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Restoration of forest resilience: An achievable goal?

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Abstract Although the concept of resilience is increasingly being incorporated into environmental policy and linked to ecological restoration goals, there is considerable uncertainty regarding how resilience should be defined and measured in practice. Here we briefly review some of the definitions of resilience that have been proposed, including those referred to as "ecological" and "engineering" resilience. We also examine evidence for the existence of multiple stable states in forest ecosystems, on which concepts of ecological resilience are based. As evidence for multiple stable states is limited, we suggest that ecological resilience may often have limited value as a goal for forest restoration. We illustrate how engineering resilience can potentially be measured by estimating the rate of forest recovery following disturbance, through analysis of recovery trajectories using meta-analysis and ecological modelling approaches. We also highlight the potential value of resistance as a restoration goal, which can similarly be estimated using such approaches. Based on application of these concepts, we suggest how guidance for restoration practitioners could potentially be developed, to support the practical achievement of both resilience and resistance during forest restoration.

Keywords Ecological resilience · Ecosystem resilience · Adaptive capacity · Biodiversity · Ecological restoration · Multiple stable states · Woodland

Introduction

The term 'resilience' is increasingly being incorporated into environmental policy. For example, Australia's national biodiversity conservation strategy identifies building ecosystem resilience as one of three main priorities for action (Natural Resource

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Management Ministerial Council 2010). Similarly in the USA, the Environmental Protection Agency's guiding principles for climate change adaptation suggest that "adaptation should, where relevant, take into account strategies to increase ecosystem resilience" (EPA 2012). US forest service policy recognises that management for increased genetic and species diversity should increase resilience and adaptive capacity (USFS 2010). At the international scale, resilience is referred to within global policy initiatives such as the convention on biological diversity (CBD) (Thompson et al. 2009) and the Intergovernmental Panel on Climate Change (IPCC 2014). References to resilience are also being included within policy initiatives relating explicitly to ecological restoration. As illustration, in Nagoya in 2010, Parties to the CBD committed to take action so that by 2020 'ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 % of degraded ecosystems' (CBD COP 10 Decision X/2). In the UK, recently developed environmental policy aims to create 'a resilient ecological network', through ecological restoration (HM Government 2011).

If policy goals relating to resilient ecosystems and landscapes are to be achieved, for example through ecological restoration, there needs to be a common understanding of how resilience can be defined, measured and monitored in practice. While resilience has long been the subject of ecological research, it continues to be the focus of significant debate, both in terms of its definition and operationalisation (Grimm and Wissel 1997; Grimm and Calabrese 2011). Given the current policy context, forest managers and other restoration practitioners will increasingly be tasked with delivering resilience in their management plans, while receiving little practical guidance regarding how this might actually be achieved in practice.

The objective of this paper is to provide an overview of resilience in the context of the restoration of forest ecosystems, to support provision of this guidance and the achievement of policy goals. First, different definitions of the concept are considered, with the aim of identifying a definition suitable for supporting ecological restoration actions. Second, this paper examines the existence of multiple stable states in forest ecosystems, with reference to the definition of resilience. Methods of measuring and forecasting resilience are then explored, with specific reference to the trajectories of forest recovery following disturbance, and analysis of disturbance gradients. Finally, consideration is given to how restoration of resilient forest ecosystems might be achieved in practice.

Defining resilience and associated concepts

Broadly speaking, resilience is a measure of the persistence of an ecosystem and its ability to absorb disturbance (Holling 1973), or the ability of an ecosystem to maintain its functions when faced with novel disturbance (Webb 2007). However, the precise definition of the term 'resilience' in an ecological context has been the focus of substantial debate. Many different definitions have been proposed (Table 1). Grimm and Wissel (1997), for example, report 17 different definitions of resilience in the scientific literature, with many additional definitions proposed for closely related concepts such as stability. Similarly Brand and Jax (2007) list ten different definitions of resilience, grouped into three main categories. This problem of semantic uncertainty is associated with a number of concepts in ecological science, and hinders the operationalisation and practical use of such concepts (Peters 1991). Clearly, any attempt to achieve resilience through ecological restoration would first need to specify an appropriate definition of the concept.

Table 1 Selected definitions of resilience that have been proposed in the ecological literature

Definition	Source
The magnitude of disturbance that can be tolerated before a system moves into a different region of state space and a different set of controls	Carpenter et al. (2001)
The ability of the system to maintain its identity in the face of internal change and external shocks and disturbances	Cumming et al. (2005)
The length of time taken to return to the pre-disturbance state	Donohue et al. (2013)
The capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure and feedbacks, and therefore identity, that is, the capacity to change in order to maintain the same identity	Folke et al. (2010a)
Returning to the reference state (or dynamic) after a temporary disturbance	Grimm and Wissel (1997)
Resilience determines the persistence of relationships within a system and is a measure of the ability of these systems to absorb changes of state variables, driving variables, and parameters, and still persist	Holling (1973): 17)
How fast the variables return towards their equilibrium following a perturbation	Pimm (1984); see also Pimm (1991)
The ability of the system to return to the original state after a disturbance	Scheffer et al. (2002)
The capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks	Walker et al. (2004)
The capacity of a system to experience shocks while retaining essentially the same function, structure, feedbacks, and therefore identity	Walker et al. (2006)

Two principal meanings of resilience have been identified in the literature (Brand 2009; Brand and Jax 2007). The first is defined as the time required for a system to return to an equilibrium point following a disturbance event (Pimm 1984; Pimm 1991). This has been referred to as 'engineering resilience' (Holling 1996). The second relates to a situation far from any equilibrium state, and is defined as the amount of disturbance that a system can absorb before changing to another stable state (Brand and Jax 2007). This has been referred to as 'ecological resilience' (Gunderson 2000) or as 'ecosystem resilience' (Holling and Gunderson 2002). The key difference between these two meanings relates to whether ecosystems display a single equilibrium state, as in the case of 'engineering resilience', or multiple stable states, as in the case of 'ecological resilience'. In a system with multiple states, disturbance may result in the system crossing a threshold from one state (or stability domain) to another, which is qualitatively different from returning to the original state (Folke et al. 2010a). This clearly has significant implications in a restoration context, and therefore the issue of whether forest ecosystems exist in multiple stable states is considered further below.

Much of the current research on ecological resilience has been informed by consideration of theory relating to complex adaptive systems (CAS). CAS are composed of agents (such as organisms) that interact locally in time and space based on information they use to respond to their environments. Macroscopic system properties such as resilience emerge from such interactions, and may feed back to influence the subsequent development of those interactions (Levin 1998). Rather than being characterised by a single equilibrium, many CAS have multiple attractors, implying that disturbance could result in the system crossing a threshold and shifting from one state to another (Folke et al. 2010a). Ecological resilience concepts therefore focus on behaviour of a system near the boundary of a particular state, which represents an unstable equilibrium (Folke 2006). Resilience of a system can be evaluated in terms of the amount of disturbance a given system can experience and still remain within the same configuration of states, rather than within a single state (Turner et al. 2003a). Although the concept of alternative stable states represents a simplification of the complexity that occurs in ecosystems (Folke et al. 2010a), rapid transitions or "regime shifts" between different states have been documented in a number of different ecosystem types (Carpenter 2003; Scheffer et al. 2001). Such regime shifts are thought to occur when the controlling variables in a system (including feedbacks) result in a qualitatively different set of system structures and dynamics (Walker et al. 2004).

CAS approaches have also informed the development of resilience definitions that focus on the capacity of a system to absorb disturbance and re-organize while undergoing change (Walker et al. 2004; Table 1). In this context, resilience encompasses a "buffering capacity" that allows persistence during disturbance, but also the capacity for reconfiguration of a system and emergence of new trajectories (Folke 2006). In this sense, it has been suggested that "adaptive capacity" can be considered as a property of resilience, and refers to the degree to which a system can build the capacity to learn and adapt to disturbance (Carpenter et al. 2001; Gunderson 2000). Adaptive capacity has attracted particular attention in the context of research on social-ecological systems (Folke 2006; Smit and Wandel 2006).

CAS approaches often employ a "ball and cup" heuristic, which can be used to help visualise resilience. In this conceptual model, different states or stability domains are represented by the cups or valleys in a diagram (Fig. 1). The position of the ball, representing the system state, can potentially move from one valley to another, but comes to rest within an individual valley (or "basin of attraction") until the system is perturbed in some way. In this type of diagram, resilience is related to the characteristics of the valley. Engineering resilience can be visualised as the steepness of the valley sides, which affects the return time of the ball to the lowest point in the valley following a perturbation. Ecological resilience is based on the idea that multiple valleys exist. Resilience in this context is related to the width of the valley, which affects the size of the perturbation required to move the ball out of one valley and into another (Beisner et al. 2003; Gunderson 2000). In addition, the landscape of valleys can change over time; repeated perturbations might reduce the steepness of the slopes, enabling a smaller disturbance to move the ball between valleys (Beisner et al. 2003). As a result, the system would shift more readily between states. This is considered to be analogous to the loss of resilience in ecological systems (Gunderson and Walters 2002; Scheffer et al. 2009, 2012), which ecological restoration might potentially seek to reverse. Adaptive capacity can be said to refer to the ability of the system to remain in a particular stability domain, as the shape of the domain changes over time (Gunderson 2000).

These contrasting perceptions of resilience have arguably influenced different approaches to environmental management. Engineering resilience encourages a focus on maintaining the constancy of the system, and resisting disturbance and change (Folke 2006). Ecological resilience concepts have encouraged managers to consider maintaining the variation that occurs naturally, rather than focusing on stability as a management goal (Carpenter et al. 2001). In a restoration context, the choice of restoration definition would therefore influence whether management was designed to deliver a single endpoint or target of restoration, or multiple alternative outcomes.

It is also important to consider the related concept of resistance. This has also been defined variously by different authors; Grimm and Wissel (1997) identified nine different definitions in their literature search. Walker et al. (2004) consider resistance as an aspect of

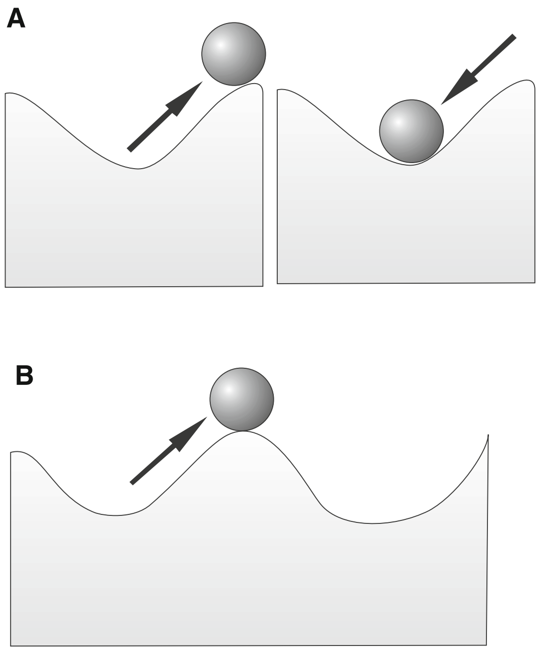

Fig. 1 Illustration of the two main meanings of resilience, using the 'ball and cup' heuristic. **a** Engineering resilience. A perturbation moves the ball up the slope of the cup; *resilience* is indicated by the time taken to return to the stable state at the bottom of the cup. **b** Ecological resilience. Perturbation can move the ball from one stability domain, or cup, to another. *Resilience* is indicated by the amount of disturbance that a system can absorb without changing from one stability regime, or stable state, to another; this is indicated by the width of the cup. Redrawn and adapted from Gunderson and Walters (2002)

resilience, referring to the ease or difficulty of changing the system. For Carpenter et al. (2001), resistance is the amount of external pressure needed to bring about a given amount of disturbance in a system. Pimm (1984) defines resistance as the degree to which a variable is changed, following a perturbation; similarly, Grimm and Wissel (1997) summarize the property of resistance as a system staying essentially unchanged despite the presence of disturbances.

Do forest ecosystems display multiple stable states?

The concept of ecological resilience depends on the presence of multiple stable states within ecological systems. What is the evidence that such multiple states occur in forest ecosystems? Over the last 30 years, it has become clear that forest dynamics are characterised by greater variability and unpredictability than were considered in early models of forest succession, supported by empirical evidence indicating multiple successional pathways within forests (Abrams et al. 1985). For example, recent field research and modelling of two North American forests has indicated that successional dynamics are highly dependent on initial conditions, namely the state of the system immediately following a disturbance event (Haeussler et al. 2013). Simulations revealed an almost infinite variety of recovery trajectories in the early stages of succession, depending on the abundance, spatial distribution and size structure of both juvenile and adult trees following disturbance. However, whether such variation in successional pathway would lead to multiple stable states depends on the existence of multiple system attractors. In the research on temperate forests described by Haeussler et al. (2013), the identities of late-successional canopy dominants were found to be far more predictable than those of earlier stages of succession, suggesting a single attractor rather than multiple attractors. Similarly, multiple successional trajectories have been documented in boreal forests, leading to many alternative plant communities potentially prevailing on an individual site. However over timescales of multiple centuries, a single forest type that appears to be in equilibrium with the prevailing site conditions and climate can often be identified (Burton 2013).

A number of studies have been interpreted as providing evidence of multiple stable states in forest ecosystems (Folke et al. 2004; Folke et al. 2010b; Table 2). Many of these studies have focused on regime shifts, where a system has undergone a transition from one state to another, either suddenly or more gradually. Such shifts can be interpreted as indicating a lack of ecological resilience (Folke et al. 2004). Examples include the impact of spruce budworm on the boreal forests of North America, which intermittently destroys large areas of conifer forests dominated by spruce and fir. Following a defoliation event, the regenerating forest is often dominated by aspen and

Table 2 Proposed examples of multiple stable states in forest ecosystems

Ecosystem type	Alternate state 1	Alternate state 2	References
Boreal forests	Tundra	Boreal forest	Bonan et al. (1992); Higgins et al. (2002)
	Spruce–fir dominance	Aspen–birch dominance	Holling (1978); Paine et al. (1998)
Temperate forests	Pine dominance	Hardwood dominance	Peterson (2002)
	Hardwood–hemlock	Aspen–birch	Frelich and Reich (1999)
	Birch–spruce succession	Pine dominance	Danell et al. (2003)
	Old growth mountain ash forest	Regrowth mountain ash forest	Lindenmayer et al. (2011)
	Sitka spruce	Alder-dominated deciduous forest	Haeussler et al. (2013)
	Oak dominance	Maple dominance	Nowacki and Abrams (2008)
	Beech dominance	Shrub dominance	Busby and Canham (2011)
	Dominance of species palatable to deer	Dominance of species unpalatable to deer	Coomes et al. (2003)
Tropical forests and savanna	Rainforest	Grassland	Trenbath et al. (1989)
	Rainforest	Sclerophyll vegetation	Warman and Moles (2009)
	Woodland	Grassland	Dublin et al. (1990)
	Lowland rainforest	Lowland rainforest, different composition	Vandermeer et al. (2004)
	Grass dominated	Shrub dominated	Anderies et al. (2002); Brown et al. (1997); Valone et al. (2002); Westoby et al. (1989)
	Forest	Savanna	Da Silveira Lobo Sternberg (2001); Hirota et al. (2011); Staver et al. (2011a, 2011b)

From Folke et al. (2010a), Messier et al. (2013), Petraitis (2013) and Resilience Alliance and Santa Fe Institute (2004) birch, although selective browsing by moose can shift the forest back to a conifer-dominated state (Folke et al. 2010a; Holling 1978). Many other examples refer to grasslands or rangelands, which in semi-arid areas can undergo a rapid transition from grass- to shrub-dominated plant communities (Gunderson 2000; Westoby et al. 1989; Table 2). Similarly, savanna and forest have repeatedly been identified as alternative states over large areas, which at least in some cases are only differentiated by the fire regime (Hirota et al. 2011; Staver et al. 2011a, b).

While alternative stable states have repeatedly been described, most available evidence is based on inference from field observations or other indirect methods. Although experimental evidence is widely recognised to comprise the strongest test for multiple stable states (Petraitis 2013), few experimental manipulations have been performed in this context. Schröder et al. (2005) reviewed 35 experiments published between 1980 and 2004, and concluded that 13 demonstrated the existence of alternative stable states whereas 8 did not; 14 did not fulfil the requirements of a conclusive test. From this sample, only one study provided evidence from a field experiment involving woody plants (Valone et al. 2002). The lack of experimental evidence partly reflects the difficulty of conducting an appropriate experiment that could convincingly demonstrate multiple stable states. Peterson (1984) identified four requirements: (1) the environment must not differ between the two putative states, (2) the site should be shown to have the potential to be occupied by two or more distinct communities, (3) the communities should be self-replicating, and (4) the disturbance should be pulse perturbations that mimic a natural event in spatial extent, duration and its effect on species in the system (Petraitis 2013). In practice, it is very difficult to meet all of these requirements.

Although the underlying theory is well established (Beisner et al. 2003; May 1974; Scheffer et al. 2001; Scheffer and Carpenter 2003; Schröder et al. 2005), the existence of multiple stable states in natural ecosystems remains the subject of intense debate, reflecting the lack of robust experimental evidence (Petraitis 2013). Much of the available evidence is based on models, which are often difficult to confirm or disprove; or on description of temporal and spatial patterns, from which it is difficult to reliably infer underlying processes and causality (Petraitis 2013). In particular, it is difficult to demonstrate that the environment is identical at the different times or places that are being compared. Further, it is challenging to show that the states identified are genuinely stable. These limitations apply to virtually all of the examples listed on Table 2. This implies that empirical evidence for multiple stable states in forest ecosystems is largely anecdotal, and that the underlying theory is best viewed as a metaphor; examples such as those listed on Table 2 might be better considered as "persistent alternative states" (Petraitis 2013). Proposals that alternative stable states are more likely in systems controlled by environmental adversity, such as drylands, savanna or tundra (Brand 2009), effectively remain untested. Whether forest ecosystems genuinely display multiple stable states must therefore be considered uncertain, a point that undermines the value of ecological resilience as an operational concept.

Measuring resilience

In order for resilience to be applied in practice, appropriate methods for measuring it need to be developed. A number of different measurement approaches have been suggested by various authors. For example in relation to ecological resilience, Holling (1973) proposed two resilience measures, namely the overall area of the domain of attraction and the height of the lowest point of the basin of attraction above equilibrium. In the context of social-ecological systems, Carpenter et al. (2005) suggested that ecological resilience cannot be measured directly, but must be estimated by means of resilience surrogates, namely indirect proxies that are derived from theory. Methods of identifying such surrogates include stakeholder assessments, model explorations, historical profiling and case study comparison (Carpenter et al. 2005), which can potentially be applied to forested systems (Newton 2011). Brand and Jax (2007) consider situations where it is possible to identify the key controlling variables. In such cases, the 'slow variable' that controls the position of the ecosystem within state space can be used as a surrogate for resilience, based on the distance of the current value of the slow variable to the ecological threshold (Brand and Jax 2007).

One of the problems with the concept of ecological resilience is that it is difficult to quantify or to formalise mathematically (Grimm and Calabrese 2011). As a result, it is unclear how to directly measure ecological resilience or to identify the underlying mechanisms. Consequently, it can be argued that it is perhaps best viewed as an analogy or metaphor rather than as an operational ecological construct (Webb 2007). In contrast, engineering resilience is more amenable to analysis. For example, Grimm and Calabrese (2011) indicate that for models formulated as differential equations, a mathematical protocol exists to calculate whether or not a system returns to equilibrium and how rapid the recovery will be. Specifically, the rate of return to a reference state (or dynamics) can be calculated using linear stability analysis. Martin et al. (2011) note that engineering resilience can be assessed using simulations as the inverse of the return time (May 1974), or the time needed following disturbance to return to some particular configuration, such as its original state. Such analyses can be performed with individual-based models and cellular automata, as well as differential equation models (e.g. Pimm and Lawton 1977).

Recovery trajectories and resilience

As noted above, the time required for an ecosystem to recover from disturbance can be used as a measure of engineering resilience (Pimm 1984). Understanding the process of such recovery lies at the heart of ecological restoration as a scientific discipline. Ecosystem recovery during ecological restoration has been conceptualized as a nonlinear trajectory, which describes the change in indicators of restoration progress over time (Clewell and Aronson 2007). Earlier research tended to suggest that recovery trajectories typically display a smooth increase over time, until the pre-disturbance or reference state is reached (e.g. Bradshaw 1984). More recent research has indicated that recovery trajectories are often complex and do not necessarily follow simple, monotonic recovery pathways that return the system to a pre-disturbance state (Bullock et al. 2011; Matthews 2014; Suding 2011).

Analysis of the factors influencing the shape of recovery trajectories has received surprisingly little attention from researchers, despite its importance for understanding resilience. Potentially, general patterns can be identified using meta-analysis, which has particular value for ecological restoration as it enables results to be integrated from the many individual case studies that have been conducted (Brudvig 2011). This is illustrated by the meta-analysis conducted by Rey-Benayas et al. (2009), which showed that restoration of a range of different ecosystem types was generally effective in increasing biodiversity and provision of ecosystem services, although values were lower than of reference sites. More recently, Martin et al. (2013) presented a meta-analysis of results obtained from more than 600 secondary tropical forest sites with nearby undisturbed reference forests, with the aim of identifying recovery trajectories. Results indicated a curvilinear increase in above-ground biomass, which approached equivalence to reference values within 80 years since last disturbance. Although below-ground biomass took longer to recover, the recovery trajectory displayed a similar shape. In contrast, soil carbon content showed little relationship with time since disturbance. The results presented by Martin et al. (2013) highlight how estimates of engineering resilience can potentially be derived from results of meta-analysis; carbon pools and measures of biodiversity showed contrasting recovery rates under passive restoration.

One of the limitations of many meta-analyses is that they tend to focus on overall average trajectories, which masks the potentially important variation among sites (Matthews 2014). Understanding the causes of this variation could provide valuable insights into the mechanisms underlying resilience. Potentially, this could be achieved by examining the trajectories shared by subgroups of restoration sites, using approaches such as group-based trajectory modelling (Matthews 2014).

Other approaches to ecological modelling can also be used to study recovery trajectories. As noted by Brudvig (2011), one of the goals of restoration ecology should be to develop a predictive science at the landscape scale; potentially, spatially explicit models of forest dynamics could be of value in this context. We have explored the recovery of forest landscapes under different disturbance regimes using LANDIS II, a spatially explicit model that uses an object-oriented approach operating on raster maps. Each cell contains species, environment, disturbance, and harvesting information, and tree species are simulated as the presence or absence of species age cohorts in each cell at each timestep, according to ecological processes including succession, disturbance and seed dispersal (Mladenoff 2004; Scheller et al. 2007). Although originally developed in the USA, LANDIS II has since been used to explore forest landscape dynamics in many parts of the world (Scheller et al. 2007).

In our research, we have applied LANDIS II to study the impact of human disturbance on dynamics of forest landscapes in Chile, Mexico, Kyrgyzstan and the UK (Cantarello et al. 2011, 2014; Newton and Tejedor 2011; Newton et al. 2011, 2013; Table 3). Research has focused on protected areas of high biodiversity value that are subjected to multiple forms of anthropogenic disturbance, which is chronic and is distributed widely throughout the study landscapes. In each case, the model was parameterised and calibrated using empirical data obtained from field surveys and the scientific literature, then used to develop scenarios of different disturbance regimes relevant to each individual location, including the effects of fire, browsing, tree cutting and invasive species. Scenarios of ‘no disturbance’ were included (Table 3), representing cessation of the currently prevailing anthropogenic disturbance regime.

Model projections indicated a similar trajectory in recovery of forest cover in all study areas, with the exception of Tablon, Chiapas, where forest cover was relatively high at the outset. In each case, forest cover increased rapidly over the first timestep, reaching a plateau value, with relatively little change thereafter (Fig. 2). These trajectories are therefore similar in shape to the curvilinear responses identified by Martin et al. (2013) in tropical rainforest, using meta-analysis. In the LANDIS II projections, the type of disturbance regime had relatively little impact on the shape of recovery trajectories, although as expected, rate of recovery was highest under a regime of no disturbance after the initiation of the scenarios. In general, increasing intensity of disturbance reduced the final

Table 3 Summary of study areas used in landscape modelling research

Name	Forest type	Disturbance regimes employed in research	References
Central Veracruz, Mexico	Tropical dry	1. No disturbance	Cantarello et al. (2011)
		2. Grazing	
		3. Small infrequent fires	
		4. Large frequent fires	
		5. Large frequent fires + grazing	
El Tablon, Chiapas, Mexico	Tropical dry	1. No disturbance	Cantarello et al. (2011)
		2. Small infrequent fires	
		3. Large frequent fires	
		4. Grazing	
		5. Small infrequent fires + grazing	
		6. Large frequent fires + grazing	
Quilpué, Chile	Mediterranean dry	1. No disturbance	Newton et al. (2011)
		2. Invasive species (Acacia dealbata)	
		3. Invasive species + fire	
		4. Invasive species + fire + browsing	
		5. Invasive species + fire + cutting	
		6. Invasive species + cutting	
		7. nvasive species + fire + browsing + cutting	
Sary-Chelek, Kyrgyzstan	Temperate montane	1. No disturbance	Cantarello et al. (2014)
		2. Grazing	
		3. Wood cutting	
		4. Grazing + wood cutting	
New Forest, UK	Temperate lowland	1. No disturbance	Newton et al. (2013)
		2. Browsing + fire but with protection from herbivory	
		3. Browsing	
		4. Fire	
		5. Fire + browsing	

The disturbance regimes have been ranked in terms of their overall intensity, with respect to biomass removal

extent of forest cover, but this was more pronounced in some case study areas (e.g. Central Veracruz) than others. These trajectories can potentially be used to estimate engineering resilience. For example, under the no disturbance scenarios, Sary Chelek was projected to reach 100 % forest cover within 50 years, whereas plateau values were reached after approximately 40 and 50 years in Quilpué and Central Veracruz respectively.

Recovery trajectories can be at least partly understood in terms of the process of succession, which can often result in an accumulation of species over time that tends to saturate as a stable equilibrium is approached (Matthews et al. 2009; Suding 2011). The curvilinear recovery trajectory documented here may therefore be explicable in terms of successional processes. However, results also highlighted pronounced variation in the shape of restoration

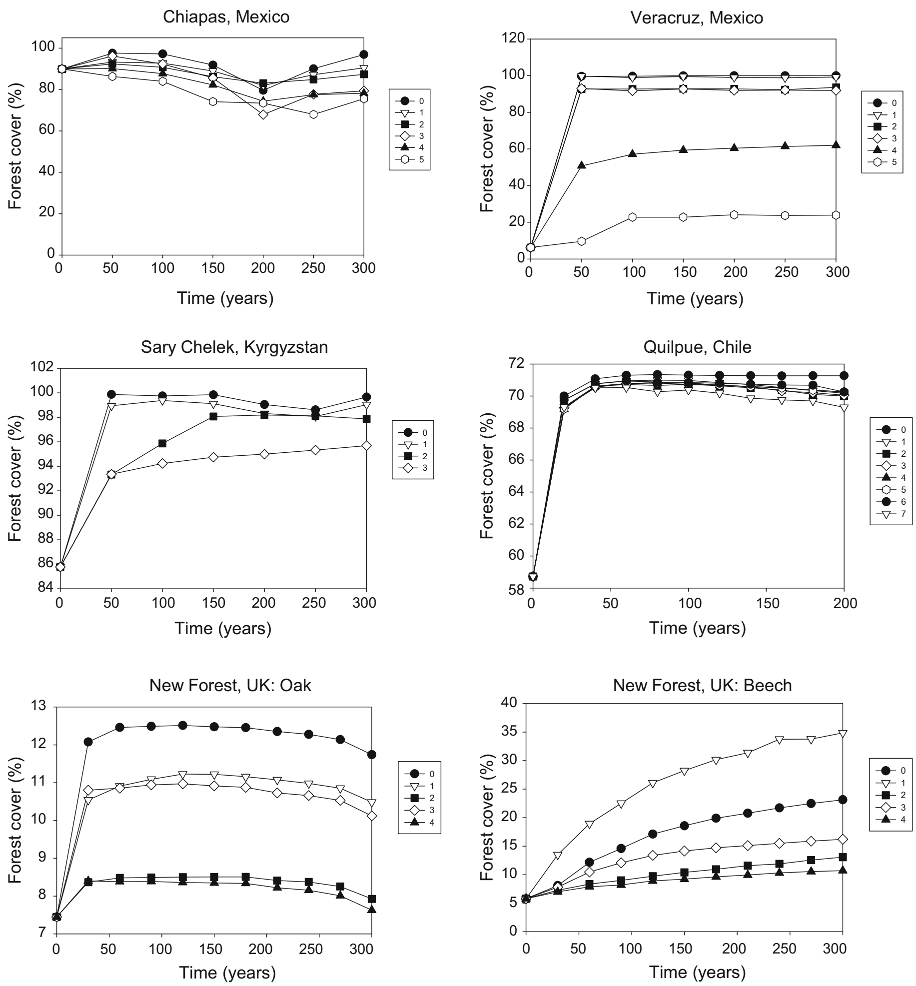

Fig. 2 Dynamics of the extent of forest cover over time under different disturbance regimes, projected for five case study landscapes undergoing ecological restoration, using the LANDIS II model (see text). For details of study areas, and the disturbance regimes illustrated by *different symbols*, see Table 3. *Filled circles* represent scenarios with no anthropogenic disturbance. Note that for the New Forest, UK, the spatial extent of two canopy dominant tree species (oak and beech) is illustrated, rather than the entire forest community. Model outputs derived from studies described by Cantarello et al. (2011, 2014) and Newton et al. (2011, 2013)

trajectories. This may reflect variation in the course of succession, which does not always follow a simple or predictable pattern (Suding 2011). Factors that influence successional trajectories include the frequency and spatial extent of disturbances, interactions between disturbances, spatial heterogeneity, dispersal ability, local environmental conditions, site history, substrate, climate, competitive interactions, chance, and the state of a system immediately following disturbance (Brudvig 2011; Haeussler et al. 2013; Matthews 2014; O'Neill 1998; Suding 2011). Variation in such factors together with the complexity of the system can produce a wide variety of different potential successional pathways (Bullock et al. 2011), and as a result it has been suggested that recovery trajectories can be very difficult or impossible to predict (Folke 2006; Matthews et al. 2009). However, as demonstrated here, both meta-analysis and modelling can be used to identify common patterns among recovery trajectories, which can potentially be used to infer general mechanisms of the process of ecosystem recovery (Grimm and Calabrese 2011). A further important source of information can be obtained through long-term monitoring of post-disturbance recovery (Chai et al. 2012; Suding and Gross 2006). Integration of these different approaches, and their application in a variety of different contexts, could greatly strengthen the understanding of the process of ecological recovery, and consequently resilience.

Disturbance gradients and resistance

Concepts of both resistance and adaptive capacity were considered by Scheffer et al. (2012) in the context of critical thresholds in CAS. Here, a system with adaptive capacity was described as adjusting gradually to change, along a gradient of increasing disturbance (or stress). This was contrasted with a system initially displaying resistance to change, then undergoing a critical transition leading to sudden collapse (Fig. 3). This can occur where a minor perturbation can cause a self-propagating shift to a different system state through a process of positive feedback (Scheffer et al. 2012). This suggested response implies that loss of resistance is linked to sudden shifts in ecosystem states, a view that is widespread in the literature. For example, Folke et al. (2004) suggest that rapid regime shifts may occur more easily if resilience has been reduced as a consequence of human actions. However, according to Petraitis (2013), the transition to an alternative state could just as plausibly take a very long time, and the rapidity of change provides little insight into the existence of multiple stable states.

Analysis of the state of a system along a gradient of increasing disturbance, as illustrated by Scheffer et al. (2012), could provide an indication of resistance. Robust testing of this relationship would require appropriate experiments to be conducted (Petraitis 2013), but potentially it can also be explored through modelling approaches such as those described above. To provide a preliminary example, selected model outputs from applying

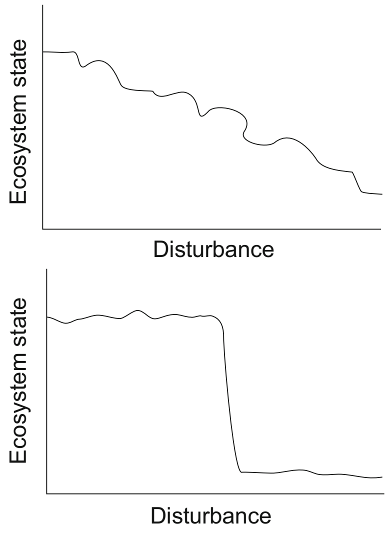

Fig. 3 Two contrasting responses to increasing disturbance. **a** Gradual change in ecosystem state, reflecting adaptive capacity. **b** Initial lack of change in *ecosystem state*, indicating resistance, followed by a rapid decline as an ecological threshold or critical transition is crossed. Adapted and redrawn from Scheffer et al. (2012)

LANDIS II to the five study areas (Table 3) were replotted to illustrate forest landscape responses to gradients of increasing disturbance, analogous to the relationships postulated by Scheffer et al. (2012).

Results indicated a general decline in forest cover with increasing disturbance, although this was more pronounced in some study areas than in others (Fig. 4). For example, Quilpué demonstrated little variation in forest cover with disturbance intensity, suggesting a relatively high degree of resistance. Suggestions of a sudden transition or threshold in forest cover was only evident for Central Veracruz, where forest cover was found to decline rapidly at the highest two intensities of disturbance. In most other study areas forest cover tended to decline gradually with increasing disturbance intensity. These results support suggestions that environmental degradation is often gradual and continuous rather than catastrophic (Davidson 2000). Species richness tended to display much greater variation both among study areas, and at different time intervals (Fig. 5). Peaks in species richness at intermediate disturbance intensities were observed in Quilpué and to a degree in Central Veracruz; this is consistent with the intermediate disturbance hypothesis, which states that species richness is maximized at intermediate frequencies or intensities of disturbance (Connell 1978). Other study areas tended to demonstrate a decline in species richness with increasing disturbance intensity (e.g. Sary Chelek), or uneven responses (e.g. El Tablon and New Forest). Overall, species richness tended to display less resistance to disturbance than did forest cover, highlighting the fact that different measures of forest ecosystem state can be associated with very different responses to disturbance.

These results should be interpreted with caution, as disturbance intensity was here assessed on an ordinal scale, limiting the comparisons that can be drawn between case studies. This reflects a broader limitation of research into anthropogenic disturbance; robust and comparable measures of disturbance intensity are difficult to obtain (Turner et al. 2003b), and the precise ecological impacts can vary markedly between different types of disturbance. This highlights the importance of characterising the disturbance regime in detail, including its spatial heterogeneity and variability (Fraterrigo and Rusak 2008), when assessing either resilience or resistance.

Resilience as a policy and management goal

Given the widespread incorporation of resilience into environmental policy, forest managers and restoration practitioners will increasingly be required to deliver resilient forest ecosystems. The semantic uncertainty associated with resilience concepts, reflected in the large number of different definitions that have previously been proposed, will impede successful policy implementation. It is therefore pertinent to consider how the concept of resilience can best be defined as a forest restoration goal.

We recognise that a substantive and valuable literature has developed on concepts of ecological resilience, which is based on theories of CAS and on the assumption that ecosystems display multiple stable states. However, as noted earlier, evidence for the existence of multiple stable states in forest ecosystems is limited, and the assumption is difficult to test in a robust manner (Petraitis 2013). We therefore support the conclusions of Grimm and Calabrese (2011) that the concept of ecological resilience is of limited value because it has not been operationalized; it is unclear how to quantify or measure this definition of resilience and to identify its underlying mechanisms.

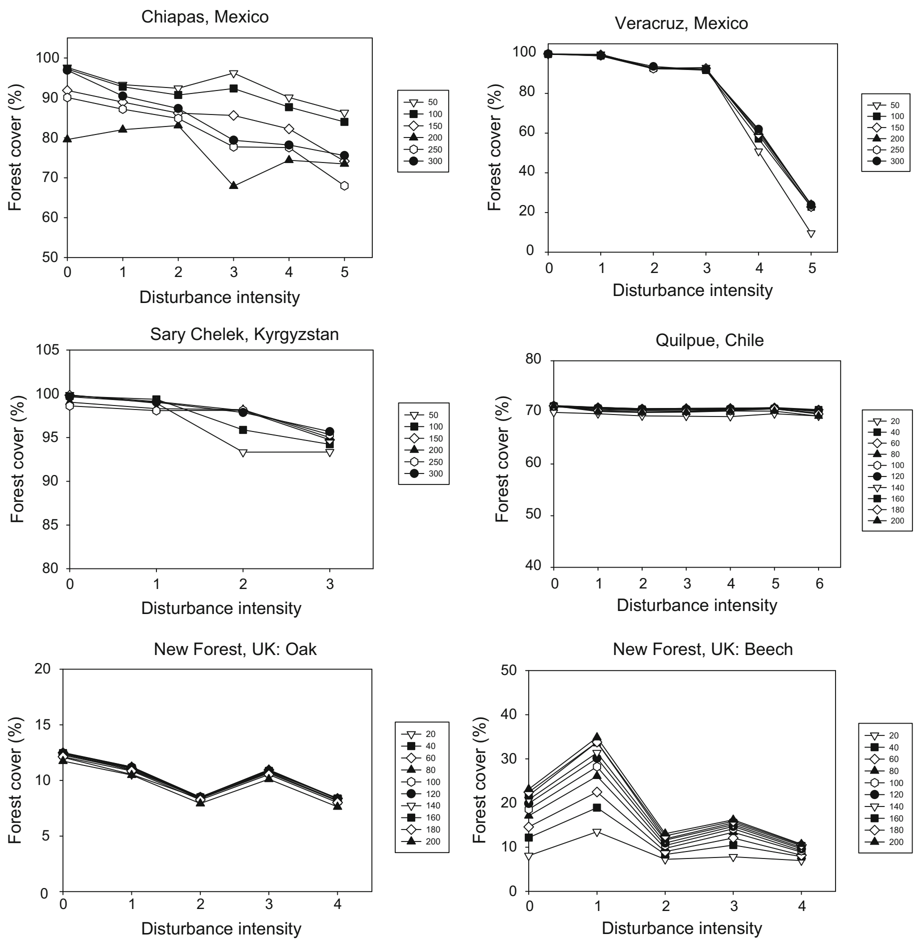

Fig. 4 Extent of forest cover under different disturbance regimes in relation to the intensity of disturbance, projected for five case study landscapes using the LANDIS II model (see text). For details of study areas, and the disturbance regimes illustrated by *different symbols*, see Table 3. *Different symbols* refer to time (in years) from the start of the modelled scenario. As detailed on Table 3, the precise disturbance regimes were different in each study area, reflecting the local situation. To enable comparison across study areas, disturbance regimes were ranked in terms of increasing order of disturbance, based on the degree of tree mortality. Model outputs derived from studies described by Cantarello et al. (2011, 2014) and Newton et al. (2011, 2013)

We therefore suggest that resilience could usefully be defined as the rate in recovery of variables as they return towards their equilibrium following a perturbation (Pimm 1984). It is this definition (often referred to as engineering resilience) that is most easily applied in practice, for example by measuring the rate of recovery following disturbance. We suggest that engineering resilience might therefore be considered as a forest restoration goal, a proposal that has a number of practical management implications. First among these is the identification of the restoration target, or the state of the forest ecosystem that restoration is aiming to achieve. This definition of resilience implies identifying a single equilibrium state that restoration actions will ultimately

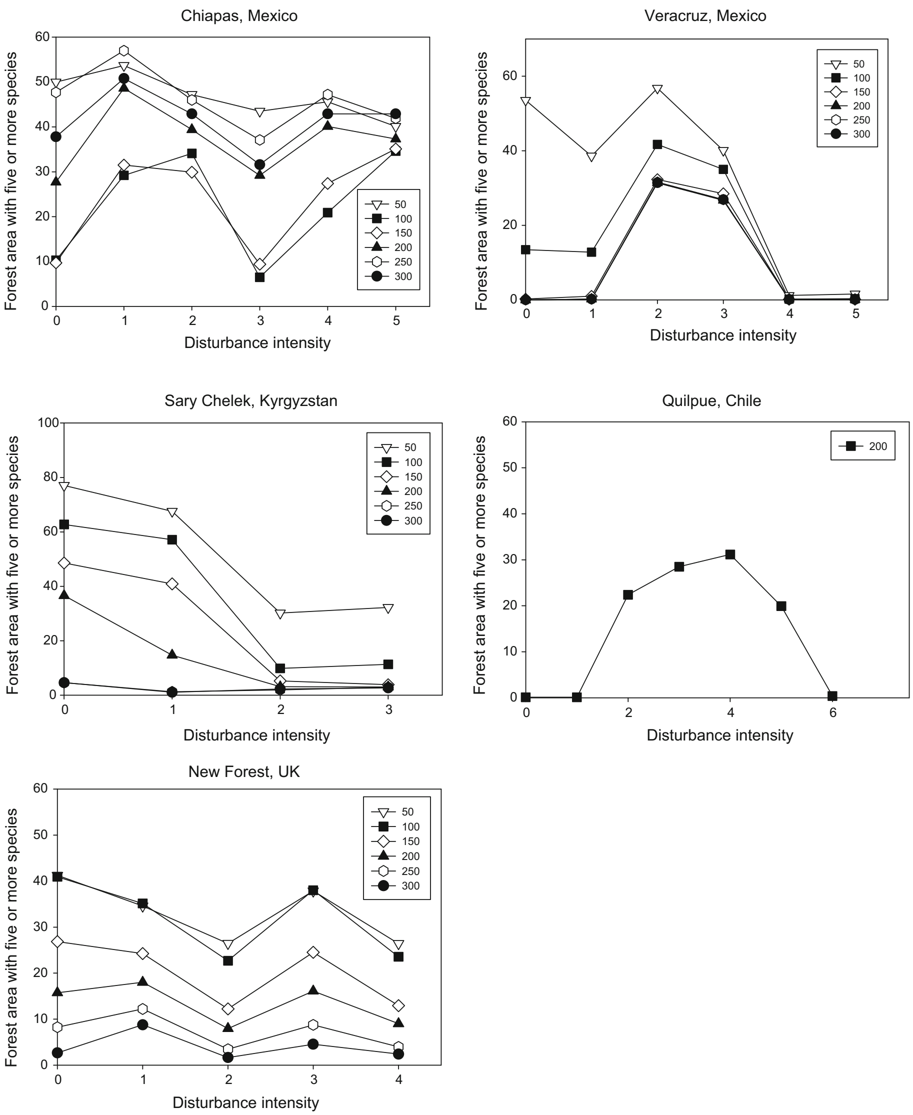

Fig. 5 Species richness of forest landscapes under different disturbance regimes in relation to the intensity of disturbance, projected for five case study landscapes using the LANDIS II model (see text). For details of study areas, and the disturbance regimes illustrated by *different symbols*, see Table 3. *Different symbols* refer to time (in years) from the start of the modelled scenario. As detailed on Table 3, the precise disturbance regimes were different in each study area, reflecting the local situation. To enable comparison across study areas, disturbance regimes were ranked in terms of increasing order of disturbance, based on the degree of tree mortality. Model outputs derived from studies described by Cantarello et al. (2011, 2014) and Newton et al. (2011, 2013)

deliver, which may be a state that prevailed prior to the onset of disturbance (Grimm and Wissel 1997). The identification of appropriate restoration targets has long been the subject of debate in restoration ecology, with the target selected depending on the particular objectives and context of any individual restoration initiative. Although the ultimate goal of ecological restoration is to return a system to some historical state, the difficulty of achieving this is widely recognised. Consequently many restoration projects aim to achieve a target that is defined in terms of ecosystem composition, structure, function and/or dynamics, compared to a relatively undisturbed reference ecosystem, rather than a historical state (Bullock et al. 2011; Palmer et al. 2006; Suding 2011). In such circumstances, resilience would have to be estimated as the rate of recovery to this specified state.

Further, we suggest that resistance might also be considered as a distinct restoration goal. This supports suggestions made by Côté and Darling (2010) that resistance, rather than recovery, might form a more appropriate management goal in relation to climate change. Although some authors have considered resistance to be an aspect of resilience (e.g. Walker et al. 2004), we follow Grimm and Wissel (1997) in identifying this as a separate property, which measures the degree to which a system remains unchanged despite the presence of disturbance. This might be of greatest relevance once a restoration target has been achieved; resistance would then provide an indication of how likely a system would persist in the restored state under a particular disturbance regime. This is relevant to forest restoration in landscapes subjected to chronic disturbance, as in the case studies presented above.

In many situations, however, forests may not readily return to an equilibrium state following disturbance. Many forest ecosystems are maintained in a non-equilibrial state because they are subjected to dynamic disturbance regimes; effectively, such systems can often be in some state of recovery from prior disturbance (Turner et al. 2003b), and their equilibria may be difficult to define. In such circumstances, adaptive capacity might provide a more appropriate forest restoration goal. This refers to the capacity of a system to adapt to disturbance through reorganisation (Carpenter et al. 2001; Gunderson 2000; Walker et al. 2004). Following Scheffer et al. (2012), the possession of adaptive capacity would imply a system undergoing continual change in response to continued and intensifying disturbance. In a restoration context, this would imply accepting a forest system resulting from management that might differ substantially from either a historic or reference state. In the context of current rates of environmental change, the associated development of non-analogue communities (Keith et al. 2009) and variation in recovery trajectories, a number of authors have suggested that it may not be possible to specify a single endpoint of a restoration project. Rather, the possibility of multiple end points might need to be explicitly recognised in restoration planning and management (Choi et al. 2008; Suding and Gross 2006; Suding 2011), together with the creation of entirely novel ecosystems (Hobbs et al. 2009; Richardson et al. 2010; Seastedt et al. 2008). This perspective has led to suggestions that terms other than ecological restoration should be used for such management interventions, for example reconciliation ecology or intervention ecology (Davis 2000; Hobbs et al. 2011; Suding 2011).

If a range of different outcomes are accepted as a result of restoration interventions, then engineering resilience is no longer an appropriate restoration goal. Rather, adaptive capacity would provide a more appropriate management objective, which should be made explicit both in management plans and associated policy. Currently, resilience is incorporated within many policies relating to climate change adaptation, but arguably adaptive capacity might provide a more appropriate goal for such policies. In the context of restoration, the distinction between restoring an ecosystem towards a specific target versus managing a system towards multiple or undefined states arguably represents one of the most important decisions facing practitioners. However, if adaptive capacity is the management goal, then this would no longer qualify as ecological restoration, at least in the strict sense.

Achieving resilience in forest restoration

Ideally, restoration practitioners would be provided with clear guidance regarding how resilience can be achieved in practice, but this is currently lacking. Some tentative suggestions are provided below to support the future development of such guidance. In practice, interventions undertaken to increase forest resilience will typically be combined with actions designed to achieve other management goals; managers will therefore be required to integrate different approaches into overall plans (Millar et al. 2007).

1. *Identify an appropriate resilience objective based on restoration goals*. If resilience is included among management objectives, then it is important to be explicit regarding which definition of resilience is being adopted. We suggest that if restoration is aiming to achieve a single reference or target, then engineering resilience provides the most appropriate definition. If multiple alternative restoration outcomes are acceptable, then adaptive capacity may be a more appropriate management objective. If maintenance of a forest ecosystem in its current state is a priority, despite a changing disturbance regime, then resistance could provide an appropriate objective.
2. *To achieve engineering resilience, management should seek to increase the rate of forest recovery following disturbance*. The potential for forest recovery will depend on the type, intensity and spatial heterogeneity of disturbance, and the survival of individual trees and their propagules (Chazdon 2003). Management can potentially support the key ecological processes underpinning forest recovery, the most important of which is succession. Examples of appropriate management actions that support succession include those designed to enhance seed dispersal, seedling establishment and growth, such as intensive management during revegetation (Millar et al. 2007), and the restoration of soil fertility on sites that are severely degraded (Chazdon 2003).
3. *To achieve adaptive capacity, management should seek to enable forests to adapt to changing environmental conditions*. This may potentially be achieved by reintroducing species or ecological processes that can strengthen the capacity of forests to self-organise in response to a changing environment (Cornett and White 2013). Examples of management actions include silvicultural strategies to help forests adapt to future environmental conditions, for example by increasing heterogeneity in stand structure and age. This may also be achieved by encouraging the establishment of a wider diversity of species, including those with higher tolerance of the environmental changes that are anticipated (Cornett and White 2013), for example by assisted migration (Millar et al. 2007).
4. *To achieve resistance, management should aim to strengthen the ability of forests to tolerate disturbance*. Management practices focused on increasing resistance seek to improve forest defences against the effects of environmental changes (Millar et al. 2007). Examples of appropriate management actions include reducing undesirable effects of fires, insects, and diseases, for example by establishing fuel breaks around higher risk areas; intensive removal of invasive species; or interventions such as resistance breeding, novel pheromone applications, or herbicide treatments (Millar et al. 2007). Resistance may also be supported by encouraging tree species with particular traits, such as vegetative reproduction and tolerance of fire or browsing (Newton et al. 2013; Newton and Echeverría 2014; Standish et al. 2014).
5. *To achieve resilient forest landscapes, management should support landscape-scale processes that enable forest recovery following disturbance*. The recovery ability of forest ecosystems will likely depend on the availability of seed sources and their

spatial distribution through the landscape, together with the distribution of refugial sites that have escaped disturbance. This leads to the concept of spatial resilience, based on the principles of landscape ecology (Folke 2006). Resilient landscapes may potentially be achieved by reducing fragmentation and increasing the size and connectivity of forest patches, for example by establishing physical corridors or 'stepping stones'; and reducing pressures on forests by improving the wider landscape, for example by buffering individual sites (Lawton et al. 2010; Standish et al. 2014; Suding 2011).

6. *Conduct monitoring in order to evaluate management effectiveness.* Given the need for evidence-based policy and management (Sutherland et al. 2004), there is a need to evaluate the effectiveness of management actions designed to support resilience. This requires monitoring using appropriate indicators (Millar et al. 2007). To date, progress has been limited in developing indicators explicitly for monitoring forest resilience, although the indicators suggested for assessing forest degradation could potentially be adapted for this purpose. Examples include wood volume; the value of non-timber forest products; forest area; area fragmented; species presence and abundance; the area affected by invasive species, fire and other disturbances; soil erosion; and carbon storage (Thompson et al. 2013).

Conclusions

The semantic uncertainty that has long characterised use of the term "resilience" in the ecological literature has undermined its practical application. As resilience is increasingly being incorporated into environmental policies, including those relating to ecological restoration, this uncertainty will increasingly present a challenge to forest managers and restoration practitioners. Specifically, practitioners will be unsure how to identify which forest restoration interventions might be required to deliver resilience, and how the effectiveness of their management actions might be evaluated appropriately.

We suggest that this problem can potentially be addressed by ensuring that the meaning of resilience is defined clearly when it is incorporated among management objectives. The choice of which meaning is most appropriate in a particular context will depend upon specific restoration or management goals. We suggest that if restoration is aiming to achieve a single reference or target ecosystem, then engineering resilience provides the most appropriate definition. Engineering resilience has the great advantage of being relatively easy to measure and therefore to apply in practice. If multiple alternative restoration outcomes are acceptable, then adaptive capacity may be a more appropriate management objective. If maintenance of a forest ecosystem in its current state is a priority, then resistance might provide an appropriate objective.

As demonstrated here, engineering resilience and resistance can potentially be estimated using approaches such as ecological modelling, long-term monitoring and meta-analysis. Such analyses can provide an understanding of underlying mechanisms, which is required to inform the development of practical guidance for restoration practitioners. Further research is also required to identify appropriate indicators of both resilience and resistance, to provide a robust basis for monitoring the response of forests to disturbance, and thereby enabling the effectiveness of restoration interventions to be evaluated (Cornett and White 2013). In a practical context, owing to the uncertainty of future environmental change and how it might affect forest ecosystems, there is likely to be a need for flexible restoration approaches that provide scope for modification as situations change. Different approaches are likely to be required in different contexts, recognising that resilience is strongly influenced by the characteristics of particular disturbance regimes, the ecological characteristics of forest ecosystems, and the historical legacy of previous forest use (Millar et al. 2007). Major questions remain, such as whether a focus on addressing local-scale disturbances will increase the resilience of a forest to global-scale pressures, and whether management for resilience would also increase the resistance of forests to disturbance (Côté and Darling 2010).

One of the principal challenges facing restoration practitioners is to decide whether the management objective is to achieve a single reference or target ecosystem, or whether multiple alternative outcomes are acceptable. Given the current era of rapid environmental change, the concept of achieving a single outcome of restoration is increasingly being seen as untenable in many situations (Cornett and White 2013; Millar et al. 2007). In such circumstances, management interventions might most usefully focus on strengthening adaptive capacity, by supporting those ecological characteristics and processes that will enable forest ecosystems to adapt to future change. Such approaches represent a significant departure from traditional conceptions of ecological restoration, however, and consequently they may need to be considered as an entirely different type of management intervention (Davis 2000; Hobbs et al. 2011; Suding 2011).

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