Chapter 27
Restoration Within Protected Areas: When and How to Intervene to Manage Plant Invasions?

Loralee Larios and Katharine N. Suding

Abstract Despite the on-going efforts to set aside land for conservation, biodiversity is increasingly being threatened by factors such as invasive alien species that do not recognise these boundaries. Invasive species management programmes are widely incorporated into protected area management plans; however, the success of these programmes hinges on the ability to identify when a system will be able to recover after invader control and eradication efforts and when further intervention will be necessary to aide recovery. Invasive alien plants can alter ecosystem attributes to produce strong legacy effects that prevent the recovery of a system. Here we provide a framework for how to identify and incorporate recovery constraints into restoration efforts. Identifying recovery constraints can help improve how ecological theory – assembly rules, ecological succession, and threshold dynamics – can be used to guide restoration efforts.

Keywords Assembly rules • Ecological succession • Threshold dynamics • Recovery constraints • Invader impacts

27.1 Introduction

Protected areas (PAs) serve as the primary method to maintain and protect global biodiversity (UNEP-SCBD 2001). Therefore, an important goal in PAs is to minimise threats to biodiversity and maintain ecological communities in their natural states (Lockwood et al. 2006). Protected areas can manage certain threats such as deforestation or poaching, but even the most well managed reserves are still

L. Larios (✉) • K.N. Suding
Department of Environmental Science, Policy and Management, University of California Berkeley, 137 Mulford Hall, Berkeley, CA 94720-3114, USA
e-mail: llarios@berkeley.edu; suding@berkeley.edu

L.C. Foxcroft et al. (eds.), *Plant Invasions in Protected Areas: Patterns, Problems and Challenges*, Invading Nature - Springer Series in Invasion Ecology 7,
DOI 10.1007/978-94-007-7750-7_27,

susceptible to threats such as climate change, pollution, and invasive alien species that do not recognise these conservation boundaries and fence lines. We focus on managing one of these threats, invasive alien plant species (IAPs), in PAs. In response to this threat, many PAs have implemented large-scale invasive species management programmes that employ prevention, eradication, and control strategies aimed at slowing or stopping the process of invasion (Foxcroft and Richardson 2003; Doren et al. 2009b).

Increasingly, a challenge in this process is that simply removing the invasive species is not sufficient to restore native biodiversity. A recent review by Kettenring and Adams (2011) found that invasive removal successfully reduced the cover of invasive alien plants (IAP), but did not always result in native species recovery. Further intervention – with a focus on restoration – may be necessary to take into account the impacts an invader has on a system (D'Antonio and Meyerson 2002), as well as address recovery constraints of the native community. However, these additional intervention actions can be costly in terms of time and money and, in some cases, they have unintended consequences and actually slow recovery (Zavaleta et al. 2001; Hobbs and Richardson 2011). Integrating additional intervention efforts within an existing protected area management plan can be complicated by a variety of factors such as limited resources (e.g. staff and infrastructure), legal mandates under IUCN management categories, or differing agendas among the stakeholders in the governance group (Keenleyside et al. 2012). The isolated nature of PAs requires intervention efforts to be a concerted endeavour with agencies/land owners outside of the reserve, further complicating the success of management efforts.

In this chapter, we focus on this conundrum: when should we expect a system to recover without additional restoration efforts after invasive species control efforts? And when is further intervention necessary for recovery? Resources are often scarce for PAs, with eradication and control of invasive species often consuming a disproportionate amount of reserve budgets (D'Antonio and Meyerson 2002). Identifying necessary points of intervention prior to action is therefore critical for successful protected area management. We begin by providing an overview of invader impacts that may constrain and preclude the recovery of a system after IAP management. We then explore key ecological theories that can be used to guide restoration strategies. Finally, we discuss how land managers could adjust restoration efforts depending on the constraints present in the system.

In this chapter, we consider restoration to include both IAP control and eradication efforts as well as additional actions to aid native recovery. As emphasised elsewhere in this volume, invader management plans in PAs often include control and eradication efforts in tandem with native recovery efforts. Here, we focus on restoration after the invaders are removed or reduced. The key questions are thus: when will passive recovery following these efforts be sufficient to recover desired native communities, and when will active intervention (*sensu* Suding 2011) be needed?

27.2 Invader Impacts and Recovery Constraints

As PAs operate under the mandate to protect local biodiversity, the continuing and growing threat of IAP invaders on native biodiversity has made IAP management a priority for PAs (Macdonald et al. 1988; Vitousek et al. 1997; McNeely 2001). Understanding invader impacts on ecological communities is an important first step in understanding how native communities may recover following IAP control. We particularly focus on IAP legacy effects in PAs, where the impacts of invasion persist even after invader control or eradication. In these cases, removing the invader may not always lead to successful recovery of the degraded system (D'Antonio and Meyerson 2002); additional management and restoration actions may be necessary to put the native community on a path to recovery (Suding et al. 2004). Alternatively, if an IAP does not have strong legacy effects, additional efforts may not be necessary and native communities should be expected to passively recover following control efforts. Importantly, the impacts of invasion may occur either progressively with invader abundance or abruptly once the invader reaches a certain abundance threshold (D'Antonio and Chambers 2006; Didham et al. 2007). Consequently, whether active or passive restoration is necessary may depend on the pattern as well as the nature of legacy effects.

Native species recovery may often be limited by dispersal following IAP control (Galatowitsch and Richardson 2005; Traveset and Richardson 2006). Source populations of native species may be far from the restoration area (McKinney and Lockwood 1999) or seed dispersal networks may be altered in the invaded area (Traveset and Richardson 2006; McConkey et al. 2012). For example, in Australia, recovery of coastal dune communities invaded by *Chrysanthemoides monilifera* subsp. *rotundata* (the South African bitou bush) is limited by poor seed dispersal from existing native vegetation (French et al. 2011), and in New Zealand, native shrublands dominated by *Kunzea ericoides* (kanuka) have a different composition and a smaller abundance of the avian seed dispersers compared to *Ulex europaeus* (gorse) invaded stands (Williams and Karl 2002). Additionally, native seed bank at a restoration site could be diminished if natives have been absent or in low abundance, reducing the potential for recovery from in situ germination (D'Antonio and Meyerson 2002). In southern California, passive recovery of the native coastal sage scrub community is limited due to the depauperate native seedbank in long term invaded alien grassland sites (Cione et al. 2002; Cox and Allen 2008).

Plant invaders can alter disturbance regimes, which may create positive feedbacks that promote invader success (D'Antonio and Vitousek 1992; Mack and D'Antonio 1998). These feedbacks must be disrupted to allow the recovery of a system (Suding et al. 2004). A widespread example occurs when annual grass invaders increase the intensity and frequency of fire (D'Antonio and Vitousek 1992). In the Western United States, for example, alien annual grasses increase fuel loads, which promotes a fire frequency for which the resident community is not adapted (Whisenant 1990). Conversely, IAPs can also impact disturbance regimes by suppressing disturbances (Mack and D'Antonio 1998). *Schinus terebinthifolius* (pepper tree) invasion in Florida's Everglades National Park has suppressed fire intensity by decreasing fuel loads (i.e. understory vegetation), which enhances its own recruitment (Doren and Whiteaker 1990). In these cases, the disturbance regime may not recover following IAP control, and additional actions may be needed to re-establish the disturbance regime needed to support the native community (Davies et al. 2009).

Invasive alien plants can also impact the physical structure of soils by increasing erosion rates or sedimentation rates and directly by affecting substrate stability (D'Antonio et al. 1999), resulting in soil legacies (*sensu*, Corbin and D'Antonio 2004). For example, while increased sedimentation can promote succession and facilitate the establishment of native species in degraded forests in Algiers (Wojterski 1990), increased erosion rates can limit recovery by eliminating habitat for native species and promoting the establishment of introduced species in the South African fynbos (Macdonald and Richardson 1986).

Soil legacies can also influence belowground biological processes that promote IAP abundance and stall native species recovery (van der Putten et al. 2007; Inderjit and van der Putten 2010). An invader can be successful because it is able to escape soil pathogens (Klironomos 2002), and it may also alter pathogen incidence in the native community to reduce competitive effects and facilitate its spread (Eppinga et al. 2006; Mangla et al. 2008). Pathogen loads may slow the recovery rates of communities, as they continue to influence the performance of native species even after IAP removal (Malmstrom et al. 2005). Invasive alien plants can facilitate their invasion by allelopathy (i.e. the release of phytotoxins, which inhibit the growth of neighboring plants; Callaway and Ridenour 2004). For example, high impact invader *Centaurea maculosa* (spotted knapweed) releases a compound that inhibits root growth of its neighbouring plants (Bais et al. 2003). Additionally, *Alliaria petiolata* (garlic mustard), a widespread invader in North American forests, secretes compounds that inhibit the symbiotic mycorrhizal associations of native plants. These altered relationships can prevent the recovery of the community once the invader has been removed due to residual toxins (Perry et al. 2005). Other invaders can alter soil properties such as salt concentrations or soil pH, reducing the potential for subsequent colonization by native species (Vivrette and Muller 1977; Conser and Connor 2009).

Soil legacies also include invader impacts on biogeochemical cycles that alter resource availability (Mack et al. 2001; Ehrenfeld 2003). Nitrogen cycling rates are regularly increased by invaders by altering the microbial community (Hawkes et al. 2005), altering litter quality (Sperry et al. 2006), or directly by nitrogen-fixing species (Vitousek and Walker 1989; Le Maitre et al. 2011). Increased nitrogen availability can result in positive feedbacks that maintain the invaded state, thwarting recovery efforts (Clark et al. 2005). For example, in temperate grasslands in Australia, alien annual species that invade native perennial tussock grasslands can alter nitrogen cycling to favour their own growth. These nutrient changes are sufficient to push the system past a threshold, preventing the recovery of native grasses (Prober et al. 2009). Lastly, invaders can also alter the hydrology of a system via altered transpiration rates, rooting depths, phenology, and growth rates (Levine et al. 2003). *Tamarix* spp. (salt cedar) invasion in the south-western United States has resulted in higher transpiration rates and marginal water loss due to the salt cedar's deeper root system in this water limited system (Zavaleta 2000).

27.3 Ecosystem Models in Restoration

While it is clear that many IAPs have strong legacy effects that can influence the recovery of native communities, it is also important to put these effects in the context of ecological processes that guide the path to recovery (Young et al. 2005; Hobbs et al. 2007; Suding and Hobbs 2009). Conceptual models of ecosystem dynamics such as assembly theory, ecological succession, and threshold dynamics can guide restoration projects by providing insights into these ecological processes (Fig. 27.1). In the following paragraphs we explore these three concepts and how they can guide decisions about when and how to intervene in PAs following invasive plant control efforts. For each, we first present the basic framework, then a case study examining application to restoration in PAs.

27.3.1 Assembly Rules

Assembly theory focuses on how a suite of processes (e.g. dispersal, disturbance, environment, competition) influence which species are able to establish over time (Young et al. 2001; Temperton et al. 2004; White and Jentsch 2004; Hobbs et al. 2007). This framework integrates these processes into a series of filters (dispersal, environmental, and biotic) that act at varying spatial scales, which can explain which species from a regional species pool (large scale) are found in the local community (small scale, Weiher and Keddy 1995; Diaz et al. 1998, 1999). In the context of native species recovery following IAP control, recovery requires that filters at each scale allow native species to establish and persist (Fig. 27.1b). Additional intervention efforts would be focused on the filters that excluded the desired species from recovering (Fig. 27.1b, dashed arrow).

Three general types of filters are emphasised in assembly theory. The first filter that species must overcome is dispersal: species must have dispersal traits that allow them to arrive at a site (Levine and Murrell 2003). As discussed above, invasive plants can increase the dispersal limitation of native species in many ways, creating new barriers to the dispersal filter for some native species. If a species is able to colonise a site, the next filter acting upon it is the environmental filter. To successfully cross the environmental filter, a species must have the suite of traits that allow it to survive the given environmental conditions (Weiher and Keddy 1995; Diaz et al. 1998, 1999). Soil legacies of invasive plants, such as erosion and resource cycling impacts, can alter this filter. An extension to the environmental filter is the disturbance filter (White and Jentsch 2004), which invasive species may

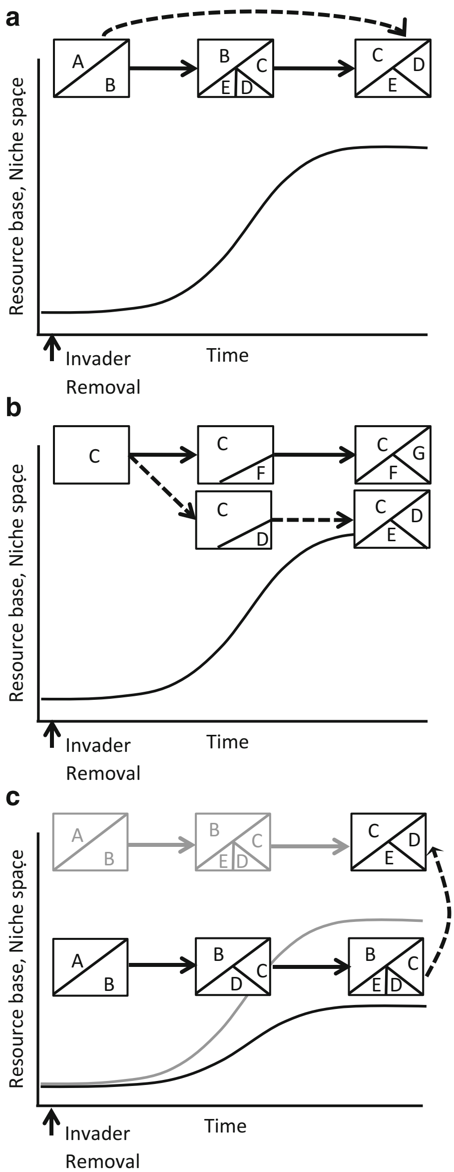

Fig. 27.1 Recovery trajectories after invader removal, (**a**) assuming little invader impact or, (**b**) and (**c**), a legacy of invader impacts. Species composition is symbolised by the *capital letters* and abundance by proportion of each square; the desired goal community is C/E/D. *Solid lines* indicate scenarios with passive restoration after IAP removal; *dotted arrows* indicate restoration intervention. We present scenarios consistent with each of the three ecosystem models of recovery. Successional theory (**a**) is most appropriate in systems where there is little expectation of strong invader impacts. In (**a**), successional theory assumes directional change in species composition over time. If the natural recovery takes too long, land managers can intervene to accelerate recovery (*dashed arrow* in **a**). In systems impacted by invader legacy affects (**b**, **c**), assembly theory and threshold theory may be most appropriate to guide restoration efforts. In (**b**), IAP legacies affect the order of species arrival. Active intervention can focus on adding species,

similarly alter. The final filter in assembly theory is the biotic filter, which restricts the community to those species that can coexist in the presence of interspecific interactions (MacArthur and Levins 1967; Tilman 1990; Chesson 2000). Under the biotic filter, competitive interactions would limit the co-occurrence of functionally equivalent species due to niche limitation resulting in limited similarity among species within community (MacArthur and Levins 1967). Under situations where invasive species have been controlled or eradicated in PAs, we would expect that this biotic filter would be less of a consideration compared to the other filters, but it would be important to manage were reinvasion possible.

The efficacy of active intervention efforts in restoration (e.g. species palette for planting, selection of planned disturbance to limit competitive interactions) can be assessed in this assembly filter framework by equating restoration actions with changes in assembly filters (Funk et al. 2008). For example, seed addition or planting of native species can be viewed as changing the dispersal filter at a site. Similarly, a trait-based approach could increase the success of restoration efforts areas where managers fear invasive species could re-invade following control efforts by identifying a suite of native species with traits similar to the IAP to enhance the invasion resistance of the community, thereby strengthening the biotic filter (Funk et al. 2008).

27.3.2 Case Study 1: California Grasslands

Protected areas such as county parks and reserves within California are often imbedded within a highly fragmented landscape (Greer 2005). In California PAs, alien annual grasses have the potential to gain access to the interior of natural areas by initially colonizing disturbed roadside areas (Gelbard and Belnap 2003). Roadsides can have large inputs of atmospheric nitrogen deposition (Pearson et al. 2000), which can interact with local grassland's N cycling to increase N availability (Sirulnik et al. 2007), and further promote these annual grasses (Padgett and Allen 1999). Furthermore, prolonged dominance of alien grasses within a site can reduce the seedbank of native species and prevent the recovery of a system once the grasses have been removed (Cione et al. 2002).

To evaluate if the biotic filter can be manipulated to slow or stop the re-invasion of aliens after control, Cleland et al. (2013) conducted a restoration experiment along a roadside edge of the Laguna Coast Wilderness Park in southern California

Fig. 27.1 (continued) affecting the order of species arrival, to guide the assembly process to arrive at the target community. In (**c**), recovery may result in a new undesired state due to invader legacy impacts, preventing the successional process that would occur naturally (*grey boxes*). A threshold model may be the most appropriate to apply in cases such as these, where multiple restoration activities would need to be done to overcome this feedback (*dashed arrow*, **c**) (Modified from White and Jentsch 2004) where they manipulated nitrogen availability and added native seeds representing different functional groups (annual/perennial grasses, early/late forbs and N-fixing legumes). In the first year, they removed alien annual grasses and forbs. Then in the second year, they allowed alien species to colonise naturally. Native communities with low N availability and in which early forb seed was added best resisted re-invasion. Thus, they found that by altering resource availability and adding species that have similar phenology to the problematic invader they could manipulate the biotic filter to increase invasion resistance.

27.3.3 Ecological Succession

Successional dynamics, the changes in species composition within a community over time, have been a classic and focal question in ecology since the 1900s (Cowles 1899; Clements 1916; Gleason 1926). Succession traditionally describes the patterns of compositional change after a disturbance (Clements 1916; Pickett et al. 1989) but recent studies have gone beyond describing the patterns to identify the mechanisms, which influence these patterns (Connell and Slatyer 1977; Tilman 1988; Pickett et al. 2009). As successional theory has expanded to incorporate the possibility of multiple successional pathways versus a single climax community (Glenn-Lewin et al. 1992), comparing and analysing successional trajectories has been adopted to describe the temporal change in community composition (Hobbs and Mooney 1995). Once a disturbance occurs at a site, the availability of safe sites and propagules for colonization in conjunction with the impacts of established species determine subsequent successional dynamics (Pickett et al. 1987). In the context of whether to intervene following invasive species control, additional intervention activities can be viewed as either altering or initiating any of these recovery processes (del Moral et al. 2007; Fig. 27.1a).

Ecological restoration can take a variety of approaches to manage succession toward a desired target. The first and simplest approach is to allow succession to occur unaided (spontaneous succession, Prach et al. 2001) and should be a viable option if most abiotic and biotic functioning remain intact after invasive species control (Lockwood and Samuels 2004; Prach and Hobbs 2008). However, in the case of large-scale invasions, natural succession is unlikely to be a viable option as legacies from the invader may influence recovery (Zavaleta et al. 2001). When legacies are present another approach is to assist succession via manipulations to the physical environment and to biotic processes that may be important within the target system (technical reclamation, Prach et al. 2007). Technical reclamation may be necessary if invasion has resulted in the complete loss of any of the overarching processes governing succession (e.g. availability of safe sites, propagules, and species impacts; del Moral et al. 2007; Prach and Hobbs 2008). The third approach, assisted succession, is a combination of technical reclamation and spontaneous succession in which site conditions are initially modified to support native species but subsequent succession is allowed to occur naturally (Prach et al. 2007;

Fig. 27.1a, dashed line). This approach has been implemented within rangeland invasive plant management, by pairing removal efforts with post-removal restoration activities (Sheley et al. 2010). While this framework is similar to assembly theory in that it emphasises identification of processes that constrain recovery, it also emphasises trajectories of community development over time.

27.3.4 Case Study 2: South African Fynbos

The fynbos vegetation in the Cape Floristic Region of South Africa is highly impacted by alien trees and shrubs (*Acacia* spp; Macdonald 1984; Le Maitre et al. 2011). *Acacia spp*. are nitrogen (N)-fixing plants, which can increase soil fertility after an extended presence in an area (Yelenik et al. 2004). They also have a large impact on water resources, as they consume more water than the native vegetation (Le Maitre et al. 2000). Under the national ‘Working for Water’ program, *Acacia* spp. and other woody invasive plants have been targeted for removal (Turpie et al. 2008). Clearing of these invaders is often a combined effort of cutting down the tree/shrub and, for those species that resprout, applying herbicide to the stumps with the felled biomass left on site. It can also involve the removal of the felled material and/or burning (Macdonald 2004). Cleared sites are often allowed to recover spontaneously after treatment; however, the success of passive recovery is often dependent on the type of treatment (i.e. spontaneous succession was the most successful with clearing and removal and the least successful under burning; Blanchard and Holmes 2008). Blanchard and Holmes (2008) found that once the biomass was removed, native species had space to establish and assisted succession approaches were needed. For example, seeding after burn treatments to overcome dispersal constraints can increase the presence of native fynbos vegetation and enhance natural recovery; however, continuous eradication efforts are needed until the large *Acacia* seedbank is reduced as natural wildfires may continue to promote the establishment of *Acacia* after initial removal (Milton and Hall 1981).

27.3.5 Threshold Dynamics

Ecological thresholds are a breakpoint between two systems that, when crossed, result in an abrupt change in community states (Holling 1973). Thresholds occur due to positive feedback mechanisms, which make systems resistant to change (Folke et al. 2004; Suding et al. 2004). While successional models and recovery pathways apply to many situations of recovery following IAP control, threshold models can help explain why some systems are not able to recover once the invader has been removed (Prober et al. 2009). In the context of these ‘stuck’ systems, threshold models point to the importance of breaking these positive feedbacks in order to facilitate recovery (Fig. 27.1c).

A useful framework for incorporating ecological thresholds into management has been to divide thresholds into two stages. The first stage is the biotic threshold, which can be identified by changes in vegetative structure or composition (Friedel 1991; Whisenant 1999). The second stage is an abiotic threshold, which identifies changes in ecosystem functioning (Whisenant 1999). Because impacts on functioning are thought to lag behind biotic changes, a system is thought to first encounter the biotic threshold and subsequently the abiotic (Whisenant 1999; Hobbs and Harris 2001; Briske et al. 2005). Invasive alien plants that trigger biotic threshold changes may be easier to control than those that cause biotic and abiotic threshold changes. Invaders that cause the system to cross both thresholds (ecosystem engineers, *sensu* Jones et al. 1994) make the success of restoration efforts highly uncertain (Ehrenfeld et al. 2005; Kulmatiski 2006; Doren et al. 2009a). Once management has identified key variables that can indicate whether a threshold has been crossed, this knowledge can be incorporated into management to identify when and what management efforts are needed to increase the success of control and subsequent restoration efforts (Foxcroft and Richardson 2003; Doren et al. 2009b).

27.3.6 Case Study 3: Australian Subtropical Rainforests

One of the world's most notorious invaders, *Lantana camara* (lantana), has invaded and replaced much of the native vegetation in the subtropical forests in eastern Australia (Lowe et al. 2000; Bhagwat et al. 2012). *Lantana camara* was introduced as an ornamental shrub in the mid-nineteenth century (Swarbick 1986) but has rapidly spread to the detriment of native diversity, including PAs within Australia's national parks. Many of the national parks within eastern Australia are isolated within a highly disturbed system, a problem common to many PAs globally (Fox et al. 1997). Edges between the reserves and disturbed areas (e.g. old agricultural fields in Australia) make reserves vulnerable to weedy invaders such as *L. camara*, which readily spread across disturbed landscapes (Gentle and Duggin 1997; Stock 2004).

However, this landscape also provides an opportunity to investigate the dynamics that allow this invader to invade pristine habitats. Stock (2004) and Gooden et al. (2009) monitored *L. camara* and native plant abundance in national parks in eastern Australia and were able to identify two separate thresholds. After measuring *L. camara* cover and canopy cover in gaps in two national parks, Stock (2004) identified a first invasion threshold: forests whose canopy cover is 75 % native species can prevent the establishment of *L. camara*, because the woody invader is shade intolerant in those forests. If *L. camara* reaches 75 % cover, however, the system crosses a second biotic threshold identified by Gooden et al. (2009) in which native species richness falls dramatically, likely due to *L. camara* effects on soil fertility (Bhatt et al. 1994) and soil seed banks (Fensham et al. 1994). These thresholds, which identify when a community can resist invasion and when invader impacts begin to increase dramatically, are being used to guide an integrated management plan (Stock 2005).

27.4 Addressing Recovery Constraints in the Context of the Three Models

The ability of ecosystems to recover after IAP control greatly varies and is often contingent on the system's intrinsic rate of recovery, its level of degradation, and its surrounding matrix (Jones and Schmitz 2009; Holl and Aide 2011; Gaertner et al. 2012). This variability makes it difficult to assess when land managers should intervene and implement additional restoration practices or leave a system to recover naturally (passive restoration, *sensu* Suding 2011). For example, in the south-western United States, invader removal without any paired plantings of native flora can detrimentally affect local fauna (Zavaleta et al. 2001), and yet, active plantings in the tropics may prevent the establishment of native flora and slow natural recovery (Murcia 1997). Furthermore, within PAs, additional complicating factors such as land use and pollution are either well documented or less severe than in non-protected areas. Therefore, restoration efforts conducted within PAs after IAP removal can help improve our collective understanding of invader impacts and recovery constraints. In this section, we suggest a series of steps to decide when additional intervention following IAP control may be needed.

First, an understanding of the extent of IAP impacts and whether they will persist following invader removal is critical (Sheley et al. 2010). A holistic assessment should try to identify the causes of the invasion as well as impacts of the invasion (see invader impacts section above; James et al. 2010). If a holistic assessment was not initially available, small scale experiments can be used to identify restraints (Kettenring and Adams 2011). Simply observing the natural recovery of a system after control efforts would also aide the decision of whether or not to intervene when assessments are not available (Holl and Aide 2011). If monitoring indicates natural recovery, land managers can use successional theory to make inferences about the trajectory of the system (Sheley et al. 2006; Prach and Walker 2011). However, if monitoring identifies invader legacies, the success of management efforts is contingent on effectively prioritizing and addressing those recovery constraints (Fig. 27.2a; Suding et al. 2004).

Identification of constraints can be done through knowledge of natural history, experimentation (Gaertner et al. 2011; Kettenring and Adams 2011) or research from other sites (e.g. recent reviews of the effects of the invaders *Acacia* (Le Maitre et al. 2011) and *L. camara* (Bhagwat et al. 2012) on ecosystems). If one single factor seems to strongly constrain recovery, natural recovery should be fairly straightforward if land managers can address the single constraint

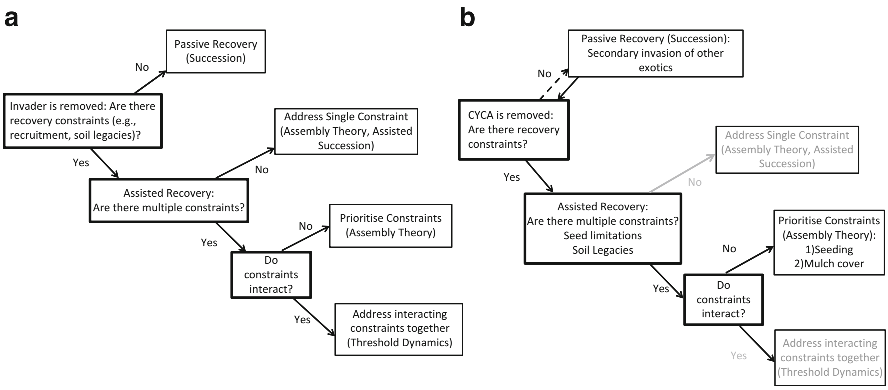

Fig. 27.2 (**a**) Decision Tree Model for assessing restoration activities following invader control efforts. Decision making nodes represent the assessment of identity, number of and interactions between recovery constraints (*bolded boxes*). At low and medium IAP abundances, control/removal efforts may be sufficient to return a community to its restored state (node 1: No intervention). However, at medium and high invader abundances management actions may not be sufficient to achieve the full recovery of the degraded system due to recovery constraints. A recovery constraint assessment can guide decisions for subsequent restoration actions (nodes 2 & 3). Ecological principles (*listed in parentheses*) can help inform which restoration tools to use. (**b**) Decision Tree model for the control and restoration efforts for *Cynara cardunculus*. Initially management efforts relied on passive recovery (*dashed line*); however after observing the ineffective recovery of native species, the land managers decided to implement efforts to overcome constraints. Evaluation of constraints (supporting citations listed in case study) indicated two potential constraints, and after determining that they likely do not interact to synergistically thwart recovery, constraints were prioritised. *Grey boxes* indicate paths that were not followed in this scenario

(Prach et al. 2001; Lockwood and Samuels 2004). Successional theory and assembly theory would be helpful in guiding restoration efforts with single constraints (Suding and Hobbs 2009). However, if multiple constraints are present, it is important to assess whether these constraints can be addressed independently or need to be addressed in tandem (Suding et al. 2004). If multiple constraints synergistically thwart the recovery of a system, it would be essential to address the constraints in tandem to disrupt any feedbacks that are preventing recovery (Fig. 27.2a; Suding et al. 2004).

Constraints can operate at multiple spatial and temporal scales (Suding and Hobbs 2009), and processes operating at one spatial or temporal scale can interact with processes operating at another scale to create strong internal feedbacks that prevent the recovery of the system (cross-scale interactions; Peters et al. 2007). In a hypothetical example, if an invader disrupted dispersal processes and produced soil legacies via allelopathy, successful restoration efforts would have to address both the soil condition as well as the dispersal constraint. Threshold models address strong internal feedbacks and nonlinear dynamics within ecosystems and would be helpful in guiding restoration efforts with interacting constraints

(Suding and Hobbs 2009). If the constraints do not interact, it would be important to prioritise constraints, and assembly theory could help elucidate which constraints and potential restoration approaches could be addressed based on how the degraded system has deviated from the historical environmental and biotic conditions (Lockwood and Samuels 2004). However, projects that incorporate several restoration actions are often more successful; therefore if resources are available, it would be wise to tackle multiple constraints (augmentative restoration; Bard et al. 2004; Buisson et al. 2008). While these approaches can help a land manager make better a priori decisions about what restoration activities to undertake, this approach is not fool proof and should always incorporate monitoring and re-assessment to ensure that the system is moving in the desired direction.

27.4.1 Case Study 4: Sustainable Control Efforts of Cynara carduncului *(Artichoke Thistle) in Orange County, California*

Cynara carduncului was introduced into southern California in the nineteenth century and has become a problematic invader across local grasslands (Thomsen et al. 1986). It is a perennial species with a deep taproot (about 1.5 m) and large inflorescences (up to 50 per rosette; Marushia and Holt 2006) filled with up to 800 wind dispersed seeds (Kelly 2000). It forms dense species-poor stands (Bowler 2008). Within the Nature Reserve of Orange County, it has invaded over 1,618 of the 14,973 ha of protected open space and its control has dominated the Reserve's budget for invasive species management (McAfee 2008). The primary control method since 1994 has been direct herbicide application with the assumption that native communities would passively recover. However, after 13 years native species did not recover in all treated areas; instead, the abundance of other alien plants increased (Seastedt et al. 2008). In an effort to implement more effective management activities for native grassland recovery, potential constraints were further identified using other published research studies. For example, seed limitation is often a constraint for native grass populations across California (Seabloom et al. 2003; Seabloom 2011). Additionally, Potts et al. (2008) identified that litter quantity and quality changes due to *C. carduncului* invasion, which can negatively impact native recovery (Bartolome and Gemmill 1981; Coleman and Levine 2007). These findings can be integrated into a potential management plan for efficient and sustainable management of treated areas (Fig. 27.2b), where the passive recovery approach would be replaced with one where seed limitation and soil legacies constraints are both prioritised within the reserve.

27.5 Conclusions

Ecosystems globally are undergoing rapid changes due to global change drivers such as CO_2 enrichment, atmospheric nitrogen deposition, climate changes, land use, and biotic invasions (Sala et al. 2000). Among these drivers are invasive species, which are an increasing threat to natural and working landscapes as the globalization of trade and interactions with other global change drivers increase the opportunities for introductions (Levine and D'Antonio 2003; McNeely 2006). A small fraction of those invaders have the potential to trigger large changes in ecosystem functioning as they spread across a landscape (Williamson 1996) and can contribute to the degradation of native communities (McNeely 2001). Protected areas have the unprecedented burden of minimizing these negative invader impacts as they are tasked with the goal of protecting and maintaining the globe's biodiversity.

Here, we emphasised ways to determine whether additional intervention is needed for native recovery following IAP control in PAs. Multiple lines of evidence need to be weighed to best gauge when and where to invest in additional intervention approaches and when to stand back and allow the native system to recover naturally. This decision-making is not clear-cut but can be based on several ecological frameworks describing how communities are assembled and recover over time. Protected areas benefit from a holistic management approach, which addresses IAP detection, sources of invaders, potential external stressors, and management thresholds dictating when management efforts need to be initiated (Zavaleta et al. 2001; Foxcroft and Richardson 2003; Clewell and McDonald 2009). Worldwide efforts to evaluate the effectiveness of management within PAs (Hocking et al. 2000) provide a unique opportunity to assess the link between ecological theories that frame the process of recovery and restoration actions.

Acknowledgements We thank the UC Berkeley Range group, Suding lab members, and Marko Spasojevic for helpful comments on early drafts. We acknowledge support from the NSF Graduate Research Fellowship programme (to L.L.)

References

Bais HP, Vepachedu R, Gilroy S et al (2003) Allelopathy and exotic plant invasion: from molecules and genes to species interactions. Science 301:1377–1380

Bard EC, Sheley RL, Jacobsen JS et al (2004) Using ecological theory to guide the implementation of augmentative restoration. Weed Technol 18:1246–1249

Bartolome JW, Gemmill B (1981) The ecological status of *Stipa pulchra* (Poaceae) in California. Madrono 28:172–184

Bhagwat SA, Breman E, Thekaekara T et al (2012) A Battle lost? Report on two centuries of invasion and management of *Lantana camara* L. in Australia, India and South Africa. Plos One 7:e32407

Bhatt YD, Rawat YS, Singh SP (1994) Changes in ecosystem functioning after replacement of forest by *Lantana* shrubland in Kumaun Himalaya. J Veg Sci 5:67–70
Blanchard R, Holmes PM (2008) Riparian vegetation recovery after invasive alien tree clearance in the Fynbos Biome. S Afr J Bot 74:421–431
Bowler PA (2008) Artichoke thistle as an ecological resource and its utility as a precursor to restoration (California). Ecol Restor 26:7–8
Briske DD, Fuhlendor SD, Smeins EE (2005) State-and-transition models, thresholds, and rangeland health: a synthesis of ecological concepts and perspectives. Rangel Ecol Manage 58:1–10
Buisson E, Anderson S, Holl KD et al (2008) Reintroduction of *Nassella pulchra* to California coastal grasslands: effects of topsoil removal, plant neighbour removal and grazing. Appl Veg Sci 11:195–204
Callaway RM, Ridenour WM (2004) Novel weapons: invasive success and the evolution of increased competitive ability. Front Ecol Environ 2:436–443
Chesson P (2000) Mechanisms of maintenance of species diversity. Ann Rev Ecol Syst 31:343–366
Cione NK, Padgett PE, Allen EB (2002) Restoration of a native shrubland impacted by exotic grasses, frequent fire, and nitrogen deposition in southern California. Restor Ecol 10:376–384
Clark BR, Hartley SE, Suding KN et al (2005) The effect of recycling on plant competitive hierarchies. Am Nat 165:609–622
Cleland EE, Larios L, Suding KN (2013) Strengthening invasion filters to reassemble native plant communities: soil resources and phenological overlap. Restor Ecol 21:390–398. doi:10.1111/j.1526-100X.2012.00896.x
Clements FE (1916) Plant succession, vol 242. Carnegie Institution, Washington, DC
Clewell A, McDonald T (2009) Relevance of natural recovery to ecological restoration. Ecol Restor 27:122–124
Coleman HM, Levine JM (2007) Mechanisms underlying the impacts of exotic annual grasses in a coastal California meadow. Biol Invasion 9:65–71
Connell JH, Slatyer RO (1977) Mechanisms of succession in natural communities and their role in community stability and organization. Am Nat 111:1119–1144
Conser C, Connor EF (2009) Assessing the residual effects of *Carpobrotus edulis* invasion, implications for restoration. Biol Invasion 11:349–358
Corbin JD, D'Antonio CM (2004) Effects of exotic species on soil nitrogen cycling: implications for restoration. Weed Technol 18:1464–1467
Cowles HC (1899) The ecological relations of the vegetation on the sand dunes of Lake Michigan. Bot Gazette 27:95–117
Cox RD, Allen EB (2008) Composition of soil seed banks in southern California coastal sage scrub and adjacent exotic grassland. Plant Ecol 198:37–46
D'Antonio CM, Chambers JC (2006) Using ecological theory to manage or restore ecosystems affected by invasive plant species. In: Falk DA, Palmer MA, Zedler JB (eds) Foundations of restoration ecology. Island Press, Washington, DC, pp 260–279
D'Antonio C, Meyerson LA (2002) Exotic plant species as problems and solutions in ecological restoration: a synthesis. Restor Ecol 10:703–713
D'Antonio CM, Vitousek PM (1992) Biological invasions by exotic grasses, the grass fire cycle, and global change. Ann Rev Ecol Syst 23:63–87
D'Antonio C, Dudley TL, Mack MC (1999) Disturbance and biological invasions: direct effects and feedbacks. In: Walker LR (ed) Ecosystems of disturbed ground. Elsevier, New York, pp 413–452
Davies KW, Svejcar TJ, Bates JD (2009) Interaction of historical and nonhistorical disturbances maintains native plant communities. Ecol Appl 19:1536–1545
del Moral R, Walker LR, Bakker JP (2007) Insights gained from succession for the restoration of landscpae structure and function. In: Walker LR, Walker J, Hobbs RJ (eds) Linking restoration and ecological succession. Springer, New York, pp 19–44

Diaz S, Cabido M, Casanoves F (1998) Plant functional traits and environmental filters at a regional scale. J Veg Sci 9:113–122
Diaz S, Cabido M, Casanoves F (1999) Functional implications of trait-environment linkages in plant communities. In: Weiher E, Keddy PA (eds) Ecological assembly rules. Cambridge University Press, Cambridge
Didham RK, Tylianakis JM, Gemmell NJ et al (2007) Interactive effects of habitat modification and species invasion on native species decline. Trends Ecol Evol 22:489–496
Doren RF, Whiteaker LD (1990) Effects of fire on different size individuals of *Schinus terbinthifolius*. Nat Areas J 10:107–113
Doren RF, Richards JH, Volin JC (2009a) A conceptual ecological model to facilitate understanding the role of invasive species in large-scale ecosystem restoration. Ecol Indic 9:S150–S160
Doren RF, Volin JC, Richards JH (2009b) Invasive exotic plant indicators for ecosystem restoration: an example from the Everglades restoration program. Ecol Indic 9:S29–S36
Ehrenfeld JG (2003) Effects of exotic plant invasions on soil nutrient cycling processes. Ecosystems 6:503–523
Ehrenfeld JG, Ravit B, Elgersma K (2005) Feedback in the plant-soil system. Ann Rev Environ Res 30:75–115
Eppinga MB, Rietkerk M, Dekker SC et al (2006) Accumulation of local pathogens: a new hypothesis to explain exotic plant invasions. Oikos 114:168–176
Fensham RJ, Fairfax RJ, Cannell RJ (1994) The invasions of *Lantana camara* L. in Forty-Mile Scrub National Park, north Queensland. Aust J Ecol 19:297–305
Folke C, Carpenter S, Walker B et al (2004) Regime shifts, resilience, and biodiversity in ecosystem management. Ann Rev Ecol Evol Syst 35:557–581
Fox BJ, Taylor JE, Fox MD et al (1997) Vegetation changes across edges of rainforest remnants. Biol Conserv 82:1–13
Foxcroft LC, Richardson DM (2003) Managing alien plant invasions in the Kruger National Park, South Africa. In: Child LE, Brock JH, Brundu G et al (eds) Plant invasions: ecological threats and management solutions. Backhuys Publishers, Leiden, pp 385–403
French K, Mason TJ, Sullivan N (2011) Recruitment limitation of native species in invaded coastal dune communities. Plant Ecol 212:601–609
Friedel MH (1991) Range condition assessment and the concept of thresholds – a viewpoint. J Range Manage 44:422–426
Funk JL, Cleland EE, Suding KN et al (2008) Restoration through reassembly: plant traits and invasion resistance. Trends Ecol Evol 23:695–703
Gaertner M, Richardson DM, Privett SDJ (2011) Effects of alien plants on ecosystem structure and functioning and implications for restoration: insights from three degraded sites in South African fynbos. Environ Manage 48:57–697
Gaertner M, Holmes PM, Richardson DM (2012) Biological invasions, resilience and restoration. In: van Andel J, Aronson J (eds) Restoration ecology: the new frontier. Wiley-Blackwell, Oxford, pp 265–280
Galatowitsch S, Richardson DM (2005) Riparian scrub recovery after clearing of invasive alien trees in headwater streams of the Western Cape, South Africa. Biol Conserv 122:509–521
Gelbard JL, Belnap J (2003) Roads as conduits for exotic plant invasions in a semiarid landscape. Conserv Biol 17:420–432
Gentle CB, Duggin JA (1997) *Lantana camara* L. invasions in dry rainforest open forest ecotones: the role of disturbances associated with fire and cattle grazing. Aust J Ecol 22:298–306
Gleason HA (1926) The individualistic concept of the plant association. Bull Torr Bot Club 53:7–26
Glenn-Lewin DC, Peet RK, Veblen TT (eds) (1992) Plant succession. Theory and prediction. Chapman & Hall, London
Gooden B, French K, Turner PJ et al (2009) Impact threshold for an alien plant invader, *Lantana camara* L., on native plant communities. Biol Conserv 142:2631–2641

Greer KA (2005) Habitat conservation planning in San Diego County, California: lessons learned after five years of implementation. Environ Pract 6:230–239
Hawkes CV, Wren IF, Herman DJ et al (2005) Plant invasion alters nitrogen cycling by modifying the soil nitrifying community. Ecol Lett 8:976–985
Hobbs RJ, Harris JA (2001) Restoration ecology: repairing the Earth's ecosystems in the new millennium. Restor Ecol 9:239–246
Hobbs RJ, Mooney HA (1995) Spatial and temporal variability in California annual grassland – results from a long-term study. J Veg Sci 6:43–56
Hobbs RJ, Richardson DM (2011) Invasion ecology and restoration ecology: parallel evolution in two fields of endeavour. In: Richardson DM (ed) Fifty years of invasion ecology: the legacy of Charles Elton. Blackwell Publishing, West Sussex, pp 61–69
Hobbs RJ, Jentsch A, Temperton VM (2007) Restoration as a process of assembly and succession mediated by disturbance. In: Walker LR, Walker J, Hobbs RJ (eds) Linking restoration and ecological succession. Springer, New York, pp 150–167
Hocking M, Stolton S, Dudley N (2000) Evaluating effectiveness: a framework for assessing the management of protected areas. Best practice protected area guidelines, vol 6. IUCN, Gland/Cambridge
Holl KD, Aide TM (2011) When and where to actively restore ecosystems? For Ecol Manage 261:1558–1563
Holling CS (1973) Resilience and stability of ecological systems. Ann Rev Ecol Syst 4:1–23
Inderjit, van der Putten WH (2010) Impacts of soil microbial communities on exotic plant invasions. Trends Ecol Evol 25:512–519
James JJ, Smith BS, Vasquez EA et al (2010) Principles for ecologically based invasive plant management. Invasion Plant Sci Manage 3:229–239
Jones HP, Schmitz OJ (2009) Rapid recovery of damaged ecosystems. Plos One 4:e5653
Jones CG, Lawton JH, Shachak M (1994) Organisms as ecosystem engineers. Oikos 69:373–386
Keenleyside KA, Dudley N, Cairns S et al (2012) Ecological restoration for protected areas: principles, guidelines and best practices. IUCN, Gland
Kelly M (2000) *Cynara cardunculus*. In: Bossard CC, Randall JM, Hoshovsky MC (eds) Invasive plants of California's wildlands. University of California Press, Berkeley, pp 139–145
Kettenring KM, Adams CR (2011) Lessons learned from invasive plant control experiments: a systematic review and meta-analysis. J Appl Ecol 48:970–979
Klironomos JN (2002) Feedback with soil biota contributes to plant rarity and invasiveness in communities. Nature 417:67–70
Kulmatiski A (2006) Exotic plants establish persistent communities. Plant Ecol 187:261–275
Le Maitre DC, Versfeld DB, Chapman RA (2000) The impact of invading alien plants on surface water resources in South Africa: a preliminary assessment. Water SA 26:397–408
Le Maitre DC, Gaertner M, Marchante E et al (2011) Impacts of invasive Australian Acacias: implications for management and restoration. Divers Distrib 17:1015–1029
Levine JM, D'Antonio CM (2003) Forecasting biological invasions with increasing international trade. Conserv Bio 17:322–326
Levine JM, Murrell DJ (2003) The community-level consequences of seed dispersal patterns. Ann Rev Ecol Evol Syst 34:549–574
Levine JM, Vilà M, D'Antonio CM et al (2003) Mechanisms underlying the impacts of exotic plant invasions. Proc R Soc Lond Ser B-Biol Sci 270:775–781
Lockwood JL, Samuels CL (2004) Assembly models and the practice of restoration. In: Temperton VM, Hobbs RJ, Nuttle T et al (eds) Assembly rules and restoration ecology: bridging the gap between theory and practice. Island Press, Washington, DC, pp 55–70
Lockwood M, Worboys GL, Kothari A (eds) (2006) Managing protected areas. Earthscan, London
Lowe SJ, Browne M, Boudjelas S et al (2000) 100 of the world's worst invasive alien species from the Global Invasive Species Database. Invasive Species Specialist Group (ISSG) a specialist group of the Species Survival Commission (SSC) of the World Conservation Union (IUCN), Auckland, New Zealand

MacArthur R, Levins R (1967) Limiting similarity convergence and divergence of coexisting species. Am Nat 101:377–385
Macdonald IAW (1984) Is the fynbos biome especially susceptible to invasion by alien plants – a re-analysis of available data. S Afr J Sci 80:369–377
Macdonald IAW (2004) Recent research on alien plant invasions and their management in South Africa: a review of the inaugural research symposium of the working for water programme. S Afr J Sci 100:21–26
Macdonald IAW, Richardson DM (1986) Alien species in terrestrial ecosystems of the fynbos biome. In: Macdonald IAW, Kruger FJ, Ferrar AA (eds) The ecology and management of biological invasions in Southern Africa. Oxford University Press, Cape Town, pp 77–91
Macdonald IAW, Graber DM, Debenedetti S et al (1988) Introduced species in nature reserves in Mediterranean-type climatic regions of the world. Biol Conserv 44:37–66
Mack MC, D'Antonio CM (1998) Impacts of biological invasions on disturbance regimes. Trends Ecol Evol 13:195–198
Mack MC, D'Antonio CM, Ley RE (2001) Alteration of ecosystem nitrogen dynamics by exotic plants: a case study of C-4 grasses in Hawaii. Ecol Appl 11:1323–1335
Malmstrom CM, Hughes CC, Newton LA et al (2005) Virus infection in remnant native bunchgrasses from invaded California grasslands. New Phytol 168:217–230
Mangla S, Inderjit, Callaway RM (2008) Exotic invasive plant accumulates native soil pathogens which inhibit native plants. J Ecol 96:58–67
Marushia RG, Holt JS (2006) The effects of habitat on dispersal patterns of an invasive thistle, *Cynara cardunculus*. Biol Invasion 8:577–593
McAfee L (2008) Nature reserve of orange county annual report. Nature Reserve of Orange County, Irvine
McConkey KR, Prasad S, Corlett RT et al (2012) Seed dispersal in changing landscapes. Biol Conserv 146:1–13
McKinney ML, Lockwood JL (1999) Biotic homogenization: a few winners replacing many losers in the next mass extinction. Trends Ecol Evol 14:450–453
McNeely JA (2001) Invasive species: a costly catastrophe for native biodiversity. Land Use Water Resour Res 1:1–10
McNeely JA (2006) As the world gets smaller, the chances of invasion grow. Euphytica 148:5–15
Milton SJ, Hall AV (1981) Reproductive biology of Australian acacias in the southwestern Cape-Province, South Africa. Trans Roy Soc S Afr 44:465–485
Murcia C (1997) Evaluation of Andean alder as a catalyst for the recovery of tropical cloud forests in Colombia. For Ecol Manage 99:163–170
Padgett PE, Allen EB (1999) Differential responses to nitrogen fertilization in native shrubs and exotic annuals common to Mediterranean coastal sage scrub of California. Plant Ecol 144:93–101
Pearson J, Wells DM, Seller KJ et al (2000) Traffic exposure increases natural (15)N and heavy metal concentrations in mosses. New Phytol 147:317–326
Perry LG, Johnson C, Alford ER et al (2005) Screening of grassland plants for restoration after spotted knapweed invasion. Restor Ecol 13:725–735
Peters DPC, Bestelmeyer BT, Turner MG (2007) Cross-scale interactions and changing pattern-process relationships: consequences for system dynamics. Ecosystems 10:790–796
Pickett STA, Collins SL, Armesto JJ (1987) A hierarchical consideration of causes and mechanisms of succession. Vegetation 69:109–114
Pickett STA, Kolasa J, Armesto JJ et al (1989) The ecological concept of disturbance and its expression at various hierarchical levels. Oikos 54:129–136
Pickett STA, Cadenasso ML, Meiners SJ (2009) Ever since Clements: from succession to vegetation dynamics and understanding to intervention. Appl Veg Sci 12:9–21
Potts DL, Harpole WS, Goulden ML et al (2008) The impact of invasion and subsequent removal of an exotic thistle, *Cynara cardunculus*, on CO2 and H2O vapor exchange in a coastal California grassland. Biol Invasion 10:1073–1084

Prach K, Hobbs RJ (2008) Spontaneous succession versus technical reclamation in the restoration of disturbed sites. Restor Ecol 16:363–366

Prach K, Walker LR (2011) Four opportunities for studies of ecological succession. Trends Ecol Evol 26:119–123

Prach K, Bartha S, Joyce CB et al (2001) The role of spontaneous vegetation succession in ecosystem restoration: a perspective. Appl Veg Sci 4:111–114

Prach K, Marrs R, Pyšek P et al (2007) Manipulation of succession. In: Walker LR, Walker J, Hobbs RJ (eds) Linking restoration and ecological succession. Springer, New York, pp 121–149

Prober SM, Lunt ID, Morgan JW (2009) Rapid internal plant-soil feedbacks lead to alternative stable states in temperate Australian grassy woodlands. In: Hobbs RJ, Suding KN (eds) New models for ecosystem dynamics and restoration. Island Press, Washington, DC, pp 156–168

Sala OE, Chapin FS, Armesto JJ et al (2000) Biodiversity – global biodiversity scenarios for the year 2100. Science 287:1770–1774

Seabloom EW (2011) Spatial and temporal variability in propagule limitation of California native grasses. Oikos 120:291–301

Seabloom EW, Borer ET, Boucher VL et al (2003) Competition, seed limitation, disturbance, and reestablishment of California native annual forbs. Ecol Appl 13:575–592

Seastedt TR, Hobbs RJ, Suding KN (2008) Management of novel ecosystems: are novel approaches required? Front Ecol Environ 6:547–553

Sheley RL, Mangold JM, Anderson JL (2006) Potential for successional theory to guide restoration of invasive-plant-dominated rangeland. Ecol Monogr 76:365–379

Sheley R, James J, Smith B et al (2010) Applying ecologically based invasive-plant management. Rangel Ecol Manage 63:605–613

Sirulnik AG, Allen EB, Meixner T et al (2007) Impacts of anthropogenic N additions on nitrogen mineralization from plant litter in exotic annual grasslands. Soil Biol Biochem 39:24–32

Sperry LJ, Belnap J, Evans RD (2006) *Bromus tectorum* invasion alters nitrogen dynamics in an undisturbed arid grassland ecosystem. Ecology 87:603–615

Stock D (2004) The dynamics of *Lantana camara* (L.) invasion of subtropical rainforest in Southeastern Australia. Griffith University, Gold Coast Campus, Brisbane

Stock D (2005) Management of *Lantana* in eastern Australia subtropical rainforests. In: Worboys GL, Lockwood M, De Lacy T (eds) Protected area management: principles and practices, 2nd edn. Oxford University Press, Melbourne

Suding KN (2011) Toward an era of restoration in ecology: successes, failures, and opportunities ahead. Ann Rev Ecol Evol Syst 42:465–487

Suding KN, Hobbs RJ (2009) Threshold models in restoration and conservation: a developing framework. Trends Ecol Evol 24:271–279

Suding KN, Gross KL, Houseman GR (2004) Alternative states and positive feedbacks in restoration ecology. Trends Ecol Evol 19:46–53

Swarbick JT (1986) History of the lantanas in Australia and origins of the weedy biotypes. Plant Protect Quart 1:115–121

Temperton VM, Hobbs RJ, Nuttle T et al (eds) (2004) Assembly rules and restoration ecology: bridging the gap between theory and practice. The science and practice of ecological restoration. Island Press, Washington, DC

Thomsen C, Barbe G, Williams W et al (1986) "Escaped" artichokes are troublesome pests. Calif Agric 40:7–9

Tilman D (1988) Monographs in population biology No 26. Plant strategies and the dynamics and structure of plant communities. Princeton University Press, Princeton

Tilman D (1990) Constraints and tradeoffs – toward a predictive theory of competition and succession. Oikos 58:3–15

Traveset A, Richardson DM (2006) Biological invasions as disruptors of plant reproductive mutualisms. Trends Ecol Evol 21:208–216

Turpie JK, Marais C, Blignaut JN (2008) The working for water programme: evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. Ecol Econ 65:788–798

UNEP-SCBD (2001) Global biodiversity outlook. Paper presented at the UNEP Secretariat of the convention on biological diversity, Montreal, Canada

van der Putten WH, Klironomos JN, Wardle DA (2007) Microbial ecology of biological invasions. Isme J 1:28–37

Vitousek PM, Walker LR (1989) Biological invasion by *Myrica-Faya* in Hawaii – plant demography, nitrogen-fixation, ecosystem effects. Ecol Monogr 59:247–265

Vitousek PM, Dantonio CM, Loope LL et al (1997) Introduced species: a significant component of human-caused global change. N Z J Ecol 21:1–16

Vivrette NJ, Muller CH (1977) Mechanism of invasion and dominance of coastal grassland by *Mesembryanthemum crystallinum*. Ecol Monogr 47:301–318

Weiher E, Keddy PA (1995) Assembly rules, null models, and trait dispersion – new questions from old patterns. Oikos 74:159–164

Whisenant SG (1990) Changing fire frequencies on Idaho's Snake River Plains: ecological and management implications. In: Proceedings-symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management, Las Vegas, pp 4–10

Whisenant S (1999) Repairing damaged wildlands: a process-oriented, landscape scale approach. Cambridge University Press, Port Chester

White PS, Jentsch A (2004) Disturbance, succession, and community assembly in terrestrial plant communities. In: Temperton VM, Hobbs RJ, Nuttle T et al (eds) Assembly rules and restoration ecology: bridging the gap between theory and practice. Island Press, Washington, DC, pp 342–366

Williams PA, Karl BJ (2002) Birds and small mammals in kanuka (*Kunzea ericoides*) and gorse (*Ulex europaeus*) scrub and the resulting seed rain and seedling dynamics. N Z J Ecol 26:31–41

Williamson M (1996) Biological invasions. Population and community biology. Chapman & Hall, Cornwall

Wojterski TW (1990) Degradation stages of the oak forests in the area of Algiers. Vegetation 87:135–143

Yelenik SG, Stock WD, Richardson DM (2004) Ecosystem level impacts of invasive *Acacia saligna* in the South African fynbos. Restor Ecol 12:44–51

Young TP, Chase JM, Huddleston RT (2001) Community succession and assembly: comparing, contrasting and combining paradigms in the context of ecological restoration. Ecol Restor 19:5–18

Young TP, Petersen DA, Clary JJ (2005) The ecology of restoration: historical links, emerging issues and unexplored realms. Ecol Lett 8:662–673

Zavaleta E (2000) The economic value of controlling an invasive shrub. Ambio 29:462–467

Zavaleta ES, Hobbs RJ, Mooney HA (2001) Viewing invasive species removal in a whole-ecosystem context. Trends Ecol Evol 16:454–459