Toward an Era of Restoration in Ecology: Successes, Failures, and Opportunities Ahead

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Annu. Rev. Ecol. Evol. Syst. 2011. 42:465–87

First published online as a Review in Advance on August 29, 2011

The *Annual Review of Ecology, Evolution, and Systematics* is online at ecolsys.annualreviews.org

This article's doi:
10.1146/annurev-ecolsys-102710-145115

1543-592X/11/1201-0465$20.00

Keywords

resilience, ecosystem restoration, restoration ecology, recovery, degradation, ecosystem services, environmental change, novel ecosystems

Abstract

As an inevitable consequence of increased environmental degradation and anticipated future environmental change, societal demand for ecosystem restoration is rapidly increasing. Here, I evaluate successes and failures in restoration, how science is informing these efforts, and ways to better address decision-making and policy needs. Despite the multitude of restoration projects and wide agreement that evaluation is a key to future progress, comprehensive evaluations are rare. Based on the limited available information, restoration outcomes vary widely. Cases of complete recovery are frequently characterized by the persistence of species and abiotic processes that permit natural regeneration. Incomplete recovery is often attributed to a mixture of local and landscape constraints, including shifts in species distributions and legacies of past land use. Lastly, strong species feedbacks and regional shifts in species pools and climate can result in little to no recovery. More forward-looking paradigms, such as enhancing ecosystem services and increasing resilience to future change, are exciting new directions that need more assessment. Increased evidence-based evaluation and cross-disciplinary knowledge transfer will better inform a wide range of critical restoration issues such as how to prioritize sites and interventions, include uncertainty in decision making, incorporate temporal and spatial dependencies, and standardize outcome assessments. As environmental policy increasingly embraces restoration, the opportunities have never been greater.

1. INTRODUCTION

"The next century will, I believe, be the era of restoration in ecology"

–E.O. Wilson (1992)

Ecosystem: the dynamic complex of microbial, plant, and animal (including human) communities and the abiotic environment

Ecosystem restoration: the attempt to repair or otherwise enhance the structure and function of an ecosystem that has been impacted by disturbance or environmental change

Restoration ecology: the scientific study of repairing and managing disturbed ecosystems through human intervention

Although a key aim of environmental management should be the avoidance of degradation in the first place, an unfortunate truth is that humans are impacting most ecosystems around the globe. More than one-third of ecosystems have been converted for human use such as agricultural land and cities, and at least another third have been heavily degraded through fragmentation, unsustainable harvest, pollution, or exotic species invasions (Millenn. Ecosyst. Assess. 2005). As these impacts increasingly compromise biological diversity, human health, and food security, policy makers and managers will be pushed to evaluate investment in ecosystem restoration (Ferraro & Pattanayak 2006). Likewise, restoration ecology will be increasingly pushed to evaluate whether its science effectively informs successful management efforts (Hobbs & Cramer 2008, Palmer 2009).

Here, I assess where we stand in setting the course for this to become, as in E.O. Wilson's sentiment, the era of restoration in ecology. I focus on the successes and failures in restoration, how science is (and is not) informing these efforts, and ways to better address decision-making and policy needs. I first examine the current state of restoration ecology and what we know about successful ecosystem restoration, then evaluate mechanisms critical to success, and end with some recommendations.

2. ARE WE READY FOR THE ERA OF RESTORATION?

By any standard, restoration ecology is in an exciting growth period. The restoration of degraded ecosystems is becoming a primary focus of natural resource management for both terrestrial and aquatic environments (Millenn. Ecosyst. Assess. 2005). For example, at the 2010 meeting of the Convention on Biological Diversity in Nagoya, Japan, countries committed to a new target of restoring 15% of the degraded ecosystems worldwide by 2020 (Secr. Conv. Biol. Divers. 2010). In 2009 the Indonesian government received as many applications for forest restoration licenses as it did for logging concessions (Bird Life Int. 2010). The U.S. Secretary of Agriculture underscored restoration as a driving principle in forest policy by calling for a "complete commitment to restoration" (USDA 2009). Australia's Biodiversity Conservation Strategy set a five-year target of restoring 1,000 km^2 of fragmented landscapes and aquatic systems to improve connectivity (Natl. Biodivers. Strategy Rev. Task Group 2009). The United Nations Environment Programme called ecosystem restoration among the most profitable public investments for economic growth and overcoming poverty (Nellemann & Corcoran 2010).

Alongside the increased societal interest in and demand for restoration, the science of restoration ecology has grown rapidly. Ecological journal articles in the Institute for Scientific Information (ISI) Web of Science database containing the keyword "restoration" have approximately doubled every five years, increasing from 83 in 1995 to 174 in 2000, 379 in 2005, and 683 in 2010. Island Press has published a popular book series on the science and practice of ecological restoration that currently encompasses 15 volumes. Almost a quarter of U.S. institutions with programs in natural resources and biology offer courses that specifically focus on restoration ecology (Nelson et al. 2008).

Given the rapid expansion of a young discipline, growing pains are not surprising. Restoration ecology has faced critiques from both the science and practice sides of the field. From the science side, the criticism that restoration ecology is largely ad hoc, site specific, and lacking a conceptual framework has haunted the discipline almost since its beginning (Allen et al. 1997, Hobbs &

Norton 1996). From the practice side, the question of whether science is relevant for informing successful ecological restoration has plagued the field (Cabin 2007). At the heart of these critiques are questions about the success of restoration and how science can inform and improve it.

Restoration ecology is at a critical juncture: as environmental policy increasingly embraces restoration, the need to inform decision makers about effectiveness and uncertainty in restoration options has never been greater (Brudvig 2011, Hobbs 2007, Palmer 2009). When is restoration an effective conservation investment? How often does restoration achieve intended results? How can restoration become more successful and less uncertain? Together, this information will enhance conservation outcomes, aid implementation of policy, and further investment in restoration science.

3. SUCCESS IN RESTORATION ECOLOGY

Although publications, organization membership, and educational programs provide some insight into the growth of restoration science, one of the most crucial measures is the success of the practice that it informs. Certainly, a multitude of impressive success stories give excitement and credence to our ability to restore ecosystem function (**Figure 1**; Nellemann & Corcoran 2010). However, although there is wide agreement that evaluation is a key to future progress in restoration ecology (Hobbs & Norton 1996, Palmer et al. 2005), comprehensive surveys of successes—and failures—are rare.

It is difficult to evaluate the success of restoration projects in part because of limited information. Few projects predetermine criteria for success, fewer quantitatively monitor project outcomes, and the little information that is collected is often difficult to obtain. Outside of compensation standards set for mitigation, which are often based only loosely on ecological criteria (Matthews & Endress 2008, Zedler & Callaway 1999), few established standards for measures of success or methods to evaluate success exist (Palmer et al. 2005). Given the low monitoring rate and difficulties in accessing monitoring data, cross-project analyses that evaluate the factors and techniques that lead to success are rare. The National River Restoration Science Synthesis project, one of the most extensive efforts to evaluate restoration practice to date (Palmer et al. 2005, 2007), found that less than half of all projects set measurable objectives and used quantitative measurement data to evaluate project success (Bernhardt et al. 2007). Similarly low monitoring rates were found for salt marsh restorations in northwestern Europe (Wolters et al. 2005) and road construction compensation efforts in Germany (Tischew et al. 2010).

Another important factor contributing to the challenge of evaluation in restoration is the issue of setting appropriate standards with which to evaluate success. Success itself is a nebulous part of the lexicon of restoration (Ruiz-Jaen & Aide 2005, Zedler 2007); target criteria can vary widely in both ambition and rationale, even among stakeholders within the same project. Ecological outcomes also differs from success related to economics, aesthetics, recreation, and education—all valid criteria in their own right. Setting evaluation standards, which minimally involve ecological success and economic costs, requires consensus on a set of well-accepted criteria among scientists, funding agencies, and citizen groups (Palmer et al. 2005). This consensus is essential for project outcomes to be translated into reporting systems at the regional and national levels.

The question of evaluation is further complicated by the fact that baselines related to composition and function are rapidly shifting in many ecosystems. Owing to unprecedented changes in climate, shifts in land use, and changes in biodiversity, the goal of returning to some past point in time and resetting the ecological clock can be unrealistic in some cases (Choi et al. 2008, Hobbs & Cramer 2008). Many question even the name restoration itself, suggesting that reconciliation ecology, intervention ecology, or similar terms would be more appropriate to the mission of the

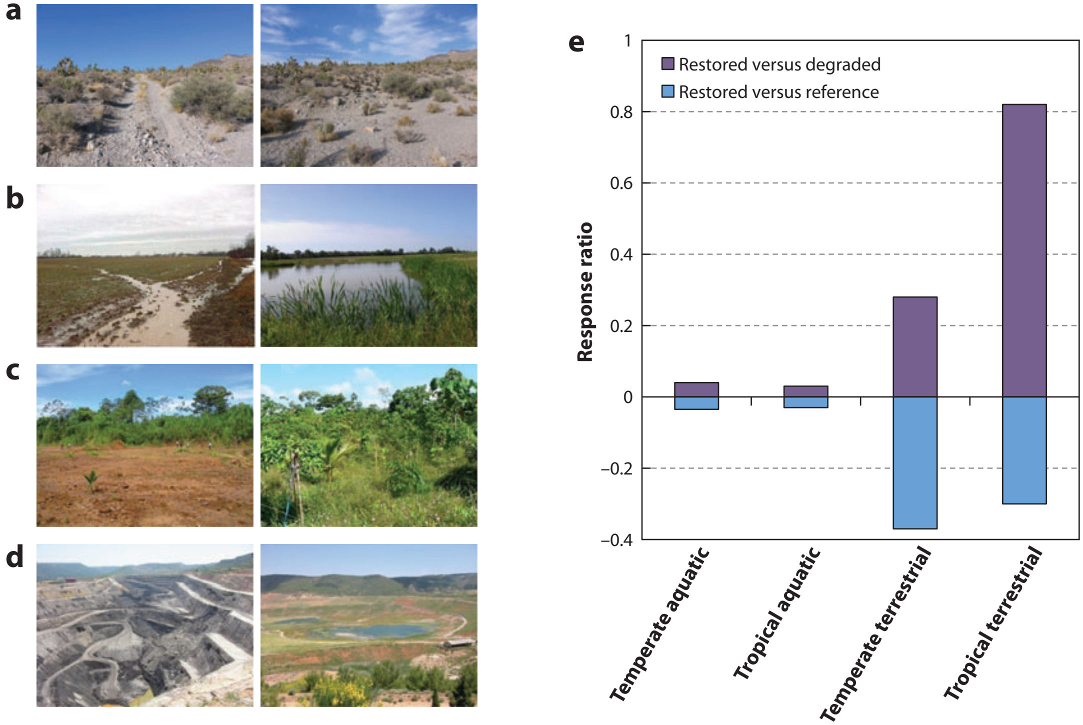

Figure 1

(*a–d*) Before (*left*) and after (*right*) pictures of restoration projects that are part of the Society of Ecological Restoration's member-submitted Restoration Project Showcase (**http://www.ser.org**): (*a*) Cow Camp Road Decommissioning at the Desert National Wildlife Refuge, Las Vegas, Nevada; (*b*) Fort Erie wetland restoration, Ontario, Canada; (*c*) Proyecto Naturaleza y Comunidad, Costa Rica; and (*d*) Corta Alloza and Utrillas Coal Mine Restoration, Spain. Photos courtesy of Soil-Tech Inc., Niagara Peninsula Conservation Authority, Jorge Borantes Montero, and Francisco A. Comin. More details on these and other case studies can be found in Nellemann & Corcoran (2010). (*e*) Ecosystem responses to restoration in different biomes from Rey Benayas et al. (2009). Response ratios are based on study comparisons of restored versus degraded sites (*purple bars*; the positive values indicate that restoration enhances ecosystem services) and restored versus reference sites (*blue bars*; the negative values indicate that the ecosystem services of restored sites remain lower than those of reference sites).

discipline (Davis 2000, Hobbs et al. 2011, Richardson et al. 2010, Zweig & Kitchens 2010). Some advocate for the expectation of the development of novel ecosystems (Hobbs et al. 2009, Seastedt et al. 2008), acknowledging cases in which we may never be able to return to a historical reference. As a consequence of these shifting baselines, restoration goals are being considered more broadly to include ecosystem services and ecological resilience to future changes (e.g., Nellemann & Corcoran 2010).

Novel ecosystems: new, nonhistorical configurations of ecosystems owing to changing species distributions and environmental alteration through climate and land use change

Bounded by these limitations, I examine success according to four general paradigms in the field. Some of these paradigms, such as guiding recovery after disturbance, have been prominent for several decades and provide a rich literature to explore outcomes. Others, such as restoration of ecosystem services, are emerging as major thrusts in the field.

3.1. Restoration to Guide Recovery

Guiding recovery of degraded systems has been a primary emphasis of restoration ecology for the past several decades. Although succession models encompass dynamics that do not assume predictable temporal trajectories, such as arrested succession (Acacio et al. 2007), the recovery paradigm in restoration is generally based on the successional concept of smooth turnover over time followed by an eventual arrival at a stable climax point (Matthews et al. 2009b). This paradigm essentially looks backward to set restoration success criteria, aiming to guide a degraded system to a condition resembling its past structure or composition. As information about past composition or structure is often hard to obtain, comparisons with reference sites, matched as much as possible to sites with similar environmental conditions or more broadly to an estimated historical range of variability, are often used to evaluate success. A second comparison with unrestored degraded systems is often used to provide a "no action" baseline. Success criteria generally include aspects of vegetation structure and composition or diversity (Brudvig 2011, Ruiz-Jaen & Aide 2005), based on the assumption that the recovery of fauna and ecosystem processes will follow the establishment of vegetation. New approaches to establish target reference levels are also under development; for instance, Partel et al (2011) suggest a criterion to minimize dark diversity, the number of species that are absent from an ecosystem but that belong to its species pool.

Passive restoration: after the removal or reduction of the adverse (degrading) disturbance, using natural regenerative processes without additional remedial actions

Active restoration: human interventions aimed at assisting in the reassembly of a damaged ecosystem

Research addressing the recovery paradigm indicates widely variable outcomes: Some projects are considered quite successful, whereas others are seen as dismal failures. Jones & Schmitz (2009) surveyed 240 studies that focused on recovery after disturbance across a wide range of aquatic and terrestrial systems. Although the authors did not distinguish passive restoration from active restoration, their survey is the most comprehensive regarding recovery to date. They found that more than a third of the studies (35%) documented complete and relatively rapid (ranging from 10 to 40 years) recovery. The remaining studies found either mixed measures of recovery (35%) or no recovery in any measured variable (30%). Analysis of the subset of the studies measuring species composition ($n = 37$), a common criterion in restoration, indicates that recovery based on species composition was less frequent (23%) than recovery based on other criteria (i.e., total biomass, diversity).

Similarly, research specifically related to restoration illustrates a wide range of outcomes: restoration can result in relatively quick and complete recovery; it can be partially successful with some degree of improvement from degraded conditions; or it can yield little or no recovery. Complete recovery relative to reference sites often occurs where colonization by the desired species is still possible—either through regional persistence or local storage mechanisms (such as seed banks)—and where soils and physical features remain largely intact. For instance, in oak depressional wetlands in Ohio, Martin & Kirkman (2009) documented complete recovery relative to reference sites after five years, a success they attribute largely to a seed bank legacy. Similarly, in Argentina, Cuevas & Zalba (2010) found that invasive pine removal allowed grassland recovery within four years if removal occurred early in the invasion process. By fencing to remove grazers, Sawtschuk et al. (2010) found that maritime clifftop vegetation recovered within 16 years. Interestingly, the few sites that did not recover in this time frame were ones invaded by an exotic species.

A more typical result of studies following restoration trajectories is that restoration is more effective than a no action option but often does not result in complete recovery (Rey Benayas et al. 2009). In a survey of 36 wet meadow restoration projects across western Europe, Klimkowska et al. (2007) found that most projects increased species richness to approximately 10% of the regional species pool; the most effective measures achieved an increase of up to 16%. Similarly, Wolters et al. (2005) found that restored salt marshes contained less than 50% of the regional target species. In studies that follow changes through time, many find indications of a trajectory toward the reference condition and then extrapolate that recovery would occur given more time. For instance, after 28 years, vegetative and avian components of a holm oak system in southern France recovered only partially relative to reference sites; by extrapolating the recovery trajectories, the authors estimate the recovery times of these two components to be at least 50 and 35 years, respectively (Jacquet & Prodon 2009). In other cases, incomplete recovery may be relatively permanent because constraints do not self-correct over time. For example, across wetland pothole restorations in the midwestern United States, Aronson & Galatowitsch (2008) identified several barriers to recovery—isolation, infrequent flooding, and invasive species—that did not weaken over time. Similarly, Ballantine & Schneider (2009) documented constraints to soil development in restored depressional wetlands remaining after 55 years owing to changed litter decomposition processes.

Complete lack of recovery can also occur in restoration. Although frequency is difficult to estimate owing to publication bias toward successful outcomes (Zedler 2007), many cases of non-recovery appear to be related to strong abiotic-biotic feedbacks that thwart restoration actions (Suding et al. 2004). For example, seagrasses in the Dutch Wadden Sea became virtually extinct in the 1930s and, despite repeated restoration attempts, have never recovered. Van der Heide et al. (2007) demonstrate that positive feedback between turbidity and seagrass stymied restoration attempts: The high turbidity of suspended sediment constrained the reestablishment of seagrass, but the presence of seagrass was necessary to reduce turbidity. For some types of degradation, few cost-effective interventions exist. For instance, the legacy of some intensive agricultural practices (e.g., erosion, fertilization) can last millennia, and few methods to reverse these changes exist (Cramer et al. 2008, McLauchlan 2006).

3.2. Restoration as Compensation for Habitat Loss

Biodiversity trading programs (which include biodiversity compensation, offsets, and biobanking) have proliferated internationally and are promoted by policy makers as facilitating both conservation and development. In this paradigm, restoration offsets the destruction of natural ecosystems. For instance, the wetland permit program established under the U.S. Clean Water Act allows wetland impacts to be offset through compensatory wetland mitigation (US Army Corps Eng. & EPA 2008). Although these policies operate under many assumptions that are similar to those in the recovery paradigm (see Section 3.1), planners face additional challenges relating to fair offset evaluation and spatial relocation.

In compensation, estimation of the likelihood of restoration success is essential because future gain is uncertain whereas the immediate loss is permanent (Moilanen et al. 2009). Even when the area restored is larger than the area lost, compensation seldom succeeds in restoring structure, composition, or function (Hilderbrand et al. 2005, Matthews & Endress 2008, Quigley & Harper 2006, Reiss et al. 2009, Tischew et al. 2010, Zedler & Callaway 1999). For instance, Reiss et al. (2009) assessed the success of 29 wetland mitigation banks in Florida. They found that 40% met permit criteria, whereas 17% were not close to compliance. In an assessment of 16 fish habitat compensation projects throughout Canada, which required reporting of fishery production, Quigley & Harper (2006) found that 63% of the sites experienced net losses in productivity whereas 12% achieved a net gain. Similarly, only a third of restoration goals were achieved in compensation projects to counteract impacts of road construction (Tischew et al. 2010).

Given this uncertainty, how much habitat compensation is required to offset the loss of high-quality habitat and result in no net loss? In biodiversity trading policy, this difference is often referred to as an offset ratio (Moilanen et al. 2009) and in many ways reflects a quantification (albeit often debatable) of anticipated restoration success. For instance, Quigley & Harper (2006) report that although policy required offset ratios to be on average approximately 7:1 (area gained to area lost), the mean offset ratio actually implemented was 1.5:1, which resulted in only 6 out of 16 cases reaching no net loss in terms of habitat productivity. In dry grasslands in Switzerland, Dalang & Hersperger (2010) estimate offset ratios that approach 200 in some cases, certainly not a viable conservation option. Moreover, in some cases, policy expectations may go beyond the capability of science. For instance, the 2008 revision of the U.S. Clean Water Act includes the creation of new stream habitats, a largely uncertain endeavor (Stokstad 2008).

Spatial connectivity and temporal lags are other critical issues in habitat compensation. To maintain regional biodiversity, trading programs must replace ecological interactions and functions lost in development. A common pattern is to replace small focal systems lost in urban areas with aggregated ones in more rural areas (BenDor et al. 2009), although we know little about spatial dependencies and how they vary among different ecosystem components (e.g., biogeochemistry versus avian population structure). In addition, restorations take time to provide the same functions that established habitats provide; the lag between habitat loss and creation can substantially affect population viability (Maron et al. 2010).

Ecosystem services: benefits people obtain from ecosystems, including provisioning (e.g., food, water), regulating (climate, disease), supporting (nutrient cycling), and cultural (aesthetic, recreation) services

3.3. Restoration to Deliver Ecosystem Services

An ecosystem services framework is increasingly being utilized to better understand costs and benefits in restoration, thus forming a promising avenue to link science, economics, and societal values. This framework emphasizes services of ecosystems that benefit humankind, such as timber resources, disease control, crop pollination, and recreation, rather than ecosystem function or biodiversity per se. For example, the United Nations Environment Programme report on ecosystem restoration argues that restoration should be considered a policy option for issues such as water supply and wastewater management, disaster prevention, carbon sequestration, and mitigation of climate change (Nellemann & Corcoran 2010). In a notable example, Birch et al. (2010) quantified the values of dry forest restoration (carbon sequestration, timber, nontimber resources, and tourism) and compared them with the costs of restoration (loss of livestock production, fencing, tree establishment) under three restoration scenarios using a simulation model in four areas of Latin America. They found passive restoration to be most cost effective; costs generally outweighed the benefits of active restoration (such as tree planting).

The effects of different restoration approaches on the recovery of ecosystem services are poorly studied despite recognition of the links between biodiversity, functional traits, and ecosystem services (Diaz et al. 2007). For example, Rey Benayas et al. (2009) conducted a meta-analysis of 89 restoration studies and found that although restored projects did not achieve increases to the level of intact reference systems, increases in biodiversity were related to increased ecosystem services (**Figure 1*e***). Restoration in terrestrial tropical ecosystems yielded the greatest improvements in biodiversity and services, and terrestrial temperate systems recovered the least compared with intact reference systems. Similarly, Worm et al. (2006) found that restoration of ocean biodiversity increased productivity fourfold. In other cases, functional redundancy among species may allow restoration of ecosystem processes to be more feasible than restoration of biodiversity. For instance, Jones (2010) found that ecosystem variables such as isotopic carbon signatures and marine-derived nitrogen levels recovered within 22 years of rat eradication in New Zealand island systems. O'Brien et al. (2010) found that 20-year-old tallgrass prairie restorations had soil profiles approaching those in reference prairies.

Many issues remain in how to value restoration-related ecosystem services (Galatowitsch 2009, Palmer & Filoso 2009). As restoration of one ecosystem service may come at a cost to another, one particular challenge is how to ensure multifunctionality in both the short and long term. For instance, although planting a few short-lived but fast-growing species is a common approach for carbon offsets, these plantations do not approach the diversity of naturally occurring tropical forests and can have a high rate of failure (Wuethrich 2007). Inclusion of long-lived, slow-growing tree species with denser wood and slower turnover would better promote long-term carbon sequestration as well as diversity (Chazdon 2008).

Ecological resilience: the capacity of a system to absorb disturbance and reorganize while still retaining similar function, structure, and feedbacks

Surprisingly, cost estimations and ecological analyses of outcomes are often not combined. In a recent economic analysis of the value of ecosystem restoration, the benefit-cost ratio of restoration ranged from 2.8 for coral reefs to 15 for rivers and lakes, 37 for tropical forests, and 75 for grasslands (TEEB 2008). Although these are appealing ratios, they are largely based on costs of restoration and valuation of increased ecosystem services; they do not incorporate uncertainty related to successful restoration outcomes. Incorporating the benefits of multiple ecosystem services and cost effectiveness of a range of treatments can greatly enhance restoration decision making. For example, in restoration decisions aimed at controlling postfire invasion of cheatgrass (*Bromus tectorum*), Wainger et al. (2010) developed an optimization model that focused on several ecosystem services (antelope hunting, property protection from fire, sage-grouse habitat provision, and forage production) and incorporated cost-effectiveness ratios of restoration options (ranging from aerial seeding to a combination of herbiciding, seeding, and chaining). Actual management decisions were based on selecting sites that provided high benefits in terms of one particular service and sometimes opting for high-intensity restoration treatments owing to greater success rates. However, the optimization model showed that service benefits would increase threefold if managers selected sites to optimize multiple services and utilized treatments with the greatest cost-effectiveness ratios (often the lowest-intensity treatments). Even when benefits cannot be quantified monetarily, a cost-benefit approach is critical to efficient use of restoration resources both within and among projects (Wilson et al. 2011).

3.4. Restoration to Ensure Resilience

A resilience-based approach in restoration is a logical extension of current ecosystem-based management practices that builds on an improved understanding of the dynamics of thresholds and reinforcing feedbacks as well as the importance of functional groups in ecosystem processes. Ecological resilience is important to ensure that the restoration is sustainable, not requiring intensive and ongoing interventions, as well as that it has adaptive capacity, the ability to adjust and maintain function in the face of environmental change. Ecological redundancy in critical functional groups that allow for a diversity of responses to environmental drivers but maintain similar effects on ecosystem function can lead to increased ecological resilience (Brand & Jax 2007, Elmqvist et al. 2003, Nystrom 2006). Landscape factors such as spatial heterogeneity, fragmentation, and connectivity should affect resilience (Millar et al. 2007, Nystrom 2006, Pardini et al. 2010). Although ecological resilience can be equated with desirable ecosystem characteristics, alternatives exist. The difficulty of restoration illustrates that degraded low-diversity systems can have the same, or even greater, levels of resilience compared with high-diversity systems (Firn et al. 2010, Suding et al. 2004) and that the spread of problematic invasive species can be a downside to connectivity (Crossman et al. 2011). In fact, Cote & Darling (2010) argue that degraded coral reefs should be considered more resilient to climate change because species representative of degraded conditions may be better able to tolerate perturbations.

Although many management agencies and conservation organizations have embraced resilience as a concept, concrete examples of actual management for resilience remain rare, perhaps because quantification of resilience and the factors that affect resilience is difficult (Hughes et al 2010, Suding & Hobbs 2009). Applications usually focus on associated factors, such as diversity, connectivity, and heterogeneity, that are assumed to lead to greater resilience. For instance, U.S. Forest Service policy assumes that management for increased genetic and species diversity should increase resilience and adaptive capacity (USFS 2010). More empirical investigations, such as comparative site analyses focused on when and why some ecosystems degrade and others do not (and vice versa, that is, when and why some ecosystems recover and others do not), is a crucial next step to broaden the scientific foundation in this emerging arena.

3.5. Conclusions

To a certain extent, restoration will always be a gamble. Improved monitoring and increased access to monitoring data are crucial to better understand the range of success in restoration projects as well as to adaptively manage projects themselves. Assessments related to the recovery paradigm (Section 3.1) indicate that restoration success is achieved relatively often—in a third to half of projects in most analyses. Success when restoration is used as habitat compensation (Section 3.2) is often lower. Thus, although restoration is often possible and results in net positive benefits, it often does not go as well as planned. The inability to meet set criteria in many projects occurs at a high enough frequency to bring into question our ability to set realistic goals and our confidence in meeting these goals (Hobbs 2007). Increasingly, it is clear that historical reference benchmarks may not be feasible goals and that restoration is also being viewed in terms of enhancement of ecosystem services (Section 3.3) and resilience (Section 3.4). Although less information exists to evaluate these paradigms, it is critical for restoration science to inform their growing use.

4. CRITICAL MECHANISMS

Clearly, restoration success cannot be described along a single axis with straightforward solutions. Because resources for restoration are limited, viewing restoration success as a dynamic concept across both space and time is one way to better understand which mechanisms may be critical for success (**Figure 2**). With this approach, the variable outcomes in restoration can be grouped

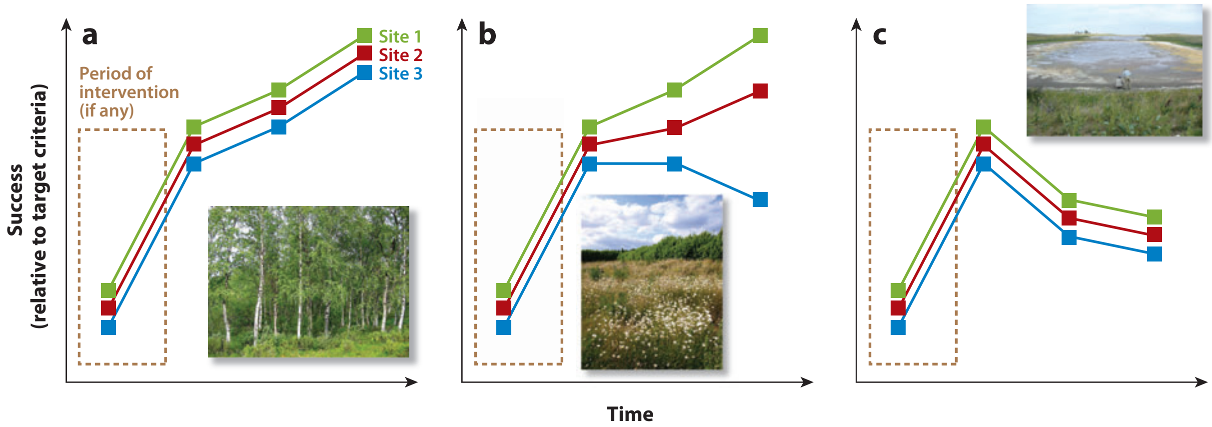

Figure 2

Spatial and temporal dynamics can indicate restoration constraints and mechanisms critical to meet project goals. Many possible dynamics exist; three are shown here: (*a*) convergence toward a target goal, (*b*) unintended divergence among restoration sites, and (*c*) trajectories that deviate from a target goal. Each line represents a different restoration site; for clarity, it is assumed that sites were similar in initial characteristics and restoration techniques employed. The brown box indicates the period of active intervention, if such techniques are employed. Inset pictures illustrate examples of each type of trajectory: (*a*) *Betula pendula* woodlands often recover unassisted in spoil heaps in coal mining restoration (Prach 2003), (*b*) restoration of lowland calcareous grasslands following agriculture often results in variable outcomes (Fagan et al. 2008), and (*c*) limited recovery in prairie pothole wetland restoration (Aronson & Galatowitsch 2008). Photos courtesy of Percita Dittmar, David Glaves, and Joanna Thamke, respectively.

by system dynamics to better understand what might lead to, or what might impede, successful restoration (Suding & Hobbs 2009). Comparison across sites with similar restoration history can point to variation in constraints such as abiotic conditions, landscape location, or past disturbance history. Comparison across time can point to whether these constraints can self-correct with time or whether further intervention is needed. A focus on the mechanistic underpinnings across this range of trajectories is a way to determine where and when we can rely on passive restoration approaches to achieve restoration goals, when more intensive intervention is required, and whether chosen restoration targets represent unrealistic expectations (Holl & Aide 2011, Matthews & Spyreas 2010, Prach & Hobbs 2008). This understanding is essential for the identification of needed interventions and effective allocation of limited resources.

4.1. Convergence Toward a Target Goal

Given similar restoration techniques, restored sites can be relatively similar to one another and all progress toward the same restoration target (**Figure 2*a***). This scenario (simple, rapid, and predictable progression) forms much of the basis of policy standards related to mitigation and project evaluation. Indeed, many examples support these dynamics. This scenario is most likely where abiotic conditions are relatively uniform, shifting alongside biotic changes, and where the species pools are relatively intact (Pickett et al. 2009, Temperton et al. 2004). Minimal intervention and maximal reliance on natural regeneration may be most effective and should be the first approach considered under these conditions (Holl & Aide 2011, Prach & Walker 2011). Restorations that linger in an intermediate state between the initial and target states (e.g., Fagan et al. 2008) may require additional intervention, but these actions should be relatively straightforward because successional pathways eventually converge upon the same target (Mitsch & Wilson 1996, Sheley et al. 2010).

4.2. Unintended Divergence Across Restoration Sites

Restorations can also diverge from one another and progress toward a range of outcomes despite use of similar restoration techniques (**Figure 2*b***). Several studies suggest that this is a common pattern in restoration (Fagan et al. 2008, Prach 2003, Pywell et al. 2002, Seabloom & van der Valk 2003). This scenario may indicate that some restoration sites need additional attention to abiotic conditions, such as reducing nutrient loads or increasing flooding frequency, or that some sites have lower restorative potential than others (e.g., prairies in loamy versus sandy soil; Baer et al. 2010). Variability could also indicate additional dispersal constraints or propagule pressure owing to landscape location (Matthews et al. 2009a). Interventions to address these additional constraints concerning abiotic conditions and landscape matrix may be difficult, and they highlight the importance of using past project outcomes as guides for prioritizing restoration sites. Alternatively, variability might better reflect natural dynamics and be considered a desired characteristic, particularly for goals related to resilience at larger spatial scales (e.g., Amoros & Bornette 2002).

4.3. Trajectories That Deviate from Target Goals

In addition, restorations can predictably converge toward a common composition or function, but one that differs from target goals (**Figure 2*c***). This dynamic may be the clearest sign of the failure of restoration techniques (Aronson & Galatowitsch 2008, Ballantine & Schneider 2009, Matthews & Spyreas 2010, Tanentzap et al. 2009). However, on the basis of short-term assessments, it is often difficult to foresee whether trajectories will be nonlinear, initially deviating from the expected target but eventually progressing toward it, or whether the lack of progression will be permanent (at least on the timescale relevant to management). For instance, Matthews & Spyreas (2010) tracked the trajectories of restored wetlands in Illinois; they found that initial successful trajectories ended up not meeting expectations because of invasions of nonnative species. Thresholds can also occur in the opposite direction. For instance, the initial lack of recovery in river macroinvertebrates impacted by historical mine pollution was followed by rapid changes toward a target goal once water quality reached a certain level (Clements et al. 2010).

There are several reasons why restorations become stuck and fail to progress toward target goals. Although active restoration can be viewed as jump-starting or facilitating a successional process, it can also have unintended consequences. For example, introduction of species may circumvent the time it takes for natural dispersal processes to occur, moving the system more quickly toward a goal community (Cox & Anderson 2004). However, introductions can also lower the eventual diversity of the community if the initial establishing species prevent the colonization of other desirable species (Fagan et al. 2008, Simmers & Galatowitsch 2010). Likewise, the genetic structure of the introduced populations can strongly influence success (Travis & Grace 2010), where ensuring both local adaptation and genetic diversity can be challenging (Vander Mijnsbrugge et al. 2010). Other techniques can also have unintended effects. For example, repeatedly liming acidified lakes increased the abundance of disturbance-related species rather than those more characteristic of more circumneutral systems (Angeler & Goedkoop 2010). Given that increased intervention is costly and can result in unintended effects, interventions should be carefully considered and weighed against less intensive (and perhaps more extensive) options.

In many cases, restoring the disturbance regime does little to undo the entrenchment of an undesired set of species (Suding et al. 2004). Historical legacies, such as effects of invasive plants on soil communities, species priority effects, and biophysical habitat changes, may be important factors. For instance, Tanentzap et al. (2009) found little recovery in a New Zealand forest system after 40 years of reduction of deer densities; the authors speculate that restoration will require a large-scale disturbance that causes mortality in the browse-resistant species and allows canopy turnover. Legacies also may be associated with degree of disturbance prior to restoration. For instance, Klanderud et al. (2010) found that the number of times tropical pastures were slashed and burned influenced recovery, and Seymour et al. (2010) found that past grazing intensity, rather than the restoration action, was the strongest influence on recovery in Karoo rangelands in South Africa. In addition, relatively small initial differences may cause sites to diverge (Houseman et al. 2008, Petraitis & Dudgeon 2005, Woods 2007); although it may be difficult to document whether the system is indeed bifurcating to alternative states, transient dynamics lasting several decades may be enough to warrant additional management actions. In these cases, establishment of a desired transition trajectory will require disrupting ecological feedbacks that constrain recovery while encouraging those that allow for progression to occur (Firn et al. 2010).

Alternatively, this dynamic may not so much indicate that the restoration is unsuccessful as that the target goals are unfeasible and need to reflect current—rather than historic—climate, land use, or species pools. Local extinctions of desirable native species accompanied by dominance of exotic species with strong impacts on ecosystem processes commonly have slowed or stopped recovery in both terrestrial and aquatic restorations (Florens et al. 2010, Hilt et al. 2010, Tanentzap et al. 2009). For instance, a 70-year restoration of a shallow lake via reduction of nutrient loading reestablished some aspects of community structure, but owing to the regional loss of rare species, diversity remained lower than in the reference period (Louette et al. 2009). If a target native species is lost from the species pool, naturally occurring replacement by a functionally similar nonnative species may not meet target goals in terms of composition but could still achieve desired functional outcomes. This may prove to be the best available outcome given the changed species pools. In cases such as these, convergence to historically novel communities may be hard to avoid and does not necessarily indicate a failure (Hobbs et al. 2009).

4.4. Conclusions

There are good examples of restoration projects exhibiting a wide range of dynamics; an important question is whether there are constraints or system characteristics that may serve as indicators. For instance, in a synthesis of wetland restoration projects, Matthews et al. (2009b) found that different indicators of restoration progress showed different trajectories over time, not all indicators of restoration progress showed an incremental linear trajectory, and in some cases recovery took much longer than the time frame during which restoration projects are typically monitored. The spatial and temporal variability in these dynamics highlight the difficulty in using short-term responses to indicate long-term recovery; it is essential to accompany spatial and temporal analyses with an understanding of the underlying mechanisms that act to constrain or facilitate change. In particular, anticipating the likelihood that constraints will self-correct over time may distinguish between a temporary delay in a successional progression and a permanent sticking point. Although research often focuses on local processes that can constrain restoration trajectories (Brudvig 2011), historical and landscape factors (e.g., land use legacies, dispersal) can often strongly constrain recovery and deserve increased attention.

5. IMPROVING SUCCESS

In a field flush with value-laden terms such as "desirable," "degraded," and "intact," it is important to recognize that restoration is largely a human endeavor. Values dictate many steps of a project: who is involved in the decision-making process, what the desirable condition for an ecosystem should be, how and when that condition is to be attained, and how economic, social, and cultural concerns will be affected (Burke & Mitchell 2007, Davis & Slobodkin 2004). Partnerships with managers and policy makers are critical (Gonzalo-Turpin et al. 2008) and require that scientists acknowledge their own value system and openly participate in on-the-ground restoration projects and decision making (Arlettaz et al. 2010). Indeed, effective ecological restoration projects are characterized by community involvement; transfer of knowledge among scientists, practitioners, community members, and administrative organizations; and inclusion of a broad range of stakeholders in the decision-making process (Bernhardt et al. 2007). In contrast, in an extensive review of restoration work published in scientific journals, Aronson et al. (2010) found that 80% of papers did not discuss or analyze direct policy impacts or implications of the restoration work. In this section, I emphasize how information transfer among science, policy, and practice increases successful outcomes in restoration.

5.1. Evaluate Project Outcomes

Although thousands, if not millions, of restoration projects occur annually throughout the globe, knowledge of restoration success is hindered by the general lack of assessment and transfer of information concerning project outcomes (Kondolf et al. 2007, Palmer et al. 2007, Tischew et al. 2010). Without comprehensive project assessment, science will have only a limited ability to inform practice: Projects will move forward without the advantage of past knowledge of what has previously worked, what has not, and how these outcomes have varied across projects. A small investment in networks that facilitate standardized monitoring, information transfer, and evidence-based evaluation (**Figure 3**) will lead to huge returns in planning for future projects.

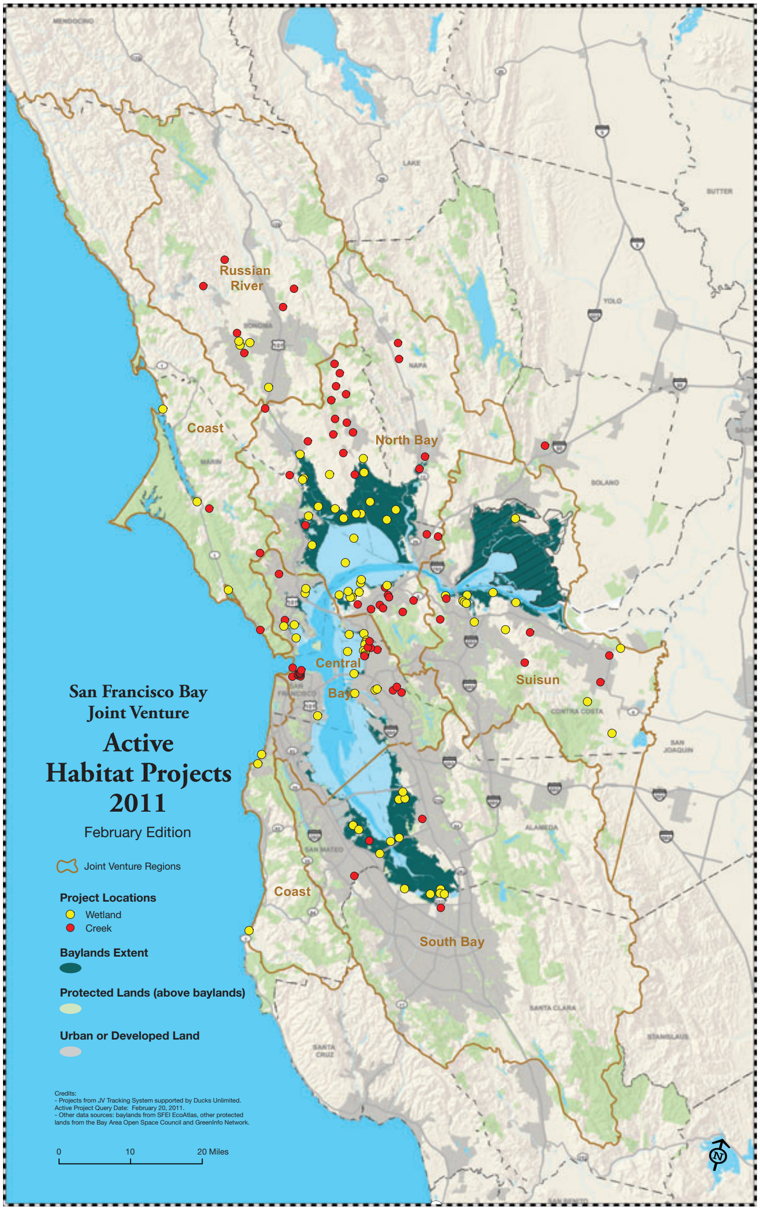

Figure 3

Restoration projects (*red and yellow symbols*) focused on wetland and creek habitats in the San Francisco Bay Area, California. Ecosystem restoration projects such as these are increasingly entering the mainstream political agenda and are growing in numbers and scope. Formation of umbrella organizations such as the San Francisco Bay Joint Venture, which put together this map, is a critical first step coordinating large-scale planning and information transfer. Such organizations serve as clearinghouses to provide monitoring data to inform science so that projects can move forward with the advantage of past knowledge of what has previously worked, what has not, and how these outcomes have varied depending on techniques and environmental context. Information on active projects is continuously updated and linked to a tracking system that provides a current and comprehensive database on restoration, enhancement, and acquisition projects throughout the region (for more information, see **http://www.sfbayjv.org/**).

For instance, with accurate information concerning the likelihood of restoration success, costs, and spatial dependencies, decision-making tools can help prioritize restoration activities (Wilson et al. 2011). This investment in evaluation will also enhance the relevance and applicability of restoration ecology research (Lindenmayer & Likens 2010, Palmer 2009).

Evidence-based practice: decision making based on systematic reviews of evidence on the effectiveness of interventions in environmental management and the environmental impacts of human activities

5.2. Move Beyond Demonstration to Decision-Making Science

Virtually any process that can be demonstrated to affect ecological patterns also can be suggested to be an important consideration in ecological restoration (Weiher 2007). Although demonstrating an effect is an initial step in informing restoration practice, decision making requires prioritization: where and what processes are most important. For example, classic ecological models predict that increased spatial and temporal heterogeneity increase biodiversity (Pacala & Tilman 1994); experiments manipulating heterogeneity have also demonstrated this phenomenon (Baer et al. 2005). In stream restoration, assumptions based on this theory are reflected in the common practices of adding meanders and physical structures such as boulders and artificial riffles to increase habitat heterogeneity. Although it has been largely assumed that these practices increase stream biodiversity, when Palmer et al. (2010) reviewed published studies that examined the response of invertebrate species richness to restoration actions that increased habitat heterogeneity, they found not only that merely one-third of the studies found a positive relationship between increased heterogeneity and diversity, but also that heterogeneity in general was a relatively unimportant factor in controlling diversity in stream restoration. Thus, although theory demonstrates that heterogeneity can increase diversity, many other factors can also affect it, and it is important to invest in restoration measures that address the most influential controls on diversity.

Provision of restoration guidance in an adaptive framework requires prioritization of what and where processes have large effects; this is best achieved through a multivariate approach. In a noteworthy example, Pywell et al. (2007) experimentally investigated the effectiveness of 13 restoration treatments in restoring the plant diversity of two grassland sites in England. Using this approach, the authors recommended restoration strategies that ranged in cost and scale: for instance, turf removal as an effective but costly small-scale option and slot (or drill) seeding as a more variable but less costly larger-scale option. Studies that take a similar approach but evaluate outcomes across sites can provide information about site and landscape characteristics that influence restoration outcomes (e.g., Matthews et al. 2009a).

5.3. Provide Evidence-Based Assessments

Quantitative evidence-based practice reviews are currently rare in restoration research (for exceptions, see Peppin et al. 2010, Stewart et al. 2009) but are crucial for understanding when and why a technique is successful and reducing the possibility of ill-informed practices. Ecological theory can point restoration toward important processes that need manipulation, for instance, lowering nutrient levels, reducing competition from exotic species, or increasing niche partitioning processes that maintain diversity. However, for this information to be relevant, restoration ecology needs to employ evidence-based assessments that both identify practices that can achieve the proximate goal (i.e., lower nutrient levels) and evaluate whether manipulation of these processes results in the intended ultimate goal in terms of ecosystem recovery (i.e., that decreased nutrient levels lead to increased native species diversity).

When evaluated quantitatively, several well-accepted practices achieve their proximate goal but do not achieve their ultimate goal in terms of guiding ecosystem recovery. For instance, because nonnative species can affect native diversity and alter ecosystem processes, their removal is a common goal of restoration efforts (D'Antonio & Meyerson 2002). Although removal often does enhance native diversity and function, the practice illustrates the importance of a critical review of scientific evidence. For instance, in the semiarid southwestern United States, invasion of salt cedar (*Tamarix* spp.), a nonnative large shrub, was associated with water wastage and large biodiversity costs, which prompted millions of dollars to be expended on its eradication (Stromberg et al. 2009). However, research indicates that *Tamarix* can be less problematic than previously assumed and that many of the control actions have unintended negative impacts on native species (Hultine et al. 2010). Similarly, in a survey of the efficiency of nonnative plant species removal projects in Australia, Reid et al. (2009) found that the majority of removal projects reported replacement with other nonnative species.

Research that develops cost-effective techniques that can be applied in large-scale restoration projects is a rare but critical extension of ecological studies. For instance, although nitrogen reduction via carbon addition has been demonstrated to slow rates of nitrogen cycling and reduce exotic plant performance (Perry et al. 2010), small-scale experimental demonstrations often use sugar as the carbon form. Efforts to scale up this technique by using a carbon source more amenable to large-scale application, such as rice straw or wood chips, have had mixed success (e.g., Corbin & D'Antonio 2004). Unless studies can supply a method that is consistently effective at the project scale, application of fundamental research will be limited. To this end, small-scale experimental findings should be validated in the context of large-scale restoration projects.

5.4. Form Partnerships: The Importance of Boundary Spanning

Increasing scientist participation in large-scale restoration projects should be a goal of both scientists and practitioners. Scientists should view project involvement not only as outreach but as an integral part of scientific activity that increases the authenticity and rigor of fundamental research and validates the effectiveness of recommended restoration guidelines (Arlettaz et al. 2010). In addition, it allows unparalleled possibilities for experimentation at the appropriate spatial scale, including inclusion of cost and methodological constraints as well as landscape-level ecological processes that typically go unconsidered when experimentation is conducted at smaller, less realistic scales.

Some training programs—for both scientists and practitioners—are beginning to emphasize spanning science-practice boundaries. At present it is common to train graduate students interested in restoration research in ecological theory and analytical approaches relevant to traditional experimental designs. However, in restoration research, the input of practitioners and policy makers with first-hand experience and in-depth knowledge is crucial if the research is to demonstrate ecological validity. One way to achieve this partnership is to supplement theoretical and analytical training with internships (as is done in other applied contexts, such as medicine, teaching, and engineering). Internships create situations in which faculty, students, and practitioners develop relations that can spawn win-win cycles of knowledge via measurement (faculty and student research) and experience (linking the research to real-world practice) (Bartunek 2007).

Likewise, ecologically based training opportunities exist for restoration professionals and policy makers. Although new professional programs in restoration ecology are under development, their speed of creation does not match the large demand for and current numbers of people practicing aspects of ecological restoration (Nelson et al. 2008). Other fields, such as landscape planning, horticulture, and ecological engineering, have also grown dramatically to meet the demand for training in restoration (Musacchio 2009); these often have little emphasis on ecological science. Short courses and certificate programs would ensure that practitioners have the ecological background they need to execute effective projects.

6. CONCLUSIONS

Given the rapidly growing need for effective restoration and the increasing interest in restoration as an active policy option, restoration ecology is in an exciting but demanding period of maturation. A rough scorecard reveals a 25-year record that is mixed. Recovery can be rapid and straightforward. However, in other circumstances, restoration is largely a gamble that most often results in improvement from degraded conditions but not complete recovery. Although we are far from being able to predict the success of a particular restoration, investigation of dynamics relating to convergence and progression is a promising avenue to indicate critical mechanisms and whether constraints may or may not self-correct over time.

Problems of biodiversity loss and environmental degradation are increasingly entering the mainstream political agenda, and restoration is at the forefront of policy solutions to both improve ecosystem services and ensure ecological resilience against future environmental change. The outlook of these newer paradigms is encouraging, and there is much more to be done concerning the application of these ideas to management decision making. Restoration ecology can inform management decision making and policy development in an evidence-based, adaptive, and experimental framework. It can continue its roots in basic ecological science, extending those discoveries toward challenges such as site prioritization, decision-making tools, and method evaluation. It can help set standardized and measurable success criteria and evaluate investment in restoration efforts at the global, national, and local levels. An era of restoration in ecology is around the corner if we can capitalize on these opportunities.

SUMMARY POINTS

1. With restoration at the forefront of policy solutions to conserve biodiversity, improve ecosystem services, and ensure resilience against future environmental change, the demand for restoration science to inform practice and policy has never been greater.
2. One of the most crucial measures of any applied science is the success of the practice that it informs. However, data are rare because of limited monitoring, limited access to monitoring data, and little consensus on standard evaluation criteria. The measures we have present a mixed picture of success.
3. Restoration results in a wide range of outcomes: It can produce relatively quick and complete recovery; it can be partially successful with some degree of improvement from degraded conditions; or it can yield little or no recovery. The widely variable outcomes in restoration can be grouped by spatial and temporal dynamics to better understand what might lead to, or what might impede, successful restoration. Constraints that do not self-correct over time, for instance owing to changed species pools, land-use legacies, and species feedbacks, require active intervention.
4. Research relevant to ecosystem restoration can go beyond site-specific investigations and be ambitious and wide-ranging. An ecosystem services framework is a promising avenue to link science, economics, and societal values. Restoration of resilience to future change may be a powerful way to integrate management with environmental change.

5. Given the uncertainties of environmental change and complexities of ecosystem dynamics, virtually all restoration strategies have risks. An increased emphasis on the evaluation of restoration practices and project outcomes will ensure that decision makers can base practices and policies on the best available evidence.

FUTURE ISSUES

1. Although positive benefits of restoration can range from education to community involvement, to biodiversity conservation, to provision of ecosystem services, a strong, data-rich case must be made about when and why projects are successful. This will increase capacity to learn from mistakes, to improve practice, and to identify instances in which alternatives to restoration are more effective. Under what conditions is investment in ecosystem restoration most appropriate? Most risky? What are the best practices given these contextual constraints?
2. As the return to historical reference conditions is becoming more and more intractable as a benchmark for restoration success, what benchmarks can be set? Given uncertainties about future environmental conditions, what approaches best enable recovery or maintenance of an ecosystem's adaptive capacity and resilience?

DISCLOSURE STATEMENT

The author is not aware of any affiliations, memberships, funding, or financial holdings that might be perceived as affecting the objectivity of this review.

ACKNOWLEDGMENTS

Ideas in this paper reflect beneficial discussions with Scott Collins, Valerie Eviner, Katherine Gross, Richard Hobbs, and Stephen Swallow, as well as collaborators in the Albany Restoration Workshop and students in the Restoration Ecology Graduate Seminar at UC Berkeley. I thank Megan Lulow, Jessie Olson, Kristine Preston, and Trish Smith for their many valuable insights into the practice of restoration and Lauren Hallett, Liana Nichols, Seema Mangla, Jessie Olson, and Claudia Stein for helpful comments on previous versions of the paper. This work was supported by grants from NSF (DEB-1001807 and DEB-0614168) and USDA (NIFA 2010-85320-20502).

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Annual Review of Ecology, Evolution, and Systematics

Volume 42, 2011

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