The Dynamic Regime Concept for Ecosystem Management and Restoration

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Because the response of ecosystem patterns and processes to disturbance is rarely linear, the dynamic regime concept offers a more realistic construct than linear models for understanding ecosystems. Dynamic regimes, and shifts between them, have been reported for a diversity of ecosystem types (e.g., terrestrial, marine, aquatic) at a variety of scales (e.g., from small lakes to the global climate). Ecosystem regimes that are obvious at one scale may not be at another. Regimes are maintained by internal relationships and feedbacks between species, and these internal dynamics can interact with large-scale external forces (such as global weather patterns) and trigger shifts to alternative regimes. The dynamic regime concept is commonly used in ecosystem management, restoration, and sustainability efforts, in what are known as "state-and-transition," "threshold," or "alternative stable state" models. Here we review the application of this concept to ecosystem management and restoration, and discuss how dynamic processes at multiple scales can affect this application.

Keywords: dynamic regimes, ecosystem management, scale, alternative stable state, nonlinear

The identification of dynamic regimes is a research frontier in many fields that deal with nonlinear systems. Dynamic regimes, as defined by systems theory, are stable basins of attraction in a state space, in which the attraction is formed by internal relationships between species and their environment. Ecosystems can have many possible regimes (e.g., a lake may either be crystal clear or turbid and prone to algal blooms), and the size and shape of those regimes are primarily dictated by internal relationships; changes in these relationships can trigger regime shifts (Scheffer and Carpenter 2003). External forces, which influence ecosystems but are not directly influenced by them, can also push ecosystems between regimes, often suddenly. The increasingly common use of the dynamic regime concept has highlighted urgent ecological and environmental issues, particularly the possibility of a rapid (2- to 4-decade), catastrophic change in global climate conditions and its potential effect on ecosystems and human societies (Alley et al. 2003, Schmitz et al. 2003). Here we explore the use of the dynamic regime concept for ecosystem management and restoration, including the ways that its use is influenced by ecosystem processes at different scales.

Nonlinear behavior and the dynamic regime concept

Nonlinear behavior in populations, communities, and ecosystems is widespread and occurs at many scales. Despite its ubiquity, there is little consistency in the terminology that has been developed to describe it. The term *equilibrium,* for example, was prevalent when ecosystems were still viewed as fairly static systems. An equilibrium, in this sense, refers to a community type to which an ecosystem returns after each disturbance (e.g., a conifer forest that regrows after a series of postfire successional stages). The phrase *multiple equilibria* conveys the newer view that ecosystems can return to a different community type after disturbance, depending on the type and intensity of disturbance and on the conditions of the predisturbance system. However, the use of *equilibria* to describe ecosystem processes is viewed by some as misleading, as the word implies a static character that ecosystems do not have. Therefore, the term *stable state* has been advocated to convey more clearly the dynamic nature of postdisturbance community types, because populations and species assemblages still fluctuate even in mature communities. Other terms that have been substituted for *stable state* include *stability domain* (Gunderson et al. 2002), *metastable state* (Leuven and Poudevigne 2002), and *basin of attraction* (Scheffer et al. 2001).

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Ecosystem ecologists use the phrase *alternative stable states* to indicate the potential for several different community types after a disturbance. However, population and community ecologists use the same term to describe a slightly different set of system dynamics. In the latter usage, populations of species interact with one another in such a way as to make possible several different, yet stable, sets of coexisting species, even under the same environmental conditions (Beisner et al. 2003a). To minimize confusion, Scheffer and Carpenter (2003) advocate using the phrase *dynamic regime* to discuss the effects of disturbances or environmental change on ecosystems. We follow their convention and avoid the phrase *alternative stable states.*

Dynamic regimes of ecosystems are often characterized by the dominance of particular species, trophic structures, energy flows, and internal feedbacks that maintain biotic and abiotic patterns within a distinctive range (Scheffer et al. 2001, Gunderson et al. 2002). Statistically, dynamic regimes are described as regions of a multidimensional state space in which the state variables (e.g., population sizes, nutrient loads) exhibit characteristic behaviors and quantities. Ecosystems are maintained within a regime through the internal dynamics of variables, such as the interaction between populations of species that coexist in an ecosystem. External disturbances, changes in internal relationships between species, or a combination of both can trigger a shift from one regime to another (Dent et al. 2002, Scheffer and Carpenter 2003). A change in regime then leads to a reorganization of the dynamic relationships and feedbacks among the internal variables in accordance with the new regime (Sazykina et al. 2000). The loss or change of variables that are critical to the characteristic properties of a particular regime (e.g., the extinction of keystone species) can negate the possibility of return to some regimes (i.e., some regimes may no longer exist; Beisner et al. 2003a).

In practice, identification of dynamic regimes is complicated, because the boundaries between regimes are not static, and hystereses can be present (i.e., the conditions under which an ecosystem shifted to one regime can be different from those in which the system will shift back; figure 1). Regime boundaries, and indeed the existence of regimes, can change over time because of ecosystem succession (the progression of community types after a disturbance from early to mature stages), changes in environmental conditions, and the continuing evolution or extinction of the species that are already present. The size of the regimes also influences ecosystem *resilience* and *stability,* which refer to the size or intensity of disturbance the system can withstand without changing and the speed at which the system recovers (Gunderson et al. 2002). Regimes formed by strong internal feedbacks may persist under much larger or more intense disturbances than regimes formed by less cohesive relationships (Suding et al. 2004).

When an ecosystem shifts from one regime to another, often many of the internal variables change very rapidly. Nonlinear behavior in an ecosystem could be a direct response to a nonlinear change in an external force (e.g., a sudden pulse of stormwater to a stream from a brief but heavy thunderstorm). Internal variables may respond in kind to this nonlinear change (e.g., the dissolved oxygen and turbidity of the streamwater may increase nonlinearly in response to the large pulse of stormwater). Even under gradually changing external variables, internal variables can respond nonlinearly but with no fundamental change in internal relationships or feedbacks. For example, the amount of impervious surface (e.g., roof, pavement) in a watershed affects streams through the quantity and quality of runoff the streams receive (Paul and Meyer 2001). High levels of impervious surface cover lead to large amounts of runoff, carrying high loads of sediment, nutrients, and pollutants to streams. The decline in water quality caused by gradually increasing impervious surface cover in a watershed may trigger rapid declines in aquatic insect richness, because many phylogenetically related species may share similar tolerance thresholds to pollutants or dissolved oxygen (figure 1a). For many watersheds, once impervious surface cover reaches about 10% to 30%, aquatic insect richness declines precipitously (Paul and Meyer 2001). For this type of relationship, ecosystem management is relatively straightforward, as the path of the ecosystem from one regime to another will be the same (i.e., the same magnitude of change in an external variable will result a return to the same observed values for the internal variables). However, it should be noted that few studies have attempted to mitigate for or remove impervious surface and measure the effect on aquatic insects; this relationship may in fact have a hysteresis.

Nonlinear behavior does not always indicate the presence of alternative dynamic regimes. Positive internal feedbacks and two or more persistent states for the same external conditions must also be evident (Scheffer and Carpenter 2003). When each regime is stable over a wide range of external conditions, ecosystems are prevented from transitioning directly between regimes, and therefore the restoration of an ecosystem to a previous regime requires mitigation of the external variable beyond the value that initially caused the shift. When the need for excessive mitigation is present, an ecosystem may not follow the same path between two regimes. For example, when limestone is added to acidified streams to raise the pH and calcium concentrations to levels matching those of unaffected streams, aquatic insect diversity does not recover to the level found in unaffected streams, even after the pH and calcium concentrations are restored (figure 1b; Bradley and Ormerod 2002). The lack of recovery of aquatic communities may be due to the inability of many individuals to persist through the brief episodes of low pH after liming, to the long dispersal distances required of new individuals from other streams, and to the slow recovery (especially after decades of acid rain) of soil conditions that directly govern water chemistry in streams (Likens et al. 1996). Riverine systems have a wide variety of nonlinear responses beyond those described here, some of which may indicate alternative dynamic regimes at some scales (and in response to some

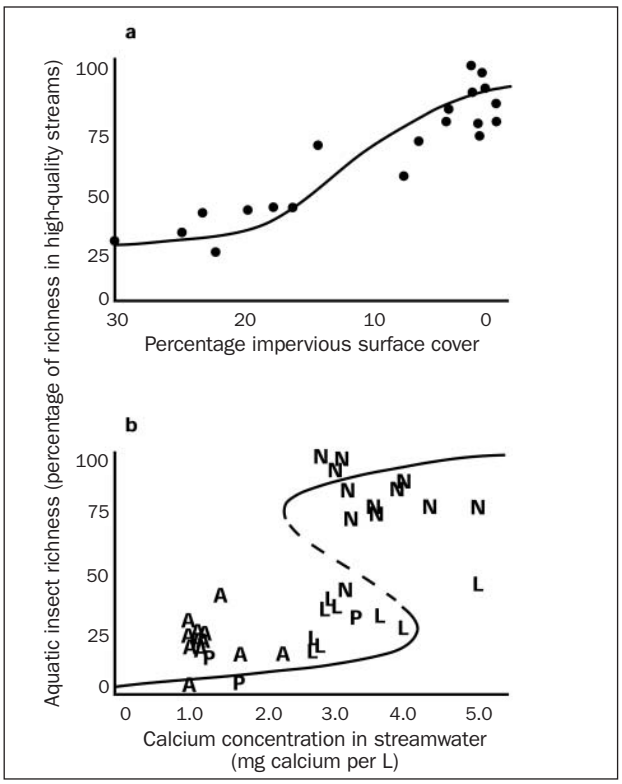

Figure 1. Species richness of aquatic insects in streams, as a percentage of insect richness in high-quality streams, in relation to (a) impervious surface cover (ISC) and (b) calcium (Ca^{2+}) concentration in streamwater. Solid lines represent stable ecosystem states; dotted lines represent unstable states. Aquatic insect richness is directly influenced by water chemistry, which is governed by land cover, soil characteristics, and the chemistry of precipitation in the watershed. (a) Insect richness remains fairly stable at low levels of ISC (less than 15%), but it decreases rapidly once the relative ISC level increases beyond 20% (based on data from Morse and colleagues [2003]). Beyond 40% ISC, aquatic insect richness is limited to several highly abundant species, such as flies, midges, and snails. (b) Aquatic insects are also affected by the pH and ion concentrations in streamwater, which reflect acidic precipitation, ISC, and geological characteristics of the watershed. Acidic precipitation leaches Ca^{2+} out of the soil and flushes it through streams, temporarily increasing the Ca^{2+} concentration of streamwater. Once most of the Ca^{2+} has been leached, the Ca^{2+} concentration and pH in streamwater drop quickly. At Ca^{2+} concentrations below about 3.0 milligrams per liter, aquatic insect richness declines quickly (Bradley and Ormerod 2002). After mitigation through liming a watershed, the Ca^{2+} concentration and pH in streams can recover quickly (although episodic low pH can occur for decades afterward), but the recovery of aquatic insect communities does not match the speed of streamwater recovery (Bradley and Ormerod 2002). Abbreviations: A, acidic streams (never limed); L, limed streams; P, prelimed streams (acidic streams before liming), N, natural streams.

external conditions) but not others (Leuven and Poudevigne 2002).

External and internal variables

Examples of ecosystems that exhibit nonlinear regime shifts are accumulating (Scheffer et al. 2001). In many of these cases, an external variable causes the system to move away from its original regime (table 1). Once the ecosystem enters a new regime, internal variables develop that then prevent the system from returning to its original regime, even after all other initial conditions are restored. Sometimes this is due to the loss of particularly dominant species, but in other cases it can be due to a restructuring of energy flows through the food web or to other such internal reorganizations. In this way, internal variables form positive feedbacks, pulling ecosystems away from unstable regimes. Subsequently, negative feedbacks outbalance and finally come to dominate over positive feedbacks when the ecosystem approaches one of the alternative dynamic regimes.

In the case of semiarid grazing systems, plants facilitate their own growing conditions by improving water infiltration into the soil (figure 2). Plants rarely colonize bare soil in these systems, because soil moisture is too low. Likewise, in shallow lakes, submerged plants maintain water clarity and therefore light penetration, fortifying their own growing conditions, whereas plants cannot colonize already turbid waters (Scheffer et al. 1993, 1994). For both of these ecosystem types, there is a value above which plant growth and establishment exceeds mortality, thereby ameliorating soil or water conditions and leading to even higher plant growth and establishment. Below this threshold, plant growth and establishment fall behind plant mortality, leading to deteriorating conditions and even lower plant growth.

Without such internal variables to maintain an ecosystem's strong attraction to a particular regime, the ecosystem will not spend a long time in that regime. Shifts to the regime will be reversible even under extreme perturbations or high levels of external forcing. The weaker the internal feedbacks that maintain an ecosystem in a particular regime, the more likely it is that an external variable will trigger the system to slide from one regime into another (Beisner et al. 2003a, Suding et al. 2004). The relative influence of internal versus external variables in causing regime shifts can vary among ecosystems and even among different regimes of the same ecosystem (Carpenter et al. 1999a, Dent et al. 2002).

Considerations of scale

The ecological concept of scale has two components, *resolution* (or *grain* for spatial data) and *period* or *extent* (Wiens et al. 2002). In theory, resolution is determined by the smallest unit that is relevant to the system dynamics of interest, although in practice it is determined by the resolution at which data are available. Time period or spatial extent is dictated by ecosystem boundaries or by observer frame of reference, determined by the longest period or largest extent over which the ecosystem can be defined or data are available. Scale is an

Table 1. Examples of nonlinear regime shifts in response to external and associated internal ecosystem variables.

System	External variables	Internal variables	Alternative regimes	References
Arid or semiarid savanna	Intense grazing	Altered fire regime	Grasslands with high fire frequency versus nonflammable invasive weeds and woodlands	Sharp and Whittaker 2003
	Grazing and rainfall	Water infiltration and nutrient retention	High plant cover of perennial grasses versus bare soil with sparse cover of annual herbs	Rietkerk et al. 1996, 1997, Rietkerk and van de Koppel 1997
	Dominance of introduced plant species (*Acacia nilotica*)	Increased spread and successful germination of introduced species through cattle ingestion and defecation	Grassland with good forage versus *A. nilotica*–dominated savanna–shrub thicket with poor forage	Brown et al. 1999
Arctic salt marsh	Grazing and grubbing (snow geese)	Sodium concentration in soil water	High plant cover versus bare soil	Srivastava and Jefferies 1996, van de Koppel et al. 1997
Tidal flat	Bottom shear stress	Benthic diatoms	High versus low cover of benthic diatoms	van de Koppel et al. 2001
Coral reef	Herbivory	Replacement of microalgae by macroalgae that are unpalatable to echinoid *Diadema antillarum* and herbivorous fish	Coral reef dominated by unpalatable macroalgae versus palatable microalgae	Knowlton 1992, Hughes 1994
Freshwater lake	Nutrient level	Turbidity caused by algae, preventing growth of submerged vegetation	Clear water with vegetation versus turbid water with high algal biomass	Scheffer et al. 1993, 1994, Beisner et al. 2003b
	Phosphorus storage and recycling from lake sediments	Phosphorus concentration in water	Clear water with vegetation versus turbid water with high algal biomass	Carpenter et al. 1999a, 1999b, 2001, Dent et al. 2002
Riparian meadow corridor, western United States	*Artemisia tridentata* spread	Lowered water tables (influencing soil moisture and temperature) and fire frequency	Wet riparian grass and sedge meadow (high water table) versus *Ar. tridentata*–dominated dry meadow (low water table)	Chambers and Linnerooth 2001
Tropical forest	Imported rainfall	Forest area above a threshold for generating a self-sustaining level of local rainfall	Tropical forest versus humid savanna	Sternberg 2001

Note: The variables listed (external and internal) are only those that have been studied, not all those that influence the ecosystem. Likewise, the regimes listed are only those that have been observed, not the entire set of possible regimes.

important consideration when determining which monitored variables are external and which are internal to a particular ecosystem regime. Changes in external triggering factors and in internal ecosystem variables occur independently over disjointed time and spatial scales. Relative to each other, the external variable is considered "slow" and the internal variable "fast."

When observations are made at a scale that does not match that of the processes influencing ecosystem regimes, changes between regimes may be missed (Peterson and Parker 1998). Regimes that appear to be distinct at one scale may simply appear as variance of a regime of a larger system. Likewise, short-term, rapid transitions between regimes may be part of a linear process at longer time scales. In this respect, making explicit the temporal and spatial attributes of an observed ecosystem can increase the likelihood of detecting patterns, understanding cause-and-effect relationships, and perhaps predicting ecosystem behavior (Dent et al. 2002). In theory, the scale at which ecologists make observations should be defined by relatively natural boundaries of ecosystems or processes, such as the shores of a lake or the grazing patterns of a distinct herd of herbivores. In practice, limitations on data-recording devices and on human life span may restrict the scale at which researchers can observe dynamic regimes and shifts between them.

For regimes on the scale of a typical grazing area for a herd of herbivores, characterized by the type of vegetative cover in semiarid grazing systems, the external variables are rainfall and grazing intensity, while the internal variables are vegetation cover and structure, and fire (figure 2). Vegetation in these systems responds rapidly to changes in grazing pressure; grazing responds more slowly, as the size of the herd changes through reproduction and emigration (van de Koppel et al. 1997). The external and internal variables are also disconnected spatially, because the external variables govern at coarser spatial scales. Therefore, the external variable is typically considered in theoretical models as a fixed parameter, neglecting

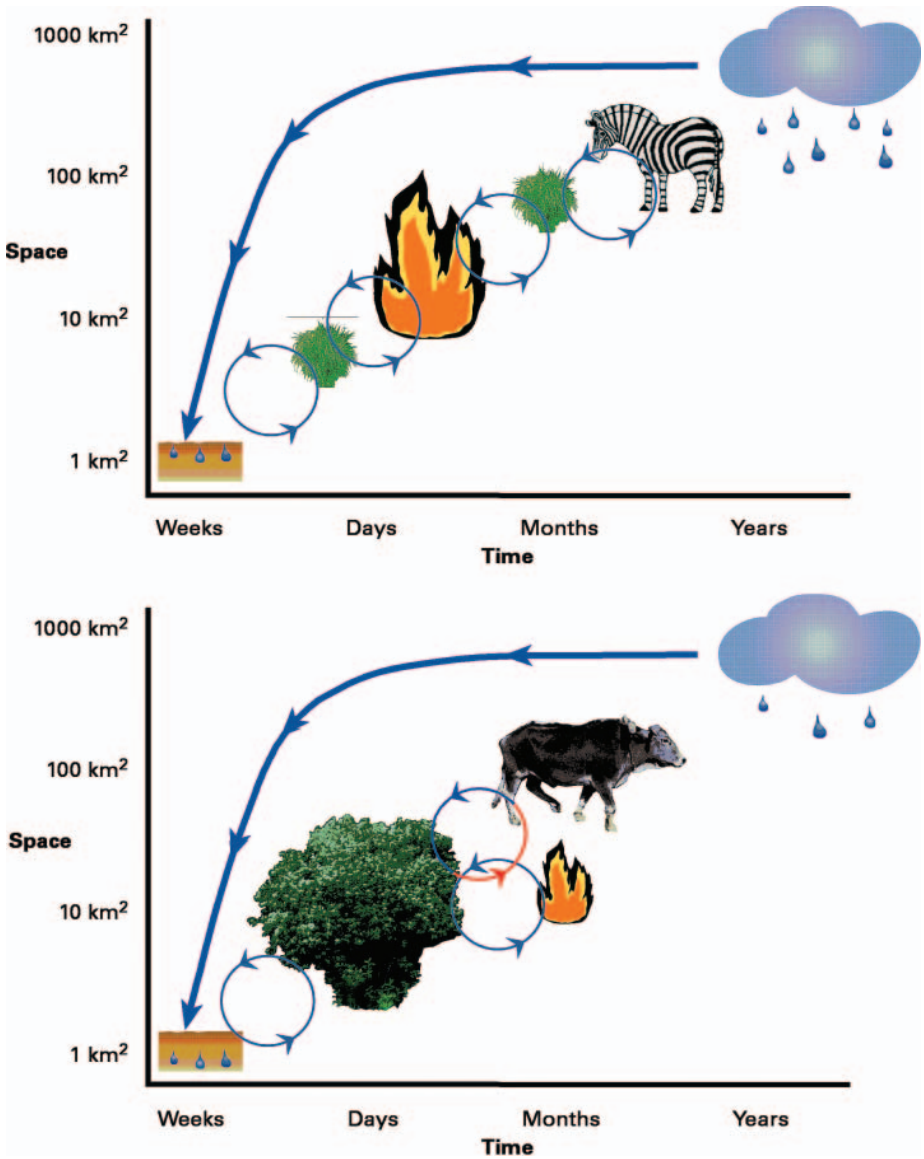

Figure 2. Dynamic regimes in arid and semiarid savannas. Arid and semiarid savannas (found primarily in Africa, Asia, and Australia) include a network of positive and negative feedbacks, such as disease and predation, that operate at multiple scales to govern ecosystem behavior. We show only a subset of these feedbacks here. Top: In savannas grazed by native large herbivores, root competition between native grasses and woody species operates at small scales to influence soil moisture and determine which spots are occupied by a grass or shrub (Knoop and Walker 1985, Skarpe 1990). Native grass species, fire, and moderate grazing form positive feedback loops (blue arrows) at intermediate spatial and temporal scales. Nomadic native herbivores stimulate grass growth through moderate grazing, and this growth maintains a high-frequency fire routine (Skarpe 1991). Annual and decadal variability in rainfall is a triggering factor, influencing soil moisture at small scales and herbivore populations and fire at intermediate scales. Bottom: In savannas grazed by domesticated herbivores, the scales of triggering and associated factors are altered and decoupled. Domesticated herbivores are corralled into small areas for extended periods, substantially increasing the intensity of grazing pressure on grassy vegetation (Skarpe 1991). Populations of domesticated herbivores are stabilized through most environmental conditions with supplemental food; this can exacerbate grazing damage in dry years. Increased grazing intensity in small areas promotes shrub invasion and dominance, especially by inedible nonnative species. Woody shrubs decrease fire intensity and frequency, and encourage further shrub dominance, leading to a shrubland regime. Shrubs are much less palatable to domesticated grazers than grasses, leading through a negative feedback (red arrow) to a drastic reduction in the size of domesticated herds that can be supported by a shrubland regime. The change from grassland to shrubland is essentially irreversible over the time scales relevant to land management (Rietkerk et al. 1997).

the possibility of its dependence on the state of the ecosystem itself through positive and negative feedbacks (figure 2).

The spatial relationship between grazing by snow geese and arctic salt marsh vegetation illustrates a regime shift produced by resource subsidy, in what has been called one of the more dramatic problems in conservation biology today (Paine and Schindler 2002). A regime shift involving vegetation damage in arctic salt marshes in Hudson Bay, which is visible from space, can be explained by the trophic boundary of snow geese populations represented by agricultural crops in the Great Plains (Jano et al. 1998, Gadallah 2002). Whereas the natural food resources of snow geese normally fluctuate, the food resources provided by crops tend to be more reliable. The shift from vegetated to nonvegetated salt marshes would be prevented if the trophic boundary of the snow geese were fully coupled with that of the vegetation, because the snow goose population would collapse in response to decreasing vegetative cover in the marshes. Such temporal and spatial uncoupling of external and internal variables undoubtedly and critically influences the possibility and extent of regime shifts and resilience in ecosystems (figure 2).

Simply changing the scale at which an ecosystem is monitored can change the observed relationship between external and internal variables. For instance, at the scale of an individual plant, precipitation and vegetation cover are independent of each other. At the habitat scale, precipitation is an external variable, dictating whether a patch will be dominated by one type of vegetation or another. However, at the ecosystem scale, vegetation may influence precipitation. For example, tropical forests in South America maintain themselves through high self-generated precipitation (Sternberg 2001). If forest area is reduced beyond a critical threshold, forest stands that are smaller than this threshold will shift to a drier savanna condition (which does not recycle as much moisture in this way), because they will not generate enough precipitation to supplement imported rainfall in most years. At the regional or global scale, over decades or centuries, climate and ecosystems appear to be coupled, as demonstrated in the positive feedbacks between vegetation and precipitation that may organize alternative dry and wet regimes in the Sahel and Sahara desert regions (Higgins et al. 2002). At this scale, precipitation is no longer the external variable; instead, it has become an internal variable. Viewed from several scales, concurrent global regime shifts in ecosystems at multiple scales are a distinct possibility.

Ecosystem restoration and management

The dynamic regime concept lends itself well to ecosystem management and restoration efforts. Using the regime concept for ecosystem management and restoration is more difficult than using more simplistic, linear relationships, because identifying dynamic regimes requires a large amount of data at multiple temporal and spatial scales (Landres et al. 1999). However, the dynamic regime concept may help create more realistic models for some ecosystems, which could decrease the likelihood of irreversible and costly shifts to undesirable regimes that may result from management strategies based on inaccurate linear models (Laycock 1991, Peterson et al. 2003). Management activities, like all forces of change, can move ecosystems from one regime to another and can also alter the size and boundaries of regimes (Bellamy and Lowes 1999). Moving the boundaries of a regime may alter its stability or resilience, which in turn may be correlated with ecosystem health or integrity (Okey 1996, Leuven and Poudevigne 2002).

Although the phrase *ecosystem management* may be used in different ways, the different versions of this concept all have as a goal the restoration and preservation of some key attributes of an ecosystem that are desirable to human societies (Yaffee 1999, Cairns 2000). Preservation, in this sense, is interpreted as maintaining the system within some "range of natural variability" (Landres et al. 1999) or "bandwidth" (Roe and van Eeten 2001). Management models use the ecosystem regime concept (also known as the "alternative state concept" or "state-and-transition model"; Westoby et al. 1989, Chambers and Linnerooth 2001) to integrate economic and ecosystem behavior and determine sustainable levels of human activity (Bellamy and Lowes 1999, Brown et al. 1999). Carpenter and colleagues (1999a, 1999b, 2001) developed a lake eutrophication model to examine the probability of transitions between two potentially irreversible regimes (eutrophic and oligotrophic), given particular economic policies. The ratio between the economic benefits derived from a eutrophic lake regime (using a lake as a receptacle for excess phosphorus) and those derived from an oligotrophic regime strongly influenced the likelihood of whether a lake would shift to and remain in a eutrophic condition. The smaller the lake, the more likely that it would remain irreversibly eutrophic, as the phosphorus input from outside the lake overwhelmed the lake's ability to mitigate phosphorus concentrations through plant growth or sediment sequestration.

Ecosystem restoration involves pushing a system back to a desired regime with particular structural and functional characteristics (Palik et al. 2000). However, forcing an ecosystem shift from one regime to another is often a nonlinear process, and because of this nonlinearity, it may require a larger amount of time, effort, and resources than expected. Hystereses occur when processes or parameters must be altered beyond the original point at which the system shifted to the undesired regime (Sazykina et al. 2000, Carpenter et al. 2001, Higgins et al. 2002). The existence of hystereses in dynamic systems is often revealed as an unwelcome surprise; hystereses may not be obvious until restoration efforts are undertaken (Hobbs and Norton 1996, Beisner et al. 2003a). Lake restoration projects have been some of the first to uncover hystereses, as a concerted effort to decrease nutrient loads from anthropogenic sources has not resulted in a linear response in lake water quality or aquatic communities (Scheffer et al. 1993, Jeppesen and Sammalkorpi 2002). Likewise, the combined influences of fire frequency, rainfall, soil conditions, and grazing in savanna ecosystems create a maze of hysteretic effects between grassland, woody shrubland, and desert regimes (Perrings and

Walker 1997, van Langevelde et al. 2003). In fact, multiple alternative basins of attraction are possible for most ecosystems, and restoration efforts may push a system from one undesirable but resilient regime to another equally undesirable and resilient regime (Carpenter et al. 2001).

The dynamic regime concept can also help explain why restoration projects fail (Suding et al. 2004). Ecosystem restoration can fail if efforts are insufficient to push the ecosystem across regime thresholds, or if efforts are concentrated at an inappropriate scale. Restoration can also appear to fail if ecosystems are not given enough time to allow efforts to take effect. As explained above, hystereses between regimes require a greater effort to shift an ecosystem than would be assumed for a linear system. The resilience of many of the possible regimes in larger systems may be higher than in smaller systems, and moving large ecosystems from one regime to another may take more time and effort (Carpenter et al. 1999a). Indeed, the longer an ecosystem spends in a particular regime, the more likely it is that the ecosystem's resilience to future disturbances will increase, as populations and trophic relationships stabilize and self-regulating feedbacks develop (Cropp and Gabric 2002).

Finally, the dynamic regime concept can illustrate the limitations of an ecosystem management or restoration approach that fails to include the socioeconomic forces on ecosystems. Global climate change serves as an excellent illustration. As atmospheric carbon dioxide (CO_2) increases at a fairly linear rate, average surface temperatures and associated climate factors also change linearly. However, these climate factors may not increase linearly forever; once a critical concentration of atmospheric CO_2 is reached, the global climate may undergo a rapid nonlinear shift from the current regime (Broecker 1997, Alley et al. 2003). This global climate regime shift may spawn a cascade of ecosystem regime shifts that may bear little resemblance to previous episodes. Given the uncertainties both in the probability of regime shift and in the characteristics of potential new regimes, it may be more feasible to limit contributions to atmospheric CO_2 levels through socioeconomic systems rather than try to manage a multitude of simultaneous ecosystem regime shifts at multiple scales.

Conclusions

Ecosystems are dynamic systems with a capacity to respond nonlinearly (and therefore surprisingly) to disturbances and interactions across scales. With increased attention to the dynamic regime concept, advances in the theory underpinning the concept will improve its application in ecosystem management and restoration efforts. However, even an accurate understanding of dynamic regimes in different ecosystems may not alleviate all the potential problems of applying the dynamic regime concept to ecosystem management and restoration. The products and services provided by alternative ecosystem regimes can differ substantially, and the demands for these services may be in conflict. For example, fishers with a preference for carp and other fish species that thrive in eutrophic lakes may not agree with a management strategy supported by fishers with a preference for trout or other species that depend on an oligotrophic environment.

At some scales, ecosystem management strategies may sufficiently influence both internal and external variables to maintain or restore a system to a desirable regime. However, at larger scales, external forces are often beyond the focus of ecosystem management and restoration; instead, they fall within the sphere of national and international policy. Therefore, the best ecosystem management strategies may ultimately fail if government policy is inadequate to control human contributions to triggering forces or, at worst, if it exacerbates these forces.

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