomic focus of how plant material is developed and incorporated into restoration towards a greater understanding of how functional trait variation in potential plant material may contribute to management goals, given existing ecological and socioeconomic constraints (Drenovsky et al. 2012; Jones 2013). By considering the possibility of multiple alternative states and that different functional traits may play different roles in each of these states, managers have an improved ability to make decisions about what types of plant material to use given their identified reference state and associated management goals. Managers have a framework to identify when local genetic material is appropriate and when enhancing or altering genetic variation among selected species may be appropriate for various management applications.

13.5 Novel Ecosystems: Are They Useful for Rangeland Application?

Impacts of invasive plants are arguably greatest on "new world" continents that were settled by European immigrants and their traditional agricultural and land management strategies. These immigrants aimed to change "foreign and unique ecosystems" to deliver services that their European culture was more accustomed too (Crosby 1986). For example, both Australia and New Zealand had Commonwealth plant introduction societies, whose aims were to replace the "inferior" plant species of these foreign ecosystems with "superior" plant species providing more for the needs of European culture (Cook and Dias 2006). European landscapes have been managed in this manner for centuries. This establishes that novel ecosystems are not a new concept (Perring et al. 2013). Not only have Europeans dramatically altered their landscapes, but Australian aboriginals and indigenous North America's had also altered landscapes prior to European arrival with use of fire for cultivating crops, as well as hunting practices (Gammage 2012). People are dependent on natural ecosystems for food, water, and their overall livelihoods. Ecosystem conversion becomes an issue when land use is changed, often leading to instability and degradation, or when management actions, such as the introduction of exotic species for forage production or erosion control, did not succeed.

The concept of novel ecosystems is useful in that it provides new terminology for describing complicated ecosystem whose original function, structure, and use are no longer accessible in the short or medium-term. Furthermore, it provides an explicit model for why managers may choose an alternative stable state, rather than attempting to return to the historical state. The concept of novel ecosystems needs to be packaged with other tools, such as alternative states models and adaptive management, which extend beyond new terminology to create a strategy with the goal of controlling invasive plants and favoring more desirable plant species.

13.5.1 Invasive Plant Control Strategies Are Dependent on Goals

As mentioned above, restoration is considered a science and practice that is driven by clearly defined goals to explicitly define realistic short-, medium-, and long-term goals (Hobbs 2007). Goals change depending on land use needs and the state within which an ecosystem resides. The length of time that a site should be managed under each restoration goal will depend on the extent to which biotic and abiotic factors of the ecosystem are degraded (e.g., nutrient cycling, hydrology, energy flows, and native species richness). For these reasons, restoration goals should be flexible so that changes can be made to management strategies when systems do not react according to predictions.

Ecological evidence from more than 20 years of research indicate that increasing number of species within a plant community can produce beneficial effects to ecological functions such as increased production and nutrient cycling (Tilman et al. 1997; Hector et al. 1999). There is agreement that biodiversity matters intrinsically, but also for other values (Naeem and Wright 2003), although how and why it matters remains contested (Kaiser 2000; Naeem and Wright 2003).

The debate has largely concentrated around two hypotheses. One is the "niche complementarity" hypothesis, which proposes that species-rich communities are able to access and utilize limiting resources more efficiently because they contain species with a diverse set of ecological traits. The ecosystem is thought to be more functionally "complete" because species complement each other, allowing them to optimize resource use (Tilman et al. 1997; Hector 1998). An alternative is the "sampling effect" hypothesis, which proposes that more biologically diverse communities have increased productivity because they are likely to contain at least one species that is particularly efficient in how it uses resources. That is, only one or two species within the community may be largely responsible for most of the production. In this case, the subordinate and transient species present may not immediately contribute to functioning of the system, but their presence could provide an ecological buffer by responding to changes in environmental conditions or disturbance regimes (Grime 1998). Recent evidence suggests that when the multiple services provided by grasslands are considered, even higher levels of biodiversity are needed to maintain stability (Isbell et al. 2011).

The benefits of diversity are not limited to ecosystem functions such as production and nutrient cycling, but also increased resilience—defined as the ability to recover after disturbance (Hautier et al. 2014). This idea is based on the insurance hypothesis of ecosystem stability (McCann 2000): an ecosystem with more species contains a diverse set of traits, and therefore, has a higher likelihood of recovering from disturbance (McCann 2000; Naeem and Wright 2003; Suding et al. 2008). Because of the role biodiversity plays in maintaining key functions and building resilience to change, there is increasing recognition that encouraging diversity in rangelands is not counter to production. Instead, focusing management activities on both biodiversity and production could, in the long-run, ensure the sustainability of production and environmental integrity (Firn 2007). These issues of sustainable production and resilience are highly topical, considering predictions for an increase in extreme rainfall variability with global climate change (Hellmann et al. 2008).

However, ecological evidence also suggests that very high biodiversity may reduce ecosystem stability (Pfisterer and Schmid 2002). For example, grassland plots with the lowest species diversity recovered more quickly from drought conditions (Pfisterer and Schmid 2002). These results along with those of other studies (Wardle et al. 1997; Firn et al. 2007) suggest that what matters may not be just the number of species, but the "quality" of the biodiversity; more specifically, the collective traits of the species present and how they respond to perturbations, and in turn, how these responses effect functionality (Lavorel and Garnier 2002; Suding et al. 2008). An increase in "quality" species within a plant community may contribute considerably to function and resilience, although choosing the optimal set of plant traits may lead to altered abiotic and biotic conditions such as fire frequency, grazing intensity, and nitrogen deposition. We infer from this work that with a greater number of species there is a higher likelihood that the "right" species will be present to maintain ecosystem function.

13.5.2 Adaptable Theoretical Framework for Recovery of Degraded Communities

Restoration objectives for degraded ecosystems should be clearly defined for the success of management actions, monitored to determine restoration success and then adapted when necessary (McCarthy and Possingham 2007). These objectives should be more holistic than simply targeting the invasive species, adopting the whole-ecosystem approach discussed above (Zavaleta et al. 2001). Three broad objectives are suggested for the restoration of degraded rainforests, each representing different levels of trade-offs between biodiversity and socioeconomic values (Lamb et al. 2005) (Fig. 13.5). The benefits of adapting a simple framework to underpin the design of restoration goals for rangelands are that trade-offs between multiple objectives can be clearly defined and progress monitored and evaluated in a systematic manner, and most importantly, that strategies and perhaps even broad objectives changed as knowledge of the system increases.

Reclamation involves the complete conversion of an ecosystem to a monoculture of a highly productive species, with the prime objective to recover production (Lamb and Gilmour 2003; Firn et al. 2013). In a degraded rangeland dominated by an invasive plant, the forage species chosen for reclamation should be palatable, high in nutritional content, and competitively superior to the invasive species, so as to gain and maintain dominance. If the establishment of another species is successful this approach could act to increase the livelihood of enterprise operators once the high cost of establishment is recovered (Fig. 13.5; Table 13.2). It could also act to increase income in the long-term; however, this will depend on the consistency of the management practices, environmental conditions, and the disturbance regime over time. Because the prime objective is the recovery of productivity, this approach may reduce biodiversity values even further from the degraded state (Fig. 13.5).

In contrast to reclamation, the goal of restoration involves returning the assemblage of species that were present prior to the dominance of invasive plant, and here the main goal may be to increase biodiversity values (Fig. 13.5). Biodiversity values could be defined as native species richness and abundance, conservation of threatened species, and/or properties of an ecosystem. To apply this approach, production related activities such as grazing livestock may need to be excluded to provide the plant community with an opportunity to recover over time. Sites that have been severely degraded may also need costly strategies to reduce soil nutrients, and return species that are no longer present in the landscape and seed bank. If an area is protected then social and economic value, in terms of income, may need to cease.

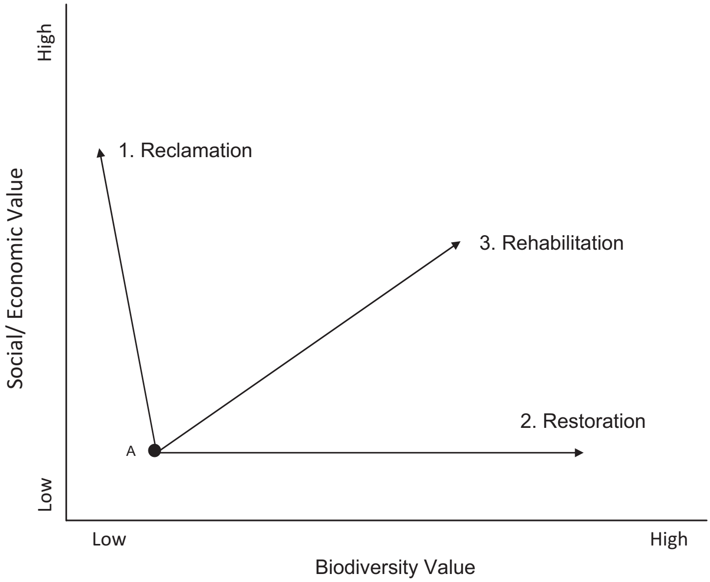

Fig. 13.5 Conceptual model of the trade-offs between approaches for the recovery of a degraded rainforest (adapted from Lamb and Gilmour (2003), Lamb et al. (2005)) including reclamation, rehabilitation, and restoration. Point *A* represents an ecosystem that is degraded and each vector a different theoretical approach to ameliorate both the biodiversity value and the social/economic value from low to high. Reclamation and restoration represent the social/economic and the biodiversity value extremes respectively; while, rehabilitation presents a one to one trade-off between biodiversity and social/economic values

Table 13.2 Potential short- and long-term outcomes for the application of the different approaches to recovery social/economic and biodiversity values in degraded ecosystems

	Potential		Potential	
	Short-term outcomes		Long-term outcomes	
Approaches	Benefits	Costs	Benefits	Costs
Reclamation	Income	Management biodiversity	Income	Biodiversity resilience management
Rehabilitation	Income	Lost income management	Income biodiversity resilience	Lost income management
Restoration	Biodiversity	Lost income management	Biodiversity resilience	Lost income management

Income is defined as the profit received from products derived from the land and costs is defined as the expenditure to manage and produce these products or loss of values such as biodiversity or income

Management costs may then increase from the original degraded state in both the short and long-term, while biodiversity may increase over the short and long-term. An increase in biodiversity could restore functional resilience over the long-term (Table 13.2). There is considerable argument as to whether restoration is an achievable goal (Lamb and Gilmour 2003; Suding et al. 2004; Hobbs et al. 2006) because the original suite of species may not be known and no longer present in the seed bank if the ecosystem is intended to recover by natural regeneration (passive management). Setting targets and milestones for assessing progress may also be difficult because information on the precise dynamics that governed the original ecosystem may also not be known (Lamb and Gilmour 2003; Suding et al. 2004).

A goal of rehabilitation emphasizes biodiversity and socioeconomic values equally, in a one-to-one trade-off (Fig. 13.5) (Lamb et al. 2005). This acknowledges the short-term value of returning some biodiversity, while continuing to utilize productive output. The costs of this approach includes a loss of some income because it is likely that production will decline as some implemented strategies will encourage the return of slower growing native plant species or more desirable exotics (Table 13.1). Over the long-term, the benefits may include more consistent and reliable income and, if management practices maintained biodiversity values, improved functioning such as nutrient cycling and resilience.

Whether the main goal of a resource manager is production, biodiversity conservation, or both, a rehabilitation approach is an effective option for rangeland improvement in the short-term. Investing time and money into the intensive actions needed for either reclamation or restoration will not necessarily deliver their respective and often singularly focused outcomes, particularly in the short-term.

What is clear is there are short-term benefits for the socioeconomic welfare of managers in slowly managing the transition of an invasive plant state to a more diverse state containing a greater proportion of native species. The intensive actions needed to instigate reclamation or restoration does not necessarily provide the socioeconomic values or the biodiversity values desired, without significant additional investments of time and money. Controlling invasive species with intensive strategies can detrimentally effect the remaining native species and lead to further degradation (Rinella et al. 2009), and the method of plant control has a strong effect on ecosystem response (Flory and Clay 2009).

13.5.3 A Revised Rehabilitation-Novel Ecosystem Model

The application of invasive plant control in rangelands is complex because the management practices used can also detrimentally impact functional integrity, including production and nutrient cycling. In this regard, invasive plant control practices themselves can result in further short- and long-term income loss, social distress, and environmental degradation. Common control strategies for invasive plants have generally been designed to eradicate them, but in an enterprise that is dependent on forage availability for livestock consumption, removing even low quality vegetation could prove more detrimental to the livelihood of enterprise operators than the impacts of the invasive plants. Removing available plant cover, especially in arid and semiarid systems, can accelerate soil erosion, nutrient loss, and can lead to the dominance of other invasive plant species. In this case, maintaining a small population of undesirable invasive grass species may be better, in the short-term, then reducing the majority of plant cover, including available forage, and waiting for the system to recover. Instead, control strategies are needed that balance multiple objectives, including short-term income for managers, reasonable costs for establishment and maintenance of productive species, and increased biodiversity to improve ecosystem functioning and resilience.

A revised model for the socioeconomic and biodiversity values in the rehabilitation of rangelands is needed based on the justification provided above (Fig. 13.6).

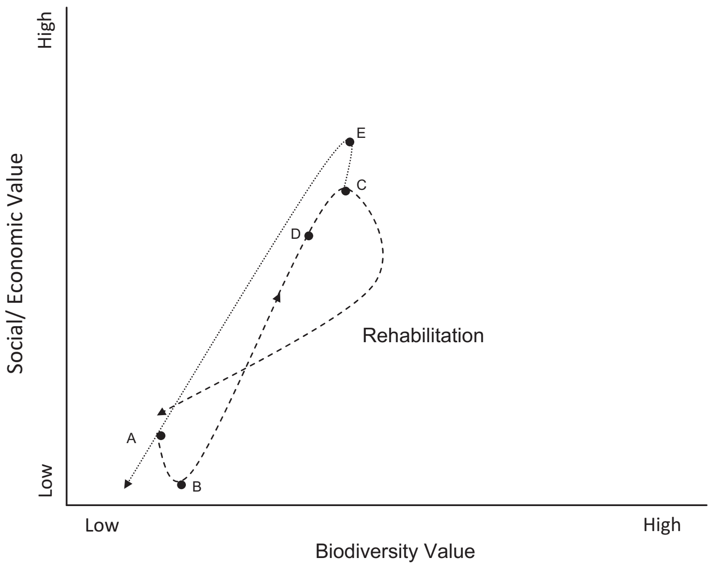

Fig. 13.6 Revised conceptual model for the rehabilitation approach for pastoral improvement. Point *A* represents a degraded state; point *B* the initial cost for the rehabilitation efforts and loss of income; point *C* represents the maximum amount of social/economic value that can be gained from a diverse pasture community; Point *D* represents the maximum social/economic value if environmental conditions become unfavorable; and point *E* represents the initial increase in income associated with over-utilization of the resource and shows the projected decrease in both social/economic and biodiversity value. Management efforts should aim to maintain the ecosystem at point *C* where both biodiversity and social/economic values can be optimized (species that transform an ecosystem), or fragmentation of existing habitats (Seastedt et al. 2008). Theses biotic changes have accelerated in the past few decades due to increased human activities arising from the breakdown of biogeographic barriers and the global human-mediated transport of non-native species (Seastedt et al. 2008). This, in combination with abiotic changes, have increased the rate of appearance of new, non-historical, novel environments, unique species combinations, and altered ecosystem functioning (Hobbs et al. 2006, 2009)

Point *A* represents the socioeconomic and biodiversity values of a degraded state. Point *B* represents the starting point of a rehabilitation approach and illustrates the initial drop in social and economic value associated with the financial costs, time, and effort to increase biodiversity value. Depending on the time since invasion and extent of ecosystem degradation, this drop in socioeconomic value could be higher or lower. Point *C* represents a maximum level of socioeconomic value that could be gained from a rehabilitated ecosystem where biodiversity and socioeconomic goals are both key. Grazing can increase the diversity of species present within ecosystems, particularly if the dominant species are palatable (Fensham et al. 2014; Lunt et al. 2007) and depending on the climatic region and evolutionary history of grazing (Milchunas et al. 1988). In contrast to the model suggested by Lamb et al. (2005) for rainforest communities, there is a higher gain in socioeconomic value with every unit of increase in biodiversity value in pasture communities. We have represented this with a steeper curve (Fig. 13.6) then the model proposed in Fig. 13.5. The grazing pressure should be maintained at a level that encourages diversity within the community, but does not reduce the abundance of key dominant species to a level that opens an opportunity for less desirable species to become established. For this reason, the shape of the curve after point C will likely vary widely depending on the biotic and abiotic characteristics of specific ecosystems.

The maximum socioeconomic benefit gain can be reduced if the favorability of environmental conditions decreases, as represented by point *D* (Fig. 13.6). Because rangelands are generally quick to respond to environmental fluctuations, grazing pressure, and therefore the socioeconomic value would need to be scaled back if conditions became unfavorable for an extended period. This represents the principles of adaptive management because grazing pressure and its impact on biodiversity would need to be regularly assessed to ensure it was at a suitable level for social welfare and to not jeopardize benefits associated with biodiversity. Point *E* represents the system response if grazing pressure is not matched with the qualities of the grazed ecosystem and the environmental conditions. In this case, there may be short-term gains in socioeconomic value due to over-utilization of the resource, but in the long-term any gains in biodiversity and socioeconomic values may be lost.

13.6 Future Perspectives

While the concept of novel ecosystem is not new, it is a useful construct for controlling invasive plant species as it provides explicit terminology for why managers may choose not to return an ecosystem to its historical state. The concept of novel ecosystems needs to be packaged with other tools, such state-and-transition models and adaptive management, which provide holistic and flexible approaches for controlling invasive plants and also considering how abiotic and biotic factors are altered so as to favor more desirable plant species. Explicitly defining reclamation, rehabilitation, and restoration goals is an important addition to the novel ecosystems concept and allows for a more detailed definition and the identification of simple, realistic targets and goals. Most importantly, reclamation, rehabilitation, and restoration goals can be changed as ecosystems and societal needs change, and thus, provide the flexibility and practicality needed for adaptive management practices.

13.7 Summary

Global rangelands provide many important ecosystem services, including food, fiber, clean water, recreational and open space, minerals, religious sites, aesthetics, plant and animal biodiversity, wildlife habitat, and carbon sequestration. Many traditional management practices, coupled with the introduction, establishment and proliferation of invasive non-native plants have led to broad-scale changes in plant communities. The most important of these invasive plants can economically impact numerous aspects of livestock production, including forage yield and quality, animal health and weight gain, and the quality of meat, milk, wool, and hides. In addition, they can reduce the ecological integrity of rangeland communities by altering fire frequency, increasing erosion, depleting soil moisture and nutrients, and reducing plant biodiversity and wildlife habitat and forage. The impacts of invasive plants on community diversity and structure can lead to ecosystem instability, and often irreversible transformational changes within the system. However, there are also many instances where invasive plants can provide benefits to the ecosystem, and thus there is often a trade-off between negative and positive impacts. As a result, the focus of many restoration efforts need to shift from preserving the historic species assemblages to conserving and maintaining a resilient, functional system that provides diverse ecosystem service, in addition to supporting human livelihoods.

Abiotic and biotic feedback mechanisms can be modified by invasive plants to completely and irreversibly transform historic communities to novel ecosystems with different species composition, ecosystem services and function. This can occur though abiotic changes in climate, land use, pollution and nutrient enrichment, altered disturbance regimes, urbanization, or biotic changes associated with local extinction of keystone species, introduction of invasive ecosystem engineers, or habitat fragmentation. In these novel ecosystems, land managers will need to consider how to feasibly and economically approach long-term management that maximizes a different set of ecosystem functions and services, yet maintains ecological and ecosystem resilience. The primary challenge will be to determine the types of changes that are intrinsically desirable and beneficial, without creating other serious problems or further degrading the system.

As many rangelands have experienced anthropogenic changes characteristic of novel ecosystem functioning, managing invasive species in this new era will require proactive strategies and conceptual frameworks that offer greater success in achieving desired outcomes. In the last 25 years, invasive species management has evolved to incorporate integrated strategies that are guided by both successional theory and process-based manipulations of abiotic and biotic factors. During this time frame, invasive species management has also recognized the need to take a broader ecosystem perspective, which has been facilitated by the development of numerous theoretical frameworks that illustrate how management inputs can best be applied to modify specific ecological processes, at specific stages along the invasion continuum. It has also become clear that proactive invasive species management must adopt preventative control strategies that are known to be more economically feasible for rangeland application.

The role of functional species groups and functional species traits have emerged as valuable predictors of ecosystem functioning and as a way to adopt management strategies that foster greater invasion resistance. In particular, research on functional species groups and functional species traits hold tremendous promise to support assessment of ecosystem susceptibility to invasion and the selection species that offer the best trait-matching to compete with invasive species under specific abiotic stresses.

As invasive rangeland plant management has shifted away from a focus on tools and technology for short-term invasive plant control and toward an emphasis on identifying, managing, and monitoring ecological processes that drive invasion, many conceptual advances have developed that have major implications for understanding constraints and opportunities for rangeland invasive plant management. An emerging opportunity that requires greater attention is a clear definition of the relationship between invasive rangeland plants and ecosystem services, and to identify species and management scenarios where management inputs yield the greatest aggregate return on ecosystem services. A second opportunity centers on bridging the science and practice of invasive plant management by evaluating adoption barriers and assessing impacts of invasive plant management decision tools under actual management scenarios. In addition, as the impacts of invasive plant management on socioeconomic systems are quantified and major adoption barriers are identified, there are large opportunities to link research and extension efforts designed to overcome these socioeconomic barriers to deploying sustainable rangeland invasive plant management programs. Third, while a broad set of stakeholders are vested in invasive plant management and restoration decision-making, different stakeholder groups can have different values and goals, which makes management decision-making challenging. Framing management decision-making under the general concepts of intervention ecology and multiple alternative states, as well as emphasizing ecosystem function and services instead of historical benchmarks, provides managers a greatly improved foundation for developing consensus toward management goals and selecting appropriate tools to achieve desired outcomes.

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