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Intervention Ecology: Applying Ecological Science in the Twenty-first Century

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Rapid, extensive, and ongoing environmental change increasingly demands that humans intervene in ecosystems to maintain or restore ecosystem services and biodiversity. At the same time, the basic principles and tenets of restoration ecology and conservation biology are being debated and reshaped. Escalating global change is resulting in widespread no-analogue environments and novel ecosystems that render traditional goals unachievable. Policymakers and the general public, however, have embraced restoration without an understanding of its limitations, which has led to perverse policy outcomes. Therefore, a new ecology, free of pre- and misconceptions and directed toward meaningful interventions, is needed. Interventions include altering the biotic and abiotic structures and processes within ecosystems and changing social and policy settings. Interventions can be aimed at leverage points, both within ecosystems and in the broader social system—particularly, feedback loops that either maintain a particular state or precipitate a rapid change from one state to another.

Keywords: intervention, restoration ecology, offsets, leverage points, ecosystem services

In a recent editorial in *Science*, Roberts and colleagues (2009) concluded that "our planet's future may depend on the maturation of the young discipline of ecological restoration." Restoration is now seen as a key element in achieving conservation and natural-resource management goals, and small- and large-scale restoration activities are increasingly common worldwide. And yet, even as activity and interest in the discipline grow, so does the extent to which its basic principles and tenets are debated and reshaped, largely in response to the growing recognition that restoration has to be conducted in a context of rapid and ongoing environmental change. In this article, we examine the ways in which these basic tenets are being reshaped and propose that, although the field is in the process of maturing, the young discipline of restoration ecology may also need to metamorphose into something related but different. Indeed, we suggest that restoration ecology, along with conservation biology and related fields, is actually a subset of a broader enterprise that can be called *intervention ecology*. There is a strong need for the development of a more effective ecology to enable the analysis and management of ecosystems in a rapidly changing world. Intervention ecology covers a wide range of active interventions in ecosystem dynamics that are increasingly required in order to ensure the continuation of ecosystem service provision (Daily et al. 2009) and biodiversity conservation (Janzen 1998, Ehrlich and Pringle 2008, Cole and Yung 2010, Hobbs et al. 2010).

The term *intervention* has been more frequently used both in ecology and in the broader field of environmental science over the past few years. For instance, a search for the topic *intervention* in the Thomson Reuters (formerly ISI) Web of Science's ecology category showed that use of the term increased from no papers in 2004 to 70 papers in 2009, whereas a similar search in the environmental science and environmental studies categories showed an increase from no papers in 2004 to 290 papers in 2009 (*http://isiknowledge.com*). This increase matches an equivalent surge in the use of the term in medical publications. An increasingly prominent focus on intervention perhaps indicates a progression in science from describing and attempting to understand ecosystem structure and dynamics to developing approaches that allow informed intervention in ecosystems (e.g., Pickett et al. 2009).

Should we move on from restoration?

All ecologists aim in their research at understanding how the world works, and most understand that the world is in ever-greater flux. Those scientists who wish to alter the function of ecosystems to enhance the services they deliver (including, especially, maintenance of a great diversity of organisms) should be explicit about both their goals and the possibilities of achieving them. We should not, for instance, give the impression that ways can be found to either hold or turn back the clock and preserve or recreate imagined Edens—an impression often still implicit in conservation policy and management (Hobbs et al. 2010).

Restoration has gained considerable attention and policy management traction over the past two decades. Consequently, early problems with the terminology of the

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field now have the potential to create unrealistic expectations and perverse policy outcomes. Restoration remains an important social and ecological practice but has been dogged by terminological issues focused on what is and what is not restoration (Hobbs and Norton 1996) and by philosophical issues concerning the definition of good restoration (Higgs 1997) and whether nature can ever be restored (Katz 1992, Elliott 2000). Above all, the term *restoration* evokes for the layperson the increasingly untenable notion that an ecosystem can be returned to some previous state and raises the subsidiary question of the date of the original condition. There are many other allied "re-" words that are used in different ways to cover different types of enterprise and aspiration. For example, *rehabilitation*, *recreation*, *reclamation*, and *revegetation* are all terms used to describe varying types or degrees of restoration activity (Clewell and Aronson 2007). Ecological engineering also has ecosystem restoration as a goal (Mitsch and Jørgensen 2003, 2004). Some commentators wish to reserve the term *restoration* for the effort to fully return a system to some preexisting condition and to use other terms for activities that do not have this aim (McDonald 2009). In recent textbooks, the difficulties inherent in focusing on past systems as targets for restoration are recognized (van Andel and Aronson 2006, Clewell and Aronson 2007), but the differences between restoration per se and other activities such as rehabilitation are still discussed in detail, and *restoration* is distinguished from *ecosystem management* and *conservation*. However, the terminology remains confusing and inconsistently used. Although we do not wish to add to the ecological verbiage (Hobbs and Norton 1996), we feel that a more overarching approach is now appropriate—one that is focused on how humans intervene in ecosystems, either to maintain or to repair them. When one recognizes that restoration sits within a broader framework of intervention, many terminological problems evaporate.

Humanity is already intervening in ecosystem function on a planetary scale (Ehrlich and Ehrlich 2008). The central question is whether our interventions can be planned and guided in a way that is beneficial both to humanity and to other organisms. In a recent book entitled *Living through the End of Nature*, Wapner (2010) suggested that "many view the end of nature as an inevitable result of age-old human intervention into nature, and contend that whether we like it or not, we must now rise to the level of responsibility that taking over nature entails."

The arguments for moving on from—or at least expanding the scope of—restoration are both theoretical and practical and are directed at past and future ecosystem change and the moral hazards underlying the presumption that complete restoration is possible.

Ecosystems in flux

Because ecosystems are not static, the term *restoration* entails the open question of *when* to restore to—that is, to what historical state do we wish to restore the system? A "when?" decision is usually hard to justify. The world has changed a lot in the past (through human- and non-human-controlled processes) and will probably change even more in the future; therefore, we need to move on from the notion that we can restore to a previous static state. An idea underlying much conservation and restoration activity, and indeed society's overall relationship with nature, is that some past ecosystem states had characteristics more desirable than those of the present ones (e.g., Eisenberg 1998). However, we may not know the character of the historic ecosystem in any detail. History is always contingent on current knowledge and understanding and is interpreted through current cultural and scientific norms (Carr 2008). Although methodology is improving (Jackson and Hobbs 2009), in many instances we are left with incomplete descriptions of the ecosystem at a particular time, without detailed information on the underlying dynamics. Even for relatively recent disturbances, there may be little information on the predisturbance state, and nearby undisturbed systems are often assumed to approximate this state. Therefore, although ecologists often feel the need to "restore" a system, it is nonetheless difficult to decide what the goal of restoration should be. Although this issue has been debated in the past (e.g., Pickett and Parker 1994, Aronson et al. 1995), it still haunts restoration ecology. Recent debate has been focused on how far we could or should consider turning the clock back (e.g., Donlan et al. 2005, 2006). With the requisite knowledge and capability, to what state might one restore a particular ecosystem, such as Yellowstone: to the condition it was in at the beginning of the ice ages, at the end of the Pleistocene, when human beings first saw it, when Europeans arrived, when it became the world's first national park? And how would ecologists determine what those earlier states were like?

Any restored system would probably be one that was modified and supported to some extent by human action. Purely "natural" systems have been rare for a long time; people have dominated many parts of Earth for thousands of years (Mithen 2003, Mann 2005). Debate continues over both the extent and the intensity of human activity in different regions, and it is likely that effects varied spatially and temporally (Vale 2002). However, for many parts of the world, in aiming to restore to a "pristine" or "natural" state, managers both ignore prior human impact and deny indigenous human societies their rightful place as effective ecosystem managers. (Although the pervasiveness of indigenous human management is still debated, evidence is accumulating from many parts of the world that the effects of such management were extensive, as was discussed by Mann [2005] and as has been illustrated, for instance, in Australia [Head 2000] and California [Anderson 2005].) Aiming to restore a "pristine" state also further exacerbates the human–nature dualism that has resulted in our current environmental mess.

In addition, the rate of change in many systems has escalated in recent times. The world is changing at an ever-increasing and unprecedented rate and in multiple ways (Steffen et al. 2004). Climate change, loss of biodiversity, nitrogen deposition, land-use change, invasive species, release of toxic chemicals, resource exploitation, and many other parameters act synergistically to push the planet in directions never before experienced in human history. The results are no-analogue environments and novel ecosystems and species combinations (Williams and Jackson 2007, Hobbs et al. 2009). Therefore, returning a system to even a semblance of a historic state is and will continue to be difficult. Even if the disturbance after which we wish to restore the community had not happened, the community is still likely to have moved on because of these external factors. Consequently, we should intervene with an eye to the future and toward managing for future change.

Moral hazards

Given the current state of the science, the term *restoration*, taken literally, offers false promise. Although one view is that humanity has a moral responsibility to try to restore damaged areas, there is also a moral hazard in promising to do the impossible (e.g., France 2008). There are numerous myths in restoration ecology that give rise to false expectations of what is possible. These myths have been discussed in detail by Hilderbrand and colleagues (2005), who characterized them as simplified and potentially misguided models for understanding and application. These models include ideas such as community assembly always being predictable, the existence of a single end point, and that fixing physicochemical conditions will allow biotic reassembly. False expectations arise partially because of an overselling of what restoration can do by some and partially because of a misunderstanding of the complexity and dynamics of the ecosystems being managed or restored. As was discussed above, it is obviously not possible to fully replicate a complex and diverse ecosystem—to paraphrase an old saying, you can't step in the same ecosystem twice. Although many restoration ecologists recognize this problem either explicitly or implicitly, the message has not been promulgated effectively. Many people, and importantly, many policymakers, believe that it is possible (and desirable) to bring a system back to its predisturbance state. For instance, current US National Park Service (2006) policy is that "the Service will seek to return such disturbed areas to the natural conditions and processes characteristic of the ecological zone in which the damaged resources are situated."

The belief that complex ecosystems are fully restorable opens the door to trade-off schemes between development and restoration and may lead to loss of high-value conservation areas. In particular, the expectation that systems can be restored underlies much policy on offsets and mitigation (Brooks et al. 2005, Gibbons and Lindenmayer 2007). Concern has been expressed for some time about the effectiveness of mitigation and offsets in maintaining biodiversity values (Roberts 1993, Race and Fonseca 1996). For instance, wetlands are one of the main ecosystem types for which mitigation is often used, and Turner and colleagues (2001) found that only 21% of wetland mitigation sites met tests of ecological equivalency to lost wetlands. These concerns remained largely unchanged in more recent evaluations (Burgin 2008, BenDor et al. 2009, Moilanen et al. 2009). A societal expectation that degradation can happen and can be either reversed *in situ* afterwards or offset by restoration somewhere else inevitably results in high-value areas being traded for restored areas elsewhere, with an overall net loss of biodiversity or ecosystem service value. A recent example is the loss of 30 hectares (ha) of good-condition *Banksia* woodland within the Perth urban area of western Australia, resulting from the construction of a new hospital. This loss is to be offset by "restoration and revegetation" of a total of 50 ha of degraded woodland across six different nearby sites and the conservation purchase of a 41-ha site 80 kilometers away in an entirely different ecosystem type (*www.fionastanley.health.wa.gov.au*). This decision will result in the replacement of one of the largest remaining remnant patches within the central urban area with a number of smaller, partially restored areas and the protection of an area not under any immediate threat.

A further hazard relates to the false dichotomy that persists between restoration and conservation, which are often seen as separate enterprises conducted with different foci, by different people, and for different reasons (Young 2000, Noss et al. 2006). There has been a false perception that restoration requires action, whereas traditional preservation-focused conservation is mainly a passive attempt to retain an existing ecosystem or assemblage (Hall 2005). However, the links between the two endeavors are already recognized (Dobson et al. 1997, Young 2000), and neither is sufficient on its own (Rosenzweig 2003). Conservation faces the same set of challenges as restoration, and old ways of working have become less appropriate (Heller and Zavaleta 2009). Because the world is changing, conservation more and more commonly requires active management, blurring the lines between it and restoration.

What is intervention?

A need to focus both on an uncertain past and on a more uncertain future has created an apparent paradox for restoration ecology. Recent attempts to deal with these problems include reframing restoration with a future rather than a historical focus (Aronson and van Andel 2006, Choi 2007, Choi et al. 2008) and the wise use of history to guide both the retention of historic systems where it is possible and the development of new systems where it is necessary (Jackson and Hobbs 2009). Most practitioners and researchers, however, acknowledge the need for intervention to achieve whatever goals are set. In a similar way, in conservation biology the challenge is to move the focus from preserving existing species and assemblages within particular designated places, such as reserves and parks, to considering how to conserve systems that are temporally and spatially dynamic. This challenge increasingly requires that answers be sought to questions of whether, how, when, and why intervention is necessary.

As Matthews and Turner (2009) pointed out, humans have long undertaken ecological interventions, usually in response to particular environmental problems. Interventions can take the form of manipulating the biotic or abiotic characteristics of the ecosystem and can vary in intensity from deliberate nonintervention through directed one-off interventions to ongoing, large-scale interventions (Hobbs and Cramer 2008). Examples of these interventions include local activities, such as fencing an area of vegetation to exclude livestock, removing problem weed species from a local preserve, or reducing pollutant inflow to a wetland. At larger scales, reinstatement of historic fire regimes, extensive revegetation to increase landscape permeability, and reinstatement of flow regimes in river systems are all interventions aimed at maintaining or repairing ecosystem services, including the conservation of biodiversity. Interventions are intended either to maintain a system in a current desirable state or to move a system away from a current undesirable state. The former type of intervention would normally be couched in the rubric of conservation or ecosystem management—and maintaining desirable states and assemblages is often the highest priority for conservation—whereas the latter type would fall under the restoration terminology discussed above.

Interventions can also be categorized as reactive, active, or proactive and can occur primarily at a local, regional, or global scale (table 1). Reactive interventions are attempts to maintain a current ecosystem state or to halt a process thought to degrade ecosystem values. Active interventions are positive steps taken to change ecosystem properties in a particular direction. Proactive interventions are designed to limit the human drivers of processes that assault ecosystems. These intervention types cover the restoration–management–conservation spectrum while avoiding the past-focused aspects of restoration and conservation. As with all categorizations, there is overlap among the terms. Because of the complex nature of the material being covered, it is perhaps to be expected that the categories blur together. Most active and proactive interventions are still carried out in reaction to perceived problems or threats, whether those threats exist or are predicted. For instance, an active intervention at the local scale is likely to form part of a reactive response to a broader degradation process. However, the intent of the intervention is important. Revegetating a burned slope is the same action whether it is done with the intent to increase native plant populations or with the intent to halt erosion, but with the former intent it is active intervention, and with the latter it is reactive. A categorization based on intention forces a clear consideration of goals up front, rather than actions based on preconceptions or a failure to clarify and agree on goals at all (Hobbs 2007).

A useful question to consider in the context of intervention in ecosystems is whether leverage points can be identified—places to intervene in the system where a small change could lead to a large shift in behavior (Meadows 2008). Useful leverage points include key elements and flows within the ecosystem, balancing-feedback loops that act to stabilize the system, and reinforcing-feedback loops that lead to rapid ecosystem change (figure 1a). Reactive management tends to be focused on the system components per se, rather than on processes or feedbacks, whereas active management tends to be focused more on processes, flows, and feedbacks. Importantly, however, interventions may also include altering policy and broader socioeconomic settings (Chapin et al. 2006). As well as internal system properties, Meadows (2008) suggested that the most effective leverage points may lie in the information transfers, rules, and paradigms constructed around the system. This area is where there is the greatest opportunity for proactive interventions. Changing rules and governance approaches may have a much more profound effect than tinkering with ecosystem properties per se. Restoration ecology is traditionally focused on the local system being restored, rather than on the broader socioeconomic and political settings, although there have been recent attempts to fuse the two (Aronson et al. 2007). Both system properties and governance also need to be considered at multiple scales, broadening the traditional focus from the local system to include regional and global scales (figure 2). This broadening

Table 1. Examples of different types of intervention at local, regional, and global scales.

Scale of intervention	Type of intervention: Reactive	Active	Proactive
Local	Removal of invasive species within a reserve	Postmining revegetation	Ecologically based urban planning codes
	Toxic site remediation	Provision of key habitat structures	Assisted migration in anticipation of climate change
	Dam removal	Reintroduction of key species	
Regional	Regional reduction of nitrogen inputs from agriculture	Large-scale reforestation	Implementing conservation networks at regional scales
	Wetland mitigation schemes	Reinstatement of regional water flows	Strengthening biodiversity legislation
		Modification of regional fire regimes	Establishing marine protected areas
Global	Convention on International Trade in Endangered Species	Global agreements on carbon emissions	Trade agreements to restrict the movement of invasive species
		Identification of biodiversity hotspots	

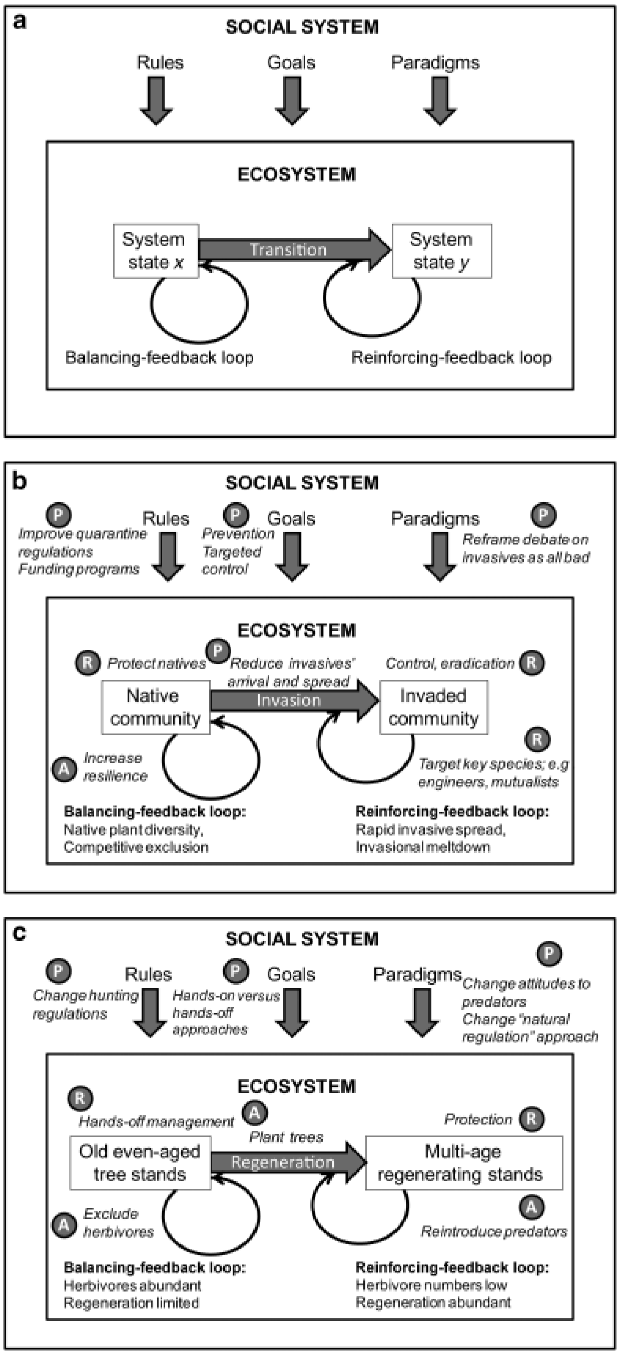

Figure 1. (a) A simplified system diagram depicting an ecosystem, with different states, transitions, and feedback loops, embedded in a social system within which rules (e.g., legislation), goals, and overall paradigms are brought to bear on the ecosystem. There may be potential leverage points, where intervention can produce a large change in system behavior, associated with each system characteristic. Examples of particular issues are given in (b) and (c). Here, the system characteristics are identified in normal text, and possible interventions are indicated in italics and categorized as reactive (R), active (A), or proactive (P). (b) Spread of invasive species in an ecosystem. (c) Lack of tree regeneration in boreal systems.

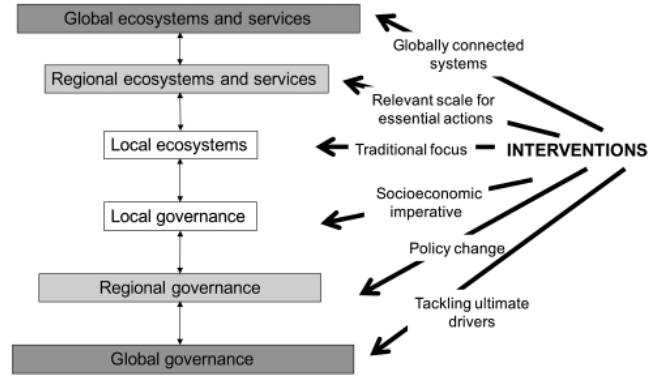

Figure 2. Using the framework developed by Carpenter and colleagues (2009), interventions are needed in both ecosystems and governance systems. The traditional focus of management intervention is the local ecosystem, and this remains the most common scale at which interventions occur. However, it is increasingly recognized that local interventions have to be conducted within a broader landscape and regional context, and that regional interventions may be required to address many issues such as hydrologic imbalance, connectivity, and maintenance of key ecosystem services. Furthermore, the interconnectedness of local and regional systems within a global context is also increasingly relevant in the context of climate change, invasive species movement, and so on. In the context of governance, there is now clearer recognition of the need to consider the socioeconomic settings within which interventions have to take place, at local, regional, and global scales. These settings often determine what interventions are possible or socially desirable at the local scale. Regional governance similarly sets the broader policy and legislative context, and interventions aimed at changing this context can significantly change the feasibility or priority accorded to ecosystem-level interventions. Finally, global governance is perhaps the hardest arena in which to intervene effectively, but the one in which we could most effectively confront the ultimate drivers of many threats and environmental problems such as climate change and geopolitical stability.

of focus suggests both that ecological aspects need to be considered in a wider socioeconomic context and that an interdisciplinary or even transdisciplinary approach is needed.

On one hand, clever use of leverage points could greatly enhance humanity's ability to manage ecosystems effectively in a rapidly changing world, but on the other hand, inappropriate application could make a difficult situation much worse. Careful identification of the most effective interventions requires that we achieve enhanced understanding of which system components and flows are key and where feedback loops might be in play (figure 1b, 1c). Possible interventions are indicated in these figures and categorized as proactive, active, or reactive. Interventions are focused on actions that alter the components of the system (biotic, physical, or social) that in turn change the status of the system in one way or another. Feedback loops can either balance a system in its current state or hasten its transition to an alternative state (figure 1a). When the current state is deemed desirable, management focus will be on balancing feedbacks and avoiding runaway reinforcing loops, whereas when the current state is considered degraded or undesirable, the aim may be to reduce the effect of balancing loops that maintain the current state while searching for reinforcing loops that can quickly drive the system to a more desirable state. For example, imagine a plant community in which an exotic species has recently been introduced (figure 1b). If the community is intact, its current composition may be maintained by balancing feedbacks, such as high plant diversity and competitive exclusion, that slow the spread of the invasive species. In a disturbed or fragmented community, balancing feedbacks may become weaker. As the invasive species establishes in the community, its spread and effects can escalate by way of reinforcing feedbacks, such as exponential population growth and invasional meltdown (Simberloff and Von Holle 1999). In this case, a balancing-feedback loop maintains the native system relatively intact, whereas the initiation of a reinforcing-feedback loop results in runaway degradation as a result of invasion.

In comparison, consider a situation in which overgrazing is thought to have placed a forest system in a degraded state with no regeneration (figure 1c). In this case, a balancing-feedback loop maintains the system in a degraded state (i.e., lacking tree regeneration). Status quo management results in the maintenance of a degraded state, and active interventions are required to reestablish tree cover. Standard management approaches such as planting seedlings are unlikely to succeed if the key issue is overgrazing by deer or other ungulates. In this case, fencing is a possibility, but more systemic treatment of the problem may be achieved by the reintroduction of predators, such as wolves, to reduce herbivore levels (Beschta and Ripple 2010) and set up a reinforcing-feedback loop. This results in predator control of herbivores and enhanced tree regeneration.

Judicious management of balancing-feedback loops and the recognition of, and early intervention in, reinforcing-feedback loops may represent managers' main hope of being able to prevent rapid deleterious shifts in ecosystem state or to force beneficial shifts (Suding and Hobbs 2009). In both examples, interventions focused on the socioeconomic and policy contexts may, in fact, be more effective than interventions in the ecosystem itself or may be a prerequisite for enabling effective ecosystem interventions. In both hypothetical examples discussed above, changes in policy and regulations can significantly alter the parameters under which the actual ecosystem can operate: In the case of invasive species, for instance, tightening quarantine and trade regulations involving species transportation can significantly reduce the risk of problem species being introduced. Changing management goals as a result of new information or an altered understanding of system dynamics can also have a profound impact on the ecosystem. For instance, in the case of herbivore overabundance, hands-off management in a preserve may be replaced by an approach in which herbivore control is acceptable. The implementation of this goal may be intimately connected to shifting paradigms and philosophies. For instance, a switch to acceptance of some degree of nonnativeness in an ecosystem will reduce the management imperative to eradicate or control all nonnative species (figure 1b). Shifting from a predator-eradication paradigm to the recognition of predators as important system components with cascading impacts on the system as a whole (Terborgh and Estes 2010) similarly alters the goals and approaches likely to be used.

Although the examples presented above primarily involve local ecosystem management, albeit set within the broader socioeconomic context, the same approach can be taken to larger-scale issues. A recent prominent example is the *Deepwater Horizon* oil spill in the Gulf of Mexico, which resulted in major environmental pollution, ecological damage, and socioeconomic hardship. Clearly, interventions in this situation form a continuum from immediate reactive interventions to longer-term active interventions. Reactive interventions are required in order to stop the flow of oil from the ruptured well: This intervention is more engineering than ecology. Further reactive interventions are required in order to limit the oil's damage to sensitive ecosystems, and these may be accompanied or followed by active interventions to attempt to repair the damage caused by the oil and to reinstate aquatic systems, fishery enterprises, and other human social and economic activities disrupted by the spill. Further proactive interventions could be envisaged that would be aimed at preventing such environmental damage from happening again; these proactive interventions could include rebuilding barrier islands and other coastal habitat, as is already advocated for the prevention of storm damage (LCPRA 2007). Finally, proactive interventions in the broader socioeconomic sphere could be aimed at reducing society's oil dependence and fossil fuel use, thereby eliminating the need for ever-more-risky resource-development projects.

Problems with intervention

We recognize that proposing intervention ecology as an approach is not a panacea. There are counterarguments to all of the points we have raised above. In particular, the concept of intervention implies no specific management goal. Intervention could send the system in any direction; it would not necessarily return the system to or maintain the system at some preferred state. This result could be seen as problematic, particularly in that it might "lower the bar" for management goals. However, this lack of de facto goals could and should give managers and society in general impetus to develop clear goals for any intervention plan with open discussion on the desirability of alternatives.

Perhaps more importantly, the term *intervention* is itself loaded and has militaristic overtones as a result of recent popular usage. It is certainly not the nurturing term that *restoration* is, and it is hardly likely to engage communities in ecosystem management in the way *restoration* does. It seems unlikely that a community group would label a site being manipulated with "Intervention in Progress" (although this would transmit a more realistic message). Maybe this connotation alone indicates that it would be wrong to advocate doing away with the idea of restoration altogether and that we should instead accept that it is one particular type of intervention that fosters community engagement with nature (while, we hope, helping community members recognize the reality of what that nature is).

There is always uncertainty about the risks of interventions (Matthews and Turner 2009), but the degree of uncertainty is very variable within and often very different among the three categories of intervention. Active interventions are generally likely to be relatively low risk: If bluebirds do not use the nesting boxes designed to help save a dwindling population or fish do not thrive in a ship sunk as a reef base, at least no serious damage has been done to an ecosystem. Activities that are intended simply to make human-disturbed areas more hospitable to native biodiversity (Şekercioğlu et al. 2007, Ranganathan et al. 2008) seem the least risky of all. If an attempt to reestablish specimen rain-forest trees or an entire stand of tropical forest on a degraded pasture results in fewer specimens or a less-than-ideal mix of trees, the forest would still support more biodiversity than the pasture, and the downside would be only in terms of wasted financial and physical effort. Reactive intervention often presents the most difficult scientific and ethical problems. For instance, introduction of a biological control agent involves trying to determine what effects the agent will have on the target organism and the probability that it will have unanticipated negative effects on the ecosystem. Controlled or prescribed fire is another type of intervention with great potential risk of adverse outcomes: Reinstating fire in a system is often necessary where long periods of fire suppression have occurred, but the risk of burning the high fuel loads resulting from suppression is often extreme (Carle 2002, Arno and Fiedler 2005). The risk of failure or even relative disaster with some reactive interventions may be very high, but that risk may be judged acceptable if the problem to be dealt with is sufficiently serious. Successes get very good press, whereas failures may mostly concern ecologists (Louda et al. 1997). Proactive interventions, such as setting up a marine reserve or a family-planning program, seem unlikely to cause negative ecosystem effects, although perverse outcomes are always possible.

Perverse outcomes are a risk with any type of intervention and even with not intervening at all. Potential or actual perverse outcomes of recent examples of ecological interventions include the potential for unintended ecological, hydrological, and socioeconomic consequences resulting from the use of biological control on *Tamarix* in the western United States (Hultine et al. 2010); an increase in nonindigenous species introductions through the use of constructed reefs intended to restore habitat (Sheehy and Vik 2010); and negative off-site impacts of riparian restoration in California (Buckley and Crone 2008). In a policy context, examples include (a) potential increases in the threat from poaching of vicuña in South America brought about by a change in policy regarding trade in their fiber products (McAllister et al. 2009); (b) the erosion of the natural capital of coastal regions, which could eliminate existing landscape protection from intense wind and waves, instigated by the development policies and practices in the United States (Bagstad et al. 2007); (c) potential unintended outcomes from payments for reduced emissions from deforestation and degradation (Venter et al. 2010); and (d) unintended and undesirable consequences of a move toward market-based approaches to wildlife conservation in Mexico (Sisk et al. 2007).

Thoughtful intervention: Hubris versus humility

The risks associated with interventions of various sorts have to be balanced against the need to act and the consequences of not doing so. Do we know enough to intervene? Leverage points are often counterintuitive, and human attempts to intervene in systems can inadvertently send them in the opposite direction of what is desired (Meadows 2008). The complexity of social and ecological systems means that perverse outcomes often arise from well-meant actions. This problem highlights the need for a certain degree of circumspection in humanity's aspirations to intervene effectively in ecosystems. How much hubris is behind this enterprise, and how much does this hubris need to be balanced by humility? Do we know enough to carefully and effectively intervene when and where it is necessary? Will we ever know enough?

Let us return to the discussion of the Gulf of Mexico oil spill: President Obama called for "a comprehensive assessment of post-spill recovery needs, as well as a plan to provide integrated federal assistance for longer-term restoration and recovery" (Eilperin 2010). However, the *Washington Post* correspondent covering the story of the planned restoration recognized that "the challenges of designing a restoration plan that does more than correct for the spill's impact are as vast and complex as the delta it would aim to revitalize, balancing environmental goals with commercial uses" (Eilperin 2010). Restoration of the Mississippi Delta and the Gulf to a historic state is rendered both unlikely and unfeasible by the massive changes brought about by levees and channelization, the loss of wetlands and barrier islands, and nutrient run-off from the US agricultural heartland and the resulting dead zone in the Gulf (Day et al. 2007, Hufnagl-Eichiner et al. 2010). Challenges abound for ecologists in this arena, from local management of ecosystems affected by the oil spill to larger-scale management of river catchments and ecological flows. Even presidential decree is unlikely to result in effective restoration: hence the pressing need to decide which interventions are feasible, desirable, and likely to result in positive outcomes for humanity and biodiversity. Using this approach could help avoid undertaking apparently sensible short-term actions that ultimately either do little to help or result in adverse outcomes. For instance, a major restoration response to the oil spill was the initiation of extensive and costly sand barriers or berms off the Louisiana coast to limit the onshore flow of oil. However, from geomorphologic and ecological perspectives, these berms are likely both to be ineffective in the long term and to have adverse impacts on the ecosystems they are supposed to protect (Froede 2007, Lavoie et al. 2010). This likelihood reinforces the need for a thoughtful experimental approach embedded in adaptive management, rather than all-out, knee-jerk responses. Intervention ecology can be thought of as the science of meaningful and thoughtful intervention in ecosystems—an extension of the ideals of restoration but not a call for intervening for intervention's sake. Wapner (2010) perhaps put it best: "The end of nature changes our historical role on earth to the degree that it calls on us to consciously take hold of the steering wheel of life, and become intelligent, compassionate, and otherwise mindful managers of the planet—quite a daunting challenge."

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