Society for Ecological Restoration International

Highlights from The Science and Practice of Ecological Restoration Series

Series Editor: James Aronson
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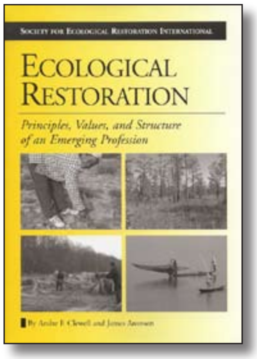

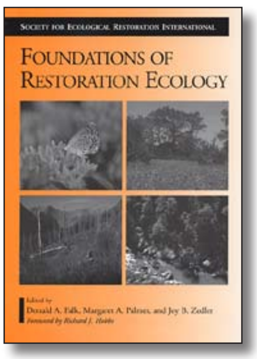

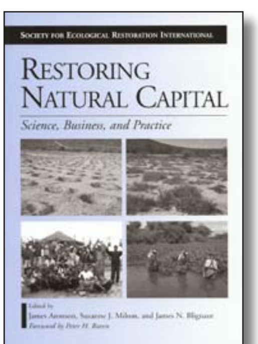

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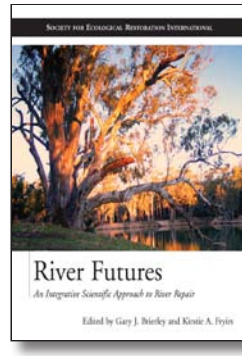

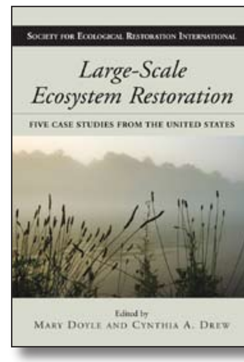

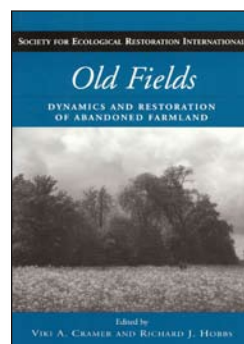

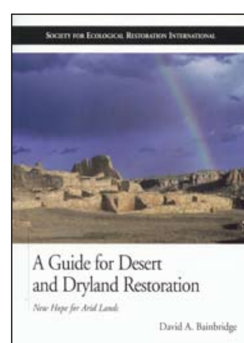

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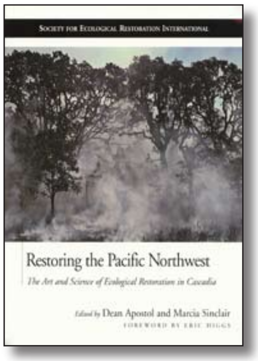

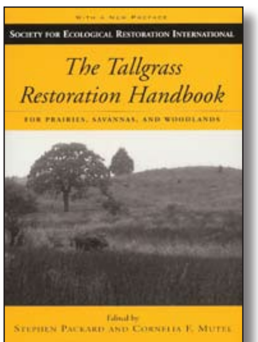

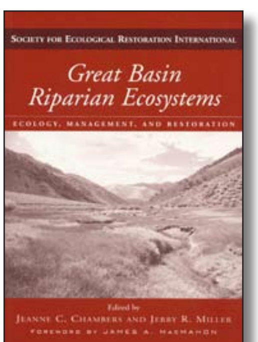

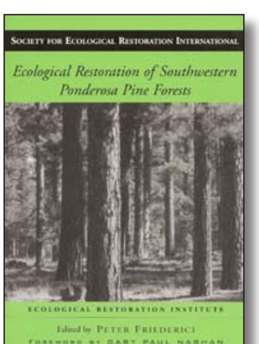

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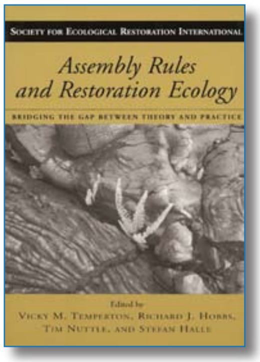

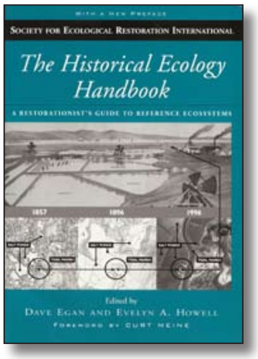

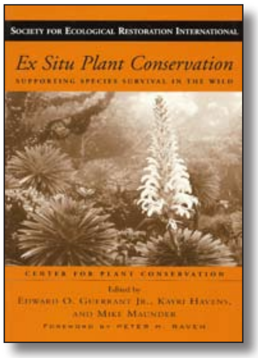

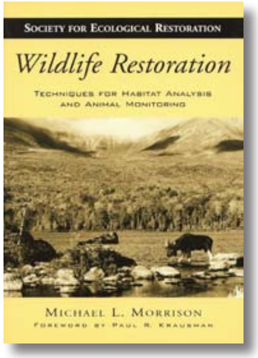

Ecological Restoration

Principles, Values, and Structure of an Emerging Profession

Andre F. Clewell and James Aronson

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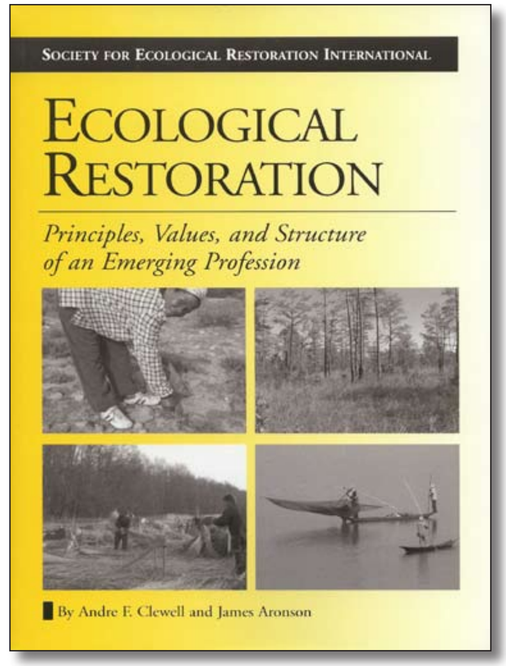

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About This Excerpt

The field of ecological restoration is a rapidly growing discipline that encompasses a wide range of activities and brings together practitioners and theoreticians from a variety of backgrounds and perspectives. In *Ecological Restoration*, Andre Clewell and James Aronson offer a long-awaited guide to the practice of this ambitious and promising new discipline. This excerpt examines the roles, contexts, and institutional structure of project work.

From chapter 10, "Project Roles and Contexts"

In this chapter we describe ecological restoration projects from the perspective of their organization and structure. We begin with the roles various personnel play in the development and execution of a project. Then we provide an outline of project contexts or circumstances in which projects are conducted. The various contexts have different strengths and weaknesses, which we identify. We note how projects tend to change over time from being exploratory and experimental at first to refined and standardized later. This information will let students and entry-level personnel know what they can expect and where they may want to concentrate their talents as their careers begin. As the chapter progresses, we will present material of broader interest.

Project Roles

Who sponsors restoration projects? Who administers them? Who makes decisions, and who carries them out? Every ecological restoration project requires personnel to fulfill certain roles, beginning with the project sponsor and continuing with the restoration practitioner, project director, restoration planner, and project manager. In small or uncomplicated projects, the same person may assume two or more of these roles. Organization charts may identify project personnel with other titles; however, each of these roles is filled by someone, regardless of his or her title. Every project has at least one practitioner and sometimes many. Larger sponsoring organizations may add other levels to the organizational table for a project, such as an administrator to whom the project manager reports. Project organization becomes even more complex as contractors and subcontractors are included, with their own hierarchies of personnel and departments with project responsibility. We ignore these complexities and describe the basic project roles in this section.

Sponsor

The organization or entity that undertakes an ecological restoration project and assumes the responsibility for its accomplishment is its sponsor. A sponsor may be a government agency or a transnational organization; a for-profit firm or corporation; a nongovernment organization (NGO); a philanthropic foundation; a school, university, or research institute; a public museum, arboretum, or zoological park; a professional association; a branch of the military; a monastery or other religious order; a tribal council of elders; a women's self-help group, which are becoming increasingly common in India and Latin America; another kind of community-based organization (CBO); or an individual landowner or manager. The sponsor approves the restoration project, provides or attracts funding, assembles personnel who will accomplish the project, provides an administrative structure, and provides oversight to ensure its satisfactory completion. The project may be accomplished in house using the sponsor's own employees or members, or some or all of the work can be delegated to outside individuals, consulting firms, or other organizations under contract, purchase order, or some other agreement to provide services. Labor can be provided by paid personnel or by volunteers who work without monetary compensation. To a restoration practitioner who is contracted, the sponsor is usually known simply as the client.

Practitioner

A restoration *practitioner* is someone who personally conducts or supervises ecological restoration in the field at project sites. Specifically, practitioners engage in project implementation and aftercare. In many projects, practitioners additionally inventory a project site before the initiation of restoration activities, select and inventory reference sites, prepare project plans, conduct or supervise site preparation activities, and monitor project sites that have undergone restoration. In other projects, sponsors delegate these responsibilities to others. A practitioner can be an employee of an organization that is conducting ecological restoration, or a consultant, contractor, subcontractor, or volunteer who is engaged by that organization. A practitioner may also be the owner of the property that is undergoing restoration. A restoration project may be accomplished by a single practitioner, or two or more practitioners who work collectively on all aspects or separately on different aspects of a project. The chief practitioner, if one is appointed, supervises other practitioners and is responsible for the overall conduct of on-site restoration activities. A practitioner may assume broad responsibilities and authority for conducting restoration or may serve as a technician who performs specific tasks assigned by a supervisor.

Project Director

The project director is the person who has a comprehensive vision for the project, including its technical, social, economic, strategic, political, historical, and other cultural aspects and implications. The project director is superior in rank to the project manager and is responsible for the overall technical direction and leadership of a project. The project director is critically involved with the conception of a project and the development of project plans. The project director formulates or approves project goals and objectives and selects or approves reference models and strategies for accomplishing restoration. The project director receives briefings from the project manager and evaluates project monitoring reports and other technical documents that may be produced. The project director ensures that executive officers, accountants, legal counsel, and other administrative officers of the sponsoring organization understand the project and carry out their respective responsibilities. The project director represents the project before the board of directors, philanthropic foundations, public officials, stakeholders, and the general public or delegates these duties to others.

Restoration Planner

The restoration planner (or a planning staff) prepares project plans, including maps, drawings, and written instructions as needed. Ideally, the practitioner contributes substantially to the planning process or even serves as the planner, as commonly happens on smaller projects that do not entail many government permits or outside contractors. The degree of detail in project plans may vary widely between projects, depending on project size and complexity and on the requirements of the sponsoring organization. Much detail may be required by government agencies whose approval is needed before project implementation. Project plans typically are appended to permits and are carried out as a permit condition. Detailed plans are also useful for preparing contract stipulations that are to be followed by the firm that provides practitioner services to the sponsoring organization. Penalties that affect monetary compensation are prescribed if contractors fail to comply with contract stipulations. In such instances, the planning function may include legal as well as technical capacity.

Project Manager

In most projects, restoration practitioners are supervised and report to a superior who is either the project manager or someone who fulfills that role. The project manager is responsible for ensuring that a given restoration project is conducted satisfactorily on behalf of the sponsoring organization. The project manager administers day-to-day operations such as scheduling personnel, arranging for deliveries of planting stocks and equipment, ensuring adherence to contract stipulations, and approving expenditures. Sometimes the practitioner does most of this work, and the project manager ensures that it is accomplished. Another firm or organization that has been engaged to provide restoration services under contract sometimes appoint its own project manager. In such instances, the two project managers may communicate with each other, and practitioners receive directions primarily from the project manager in their own firm.

Satisfactory restoration projects require that the practitioner and the project manager remain in close communication, more so than in construction projects, where tasks with more predictable outcomes are conducted. The success of many restoration projects depends on manipulating living organisms of different kinds, and the chances for surprise are much greater. The practitioner must react to unanticipated situations to ensure the success of the project. The project manager is obliged to ensure adherence to schedules, budgets, and contract stipulations, which may not allow for contingencies. In such instances, the practitioners should educate project managers and provide succinct information and persuasive logic that the managers can use effectively when interacting with people at higher administrative levels. We cannot overstate the importance of respectful and cordial relations between practitioner and project manager, particularly in ecological restoration projects of long duration.

Project Contexts

The context of a restoration project consists of the circumstances under which it is conducted. The administrative structure of a project is the most important aspect of context. Other factors contributing to it are the availability of funding, labor, equipment, and materials such as planting stocks. Project site accessibility and seasonal constraints (e.g., inclement weather) can also influence the context, as can regulatory and legal constraints. We emphasize administrative structure in the ensuing discussion.

The ways in which different projects are administered vary widely. Project administration determines the degree of responsibility that the restoration practitioner is given, the amount of authority that the practitioner is allotted, and ultimately the flexibility that the restoration practitioner can apply to solve problems that arise. The North Branch Prairie project and the Dogleg Branch project illustrate two extremes in project administration and context. The North Branch Prairie project was described in *Miracle Under the Oaks* by William Stevens (1995) and critiqued by Peter Friederici (2006). The Dogleg Branch project is described in Virtual Field Trip 7.

The North Branch Prairie project was initiated in 1977 by Steve Packard and a small group of environmental activists near Chicago, Illinois. Packard approached a public official in the Cook County Forest Preserve District and asked whether they could volunteer to clean up trash, cut some brush, scatter some seeds, and generally refurbish degraded prairies that the district owned. District personnel had wanted to begin such work themselves but were hampered by a lack of funds, and they accepted Packard's offer. The work began and soon attracted other volunteers. The idea of restoring Chicago's former ecosystems spread like a prairie wildfire. Soon, hundreds of citizens were spending their free time working alongside Packard, essentially without plans or administrative structure. By 1993, more than 3,000 volunteers had restored more than 6,700 hectares of degraded prairie and associated oak savanna in an amazing display of altruism.

Compare the North Branch Prairie story to that of the restoration of forested wetlands in the headwaters of Dogleg Branch on surface-mined and physically reclaimed land in Florida, described in the Virtual Field Trip 7. That project required two years of work simply to obtain the required government permits. Permits were eventually issued after the mining company had conducted a four-year pilot project to demonstrate that native trees could be grown and a two-year ecological inventory of local forested wetlands that served as reference sites (Clewell et al. 1982). Professionals who were involved in the project included mining engineers, mine planners, environmental consultants, native nurseries, project managers, heavy equipment contractors, attorneys, top officials in state government, and large support staffs that produced many reams of paperwork. The project was a very costly and well-orchestrated production in which the actual restoration work at the project sire seemed like an afterthought.

The contrast between the North Branch Prairie and Dogleg Branch projects could scarcely have been greater. They demonstrate extreme examples of the contexts in which restoration practitioners find themselves working. There is no preferred way to organize, plan, and implement restoration projects. The particular circumstances for a project determine its context. The underlying difference between the North Branch Prairie and Dogleg Branch projects was that the former was an elective project, whereas the latter project required prior government approval.

Let's look at these two projects from the perspective of the restoration practitioner. At North Branch

Prairie, almost everyone involved was a restoration practitioner. Steve Packard assumed the role of project director, and he and several others assumed the collective role of project manager as well. The Cook County Forest Preserve District was nominally the sponsor, and its personnel provided skeletal administration. Packard referred to existing ecological literature, a general knowledge of the few remnant patches of prairie and oak savanna, and the species list of an early naturalist as references and as an indication of historic trajectory. They essentially developed project plans as they worked on site. Their administrative mode was collegial. In other words, Packard and the other practitioners who worked most closely with him made project decisions by consensus. They assumed almost total responsibility for all restoration work. The Cook County Forest Preserve District retained basic authority for the project because the project took place on lands under their jurisdiction. District personnel established the bounds for project work to ensure that it was legal and complied with the district's overall mission. Otherwise, Packard and his cadre assumed authority for project operations. In this context, the practitioners enjoyed broad flexibility to conduct the project as they saw fit (Packard 1988, 1993).

The North Branch Prairie project was a grassroots, bottom-up endeavor that was not mandated by a public agency. Instead, the Cook County Forest Preserve District benefited from the broad public support of hundreds of citizens who volunteered their free time as restoration practitioners. This was a marvelous example of people taking collective responsibility for their own concerns in a manner that nicely reflects the four-quadrant model of ecological restoration (see Chapter 8). Ecological values were fulfilled directly by the restoration. The motivation for many volunteers was the fulfillment of individual values such as reconnecting with nature and responding to environmental crises, as described in Chapter 7. Public celebrations at the restored prairie were described by Holland (1994) and are among the evidence of the fulfillment of cultural values. The restored prairies and oak savannas represent natural capital and provide socioeconomic services.

In great contrast, the Dogleg Branch project was conducted by only a few restoration practitioners, principally Andre Clewell and several colleagues. Because of the safety and liability issues, no volunteers were invited or allowed on the property. The mining company was the sponsor, and its employees assumed the other roles of project administration, director, and project manager. Most were engineers. Detailed project plans were prepared by the company, which incorporated specific conditions that were required by permit from the State of Florida. These conditions, in turn, were based in part on a restoration plan written by Clewell that identified restoration goals, objectives, performance standards, and the reference model. The latter was embodied in the aforementioned document that described historic conditions and contemporary changes in the historic trajectory that were attributable to land use (mainly fire suppression that allowed broadleaved forest typical of river valleys to replace upland pine savannas). Much of the project work was conducted by earthmoving firms, tree planting crews, and other subcontractors hired by the mining company. The role of the restoration practitioner was largely to serve as a liaison with foremen of subcontracting companies, to test new restoration methods such as interplanting undergrowth species, to monitor forest development, and to suggest improvements to the restoration process for approval by mine managers.

The Dogleg Branch project was required by the State of Florida (primarily; other government entities were also involved) and was administered from the top down by the mining company. Stakeholder involvement was limited to formal hearings that were required by law, in which citizens could express their interests. Comments were largely limited to local residents who were concerned about mining operations near their properties and environmental organizations that were generally opposed to surface mining. The intent of the project was to repair environmental damage that was an unavoidable result of mining rather than to cause net ecological improvements. No fulfillment of the personal, cultural, and socioeconomic values described in Chapter 7 was intended. In other words, this was a compensatory mitigation project. After this and other restoration projects on mine land were complete, the land was donated to the State of Florida and became the Alafia River State Park, which was a cultural improvement. However, negotiations for the donation of the land were initiated after Dogleg Branch was nearly restored, and the restoration site has not yet been opened for public access. Dogleg Branch restoration is narrowly focused as a flatland project in terms of the four-quadrant model in Chapter 8 in that it satisfies ecological and socioeconomic elements that are expressed in state policy that presumably represents the sentiment of the electorate.

However, a number of restoration techniques were tested during the course of Dogleg Branch restoration, some for the first time. The results were made available for use by restoration practitioners who toured the site, attended conferences where the project was described, and read public descriptions (e.g., Clewell and Lea 1990; Clewell et al. 2000). Even more importantly, the Dogleg Branch restoration project, and several others that were initiated at the time (Clewell 1999), demonstrated that complex forest and stream restoration could be conducted on land that had been literally turned upside down by mining. In this regard, the Dogleg Branch restoration project changed the perceptions of people who had not previously realized the potential of ecological restoration. This and similar restoration work has had the salubrious effect of hastening the era of ecological restoration and providing jobs for many practitioners in Florida and elsewhere. However, it has also given regulated interests a rationale for convincing government agencies to issue permits for development that will cause environmental damage with the promise that the damage will be compensated by ecological restoration as a form of mitigation. This strategy could be justified if regulated interests were required to successfully restore more than they damaged, but this eventuality awaits documentation as a normally occurring outcome. We hope that restoration practitioners will be more than battlefield physicians in the environmental wars.

Foundations of Restoration Ecology

Edited by Donald A. Falk, Margaret A. Palmer, and Joy B. Zedler

Foreword by Richard J. Hobbs

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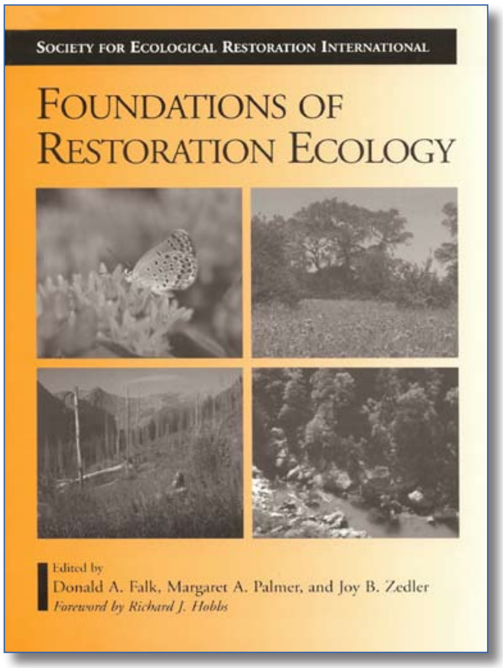

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About This Excerpt

"Restoration is the keystone strategy for conserving biodiversity, and ecology has matured into a central discipline of the biological sciences. This important work shows that their synergy offers new hope for the future of life on Earth." —Edward O. Wilson, University Research Professor Emeritus, Harvard University

Linking theoretical models of ecosystem and community change with restoration ecology has the potential to advance both the practice of restoration and our understanding of the dynamics of degraded systems. In the chapter from which this excerpt is drawn, Katharine Suding and Katherine Gross consider ecological theories that address questions about how systems change and may reduce the risk of unpredicted change in restoration projects.

From chapter 9, "The Dynamic Nature of Ecological Systems: Multiple States and Restoration Trajectories," by Katharine N. Suding and Katherine L. Gross

One feature of ecological systems is that they are ever-changing and dynamic. As ancient Greek philosopher Heraclitus claimed, "You can never step in the same river twice." Moreover, rates and directions of change in systems are shaped increasingly by human activities. These effects can be intentional or the consequences of engineering of the systems and surrounding landscapes to provide specific services to humans. The dynamics of an ecological system, particularly of a system slated for restoration, is a function of many factors, some deterministic and some stochastic, working at several temporal and spatial scales.

In considering how systems change in restoration, we address several questions:

1. What types of trajectories characterize the recovery of degraded ecosystems? Is the pathway to recovery similar to the pathway to degradation?
2. Can we predict the end states of restoration pathways? Are they similar to states prior to degradation?
3. How will dynamics that occur on very different scales of space and time relate to one another? What should be the scale of focus?
4. How much inherent variability does an ecological system require for adequate recovery and adaptive capacity for change in the future?

In this chapter, we consider ecological theories that help address these questions and may reduce the risk of unpredicted or undesired change in restoration projects. While theory can help guide restoration efforts, it does not provide simple or universal answers for the challenges that confront restoration. Restoration efforts that document species turnover and environmental attributes over time can help test and refine ecological theory related to community dynamics. Links between restoration and community dynamics advance both the practice of restoration and theories of ecological dynamics. We survey the progress and the further potential of this connection.

Major Theories and Connection to Restoration

Over the last one hundred years, extensive work has documented how communities and ecosystems change in response to disturbance. Despite the extensive documentation of patterns (Figure 9.1), a general conceptual framework concerning the controls on species turnover and ecosystem development is still debated. Several contrasting views concerning the mechanisms and predictive nature of these dynamics persist today. In this chapter, we will focus on three views: equilibrium, multiple equilibrium, and non-equilibrium. We discuss each of these and relate them to the concept of fast and slow processes (sensu Rinaldi and Scheffer 2000) as a way to evaluate mechanisms of recovery.

Single Equilibrium Endpoint

Equilibrium systems are assumed to return to their predisturbance state or trajectory following disturbance (Table 9.1). This theory predicts a classical successional trajectory: steady, directional change in composition to a single equilibrium point (Clements

1916; Odum 1969) (Figure 9.2a). Recovery in an equilibrium framework is a predictable consequence of interactions among species with different life histories and the development of ecosystem functions. Strong internal regulation occurs through negative feedback mechanisms, including competition and herbivore/predator interactions, as well as climate-ecosystem couplings and life-history tradeoffs. Many of these mechanisms are considered aspects of community assembly rules (Weiher and Keddy 1999; Booth and Swanton 2002), although assembly rules do not necessarily assume single equilibrium dynamics.

In some cases, community development can proceed "spontaneously," with little or no intervention, to reach desirable target states (Prach et al. 2001; Khater et al. 2003; Novak and Prach 2003). Mitsch and Wilson (1996) argue that nature has a "self-design" capacity as species assemble themselves. However, the extent to which this capacity can be expressed in a recovery will depend on how degraded and isolated it has become prior to restoration efforts (Bakker and Berendse 2001). Some restoration efforts are designed to accelerate natural succession so that the ecosystem develops along the same trajectory as it would in the absence of intervention but reaches the goal endpoint sooner. For instance, restoring a severely degraded river back to its more natural flow regime via dam removal can enhance recovery of the surrounding plant communities (Rood et al. 2003; Lytle and Poff 2004). Similarly, prescribed burning of degraded grasslands can promote restoration of native plant assemblages, particularly if the fire management regime is applied according to historical patterns (Baer et al. 2002; Copeland et al. 2002). Thus, restoration of some communities can take a single equilibrium approach to spur recovery along a successional trajectory.

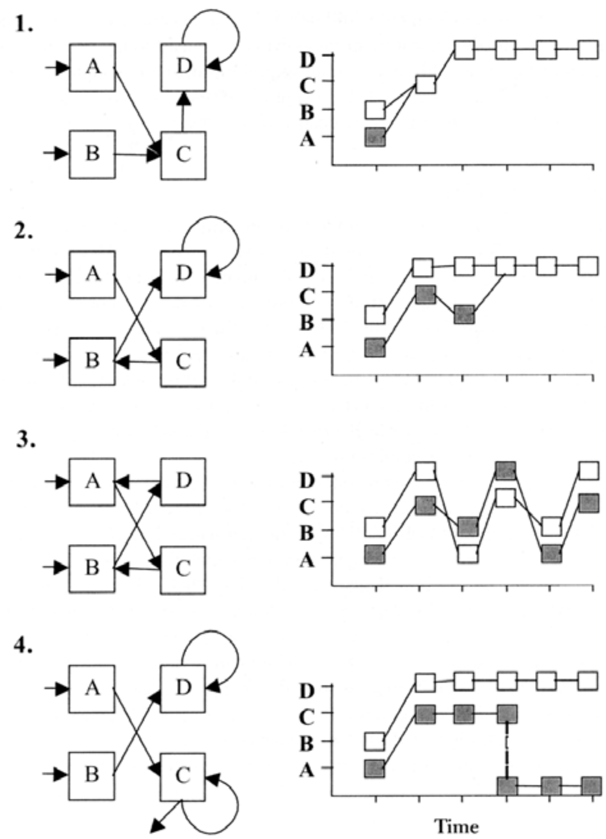

Figure 9.1: Dynamics of species replacement have been predicted to take many different forms. Four general patterns of trajectories, each with two starting points (assemblages A and B) are shown here: (1) Convergent trajectories where initial variability eventually converges to similar species composition, often termed the (D) equilibrium "climax" community. (2) Initially divergent trajectories that eventually converge to one equilibrium state. (3) Divergent trajectories that never converge and never reach a permanent state. (4) Divergent trajectories that go to two different stable states (C and D) and, in the case of C, experience an abrupt shift to a third state.

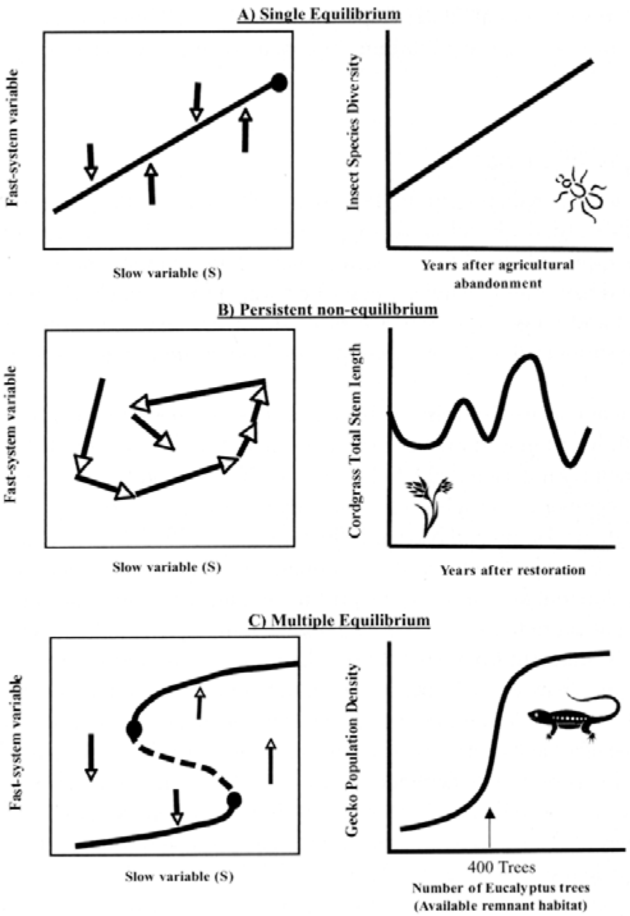

Figure 9.2: Examples of dynamics predicted by single equilibrium, persistent non-equilibrium, and multiple equilibrium theories (A–C). For each, the left frame shows predicted combinations of "fast" and "slow" variables; arrows indicate direction of change if not at equilibrium. The right frame shows a stylized example from the ecological literature that is consistent with the ecological predictions. In A, changes in the slow and fast variables are linear and unidirectional. Insect species diversity increase linearly in a Minnesota old-field with years since abandonment. Increases in aboveground productivity with time is likely the slow variable that drives the change in insect diversity (Siemann et al. 1999). In B, a persistent non-equilibrium exists with no predictable trajectory. Total stem length of cordgrass (Spartina foliosa) shows high interannual variability and no directional trends in time since restoration in San Diego Bay, CA (Zedler and Callaway 1999). In C, at a single level of the slow variable there are two possible equilibrial states. Examples of a pattern predicted by this dynamic, shown in Figure 9.1(4), are strong threshold effects as the slow variable changes. For instance, in a fragmented Eucalyptus forest in Australia, the probability that a gecko species (Oedura retiulcata) persists decreases dramatically if the forest remnant contains less than 400 trees (Sarre et al. 1995).

Table 9.1: General theories that attempt to predict how the composition and function of systems change over time and/or behave following a disturbance.

	Equilibrium	Multiple Equilibrium	Non-equilibrium
Assumptions	Climax equilibrium, unidirectional, continuous	Equilibrium, multidirectional, discontinuous	Persistent non-equilibrium, nondirectional, discontinuous
Permanent states	One (climax)	More than one	None
Trajectories	Convergent	Regime shifts, collapses	Divergent, arrested, cyclic
Predictability	High; based on species attributes	Moderate; possible but difficult	Low; chance and legacies important
Important factors	Species interactions, ecosystem development	Initial conditions, positive feedbacks, landscape position	Chance dispersal, stochastic events

Restoring Natural Capital

Science, Business, and Practice

Edited by James Aronson, Suzanne J. Milton, and James N. Blignaut

Foreword by Peter Raven

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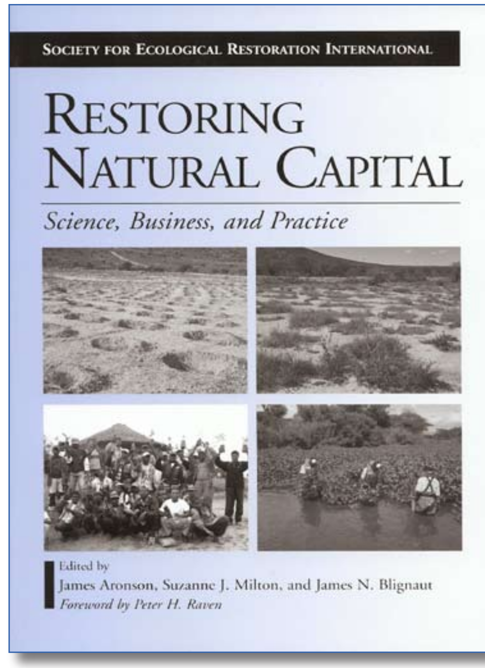

Contents

About This Excerpt

How can environmental degradation be stopped? How can it be reversed? And how can the damage already done be repaired? *Restoring Natural Capital* brings together social and natural scientists from the developed and developing worlds to consider these questions and examine specific strategies for restoring ecosystem goods and services in natural and socioecological systems. This excerpt from the final section of the book focuses on the importance of mainstreaming the restoration of natural capital, highlighting the fact that restoration is becoming an essential intervention.

From chapter 34, "Mainstreaming the Restoration of Natural Capital: A Conceptual and Operational Framework," by Richard M. Cowling, Shirley M. Pierce, and Ayanda M. Sigwela

Protected areas alone will never achieve all of the goals and targets required to ensure the persistence of the world's natural capital (e.g., Rosenzweig 2003) and the delivery of services that intact ecosystems supply (Kremen and Ostfeld 2005). Consequently, the burden of conserving (and restoring) natural capital will have to fall increasingly on sectors such as agriculture, transport, forestry, mining, and urban development (e.g., Hutton and Leader-Williams 2003). The mainstreaming of biodiversity concerns is one strategy used by the conservation community to respond to the challenge of ensuring the persistence of natural capital and ecosystem services in utilized landscapes (Pierce et al. 2002; Petersen and Huntley 2005a). In essence, mainstreaming may be defined as the process of creating awareness of the value of natural capital in sectors that currently ignore or discount it, to the extent that they will incorporate conservation actions into their routine activities.

A key conservation action in production landscapes is the restoration of degraded or transformed natural capital, as has been pointed out in many of the chapters in this book. Our aim here is to provide a conceptual and operational framework for mainstreaming the restoration of natural capital in production landscapes.

What Is Mainstreaming?

Although mainstreaming is a relative newcomer to the biodiversity and natural capital lexicon, it is an important one, since mainstreaming is a component of the institutions and strategies of some major global biodiversity initiatives. For example, the concept is embedded in several articles of the Convention on Biological Diversity (ratified 1995). It also underpins the ecosystem service approach of the *Millennium Ecosystem Assessment* (MA) and is the explicit objective of the Strategic Priority 2 of the Global Environmental Facility's GEF-3 (2004) Program of Work: "Mainstreaming biodiversity in production landscapes and sectors—to integrate biodiversity conservation into agriculture, forestry, fisheries, tourism, and other production sectors in order to secure national and global environmental benefits" (Petersen and Huntley 2005b). Undoubtedly mainstreaming initiatives will attract considerable resources from funding agencies over the next decade.

According to Petersen and Huntley (2005b) the objective of mainstreaming is "to internalize the goals of biodiversity conservation and sustainable use of biological resources into economic sectors and development models, policies and programs, and therefore into all human behavior." Cowling et al. (2002) identified the following list of desired outcomes of mainstreaming:

- The incorporation of biodiversity considerations into policies governing sectoral activities
- The simultaneous achievement of gains in biodiversity and in the economic sector (the win-win scenario)
- Sectoral activity being recognized as based on, or dependent on, the sustainable use of biodiversity
- Situations where sectoral activities result in overall reversal of biodiversity losses

Viewed as a process, mainstreaming is a means to spread the responsibility and benefits of conserving biodiversity and restoring natural capital across a diverse range of sectors. This requires the identification of scenarios that provide benefits for both the natural capital and the targeted sector, and the implementation of actions (for example, the creation of institutions, including incentives) that enable responsible bodies to accomplish these scenarios.

Mainstreaming interventions may happen at all scales of organization and geography, from encouraging backyard conservation of natural capital in a neighborhood to the impact of a multilateral environmental agreement on the global ocean-transport system. Furthermore, a wide range of actors will bear the costs and enjoy the benefits, material and spiritual, associated with mainstreaming, and these will accrue over short and long timescales (Petersen and Huntley 2005b).

There are very few documented cases of effective mainstreaming. Pierce et al. (2002) provide examples from South Africa, and Peterson and Huntley (2005a) from elsewhere in the world. Others, although not explicitly conceptualized as such, appear in Daily and Ellison (2002), Swingland (2003), and Rosenzweig (2003). Pierce et al. (2005) provide a case illustrating how conservation priorities can be mainstreamed into land-use planning through interpretation of scientific products into user-friendly, user-useful maps and guidelines. In addition, Knight et al. (2006) describe how mainstreaming can be integrated into a framework for implementing actions aimed at securing conservation priorities. The latter two are examples of conservation actions that enable or facilitate the restoration of natural capital by identifying restoration priorities.

Conceptual Framework for Mainstreaming the Restoration of Natural Capital

A conceptual framework for restoration boils down to identifying a model of the desired landscape; in other words, what mix of land uses and economic flows are required to meet the needs of different stakeholders (Salafsky and Wollenberg 2000)? Restoration implemented in an ad hoc manner is likely to fail in achieving desirable outcomes (Hobbs and Norton 1996; see also chapter 3), as has been shown for the ad hoc implementation of other conservation actions, such as the location of protected areas (Pressey 1994). Therefore, prior to restoration intervention, stakeholders need to identify an appropriate landscape model characterized by requirements for sustaining biological patterns and processes, and for supporting human needs. Effective restoration requires explicit goals and targets (e.g., Hobbs and Norton 1996) identified in a way that is consistent with a specific landscape model.

For the mainstreaming of restoration to happen, the landscape model must facilitate the identification of plausible and compelling win-win scenarios. Thus, farmers must be convinced that the direct and opportunity costs of restoring native, natural capital on their farms will be outweighed by the benefits of such restoration, for example, in enhanced production through improved pollination services or reduced soil erosion (e.g., Kremen and Ostfeld 2005). Similarly, restoration interventions aimed at achieving nature conservation goals should be guided by the achievement of explicit and defensible targets for biodiversity features, which are set in the process of systematic conservation planning (Pressey et al. 2003).

While there has been much written on conceptual frameworks, goals, and targets for restoration (e.g., Hobbs and Norton 1996), the obvious link between restoration and the systematic, target-driven, conservation planning of landscapes (Margules and Pressey 2000) has only recently been made (e.g., Pressey et al. 2003; Crossman and Bryan 2006). Systematic conservation assessments identify those areas of transformed or degraded natural capital that are required to achieve targets for the conservation of both biodiversity patterns (e.g., species, land classes) and processes (e.g., migration corridors). These areas then become defensible priorities for restoration, as illustrated by Crossman and Bryan (2006) for agricultural landscapes in Australia.

A similar systematic approach is required for the restoration of natural capital for ecosystem service delivery (e.g., Kremen and Ostfeld 2005; Pierce et al. 2005). A few conservation assessments have targeted and incorporated the spatial components of ecosystem services (e.g., Rouget, Cowling, et al. 2003). However, a great deal more research is needed before we can make significant progress in the restoration of natural capital: (1) the natural capital— both intact and degraded—in a particular planning domain needs to be identified and mapped in consultation with those stakeholders who are direct beneficiaries of the services it delivers; (2) the benefits derived from these services and their flows to specific beneficiaries need to be quantified and displayed in ways that are meaningful to stakeholders; (3) targets need to be set for each component of the region's natural capital in a way that is consistent with a landscape model (for example, a certain number of hectares of healthy watershed are required to ensure a sustainable water supply over a specified period); (4) target shortfalls should be identified as priorities for restoration; and (5) mechanisms should be sought to mainstream the restoration of these areas into those sectors that benefit from the services provided by the natural capital.

The major advantage of systematic restoration to achieve the goals for a specific landscape model is that it is target driven and, therefore, defensible, efficient, and effective (Crossman and Bryan 2006). These attributes are likely to greatly facilitate mainstreaming, especially when the actors are cash-strapped government agencies or profit-motivated corporations.

An Operational Framework for Mainstreaming the Restoration of Natural Capital

Cowling et al. (2002) developed an operational framework for mainstreaming biodiversity, informed by eleven South African case studies presented in Pierce et al. (2002). The framework is sufficiently broad to accommodate restoration interventions; along with the establishment of protected areas and effective soil and water conservation, restoration is another tool for conserving biodiversity. The framework comprises four major components:

1. *Prerequisites* essential for mainstreaming to take place
2. *Stimuli*, external and internal to the sector, that catalyze awareness of the need for mainstreaming
3. *Mechanisms* that initiate, enable, or drive mainstreaming
4. *Outcomes* that are measurable indicators of the effectiveness of mainstreaming

In the framework, the mainstreaming process was described as follows: "Given that certain prerequisites are in place, a set of specific stimuli can catalyse activities which then lead to the identification of appropriate mechanisms, with the net result that effective mainstreaming, as measured by outcomes, will happen" (Cowling et al. 2002).

River Futures

An Integrative Scientific Approach to River Repair

Edited by Gary J. Brierley and Kirstie A. Fryirs

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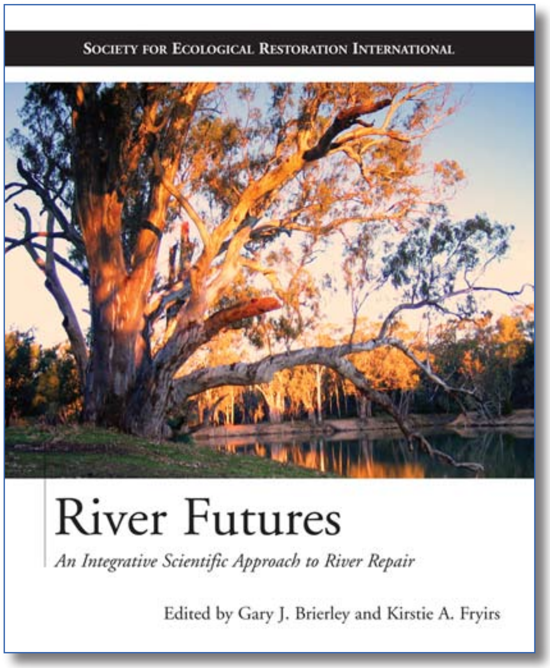

Contents

About This Excerpt

River Futures provides a holistic overview of considerations that underpin the use of science in river management, emphasizing cross-disciplinary understanding. This excerpt from chapter 8 introduces the notion of "connectivity" as an integrating theme, relating biophysical notions of landscape and ecosystem connectivity (or disconnectivity) to social relationships to place.

From chapter 8, "Social and Biophysical Connectivity of River Systems," by Mick Hillman, Gary J. Brierley, and Kirstie A. Fryirs

Place is the location . . . where the social and the natural meet.

—Dirlik 2001, 18

Successful integrative river management requires an understanding of the links between natural and cultural landscapes, ensuring that institutional and community values are meaningfully incorporated in the process of environmental repair (Harris 2006). Coherent approaches to the assessment of river health integrate biophysical and social dimensions of environmental condition, building on the relationship between healthy rivers as products of, and in turn promoting, healthy societies. Understanding and working with the concept of connectivity across both the biophysical and social dimensions is a core component of this relationship. A connected approach to integrative river management aims for a dialogue between scientific understanding and community values. Applying this principle therefore means understanding both catchment-scale biophysical linkages and community perceptions of what constitutes a "healthy river." Such understandings are specific to time and place, militating against the application of generic models and assumptions.

Contemporary perceptions of river health are contingent upon present and past connections between people and their river. They encompass a range of potential uses and values. However, in the large body of literature influenced by Eurocentric ideas of landscape, healthy rivers are often romanticized as single, continuous, constantly flowing channels (Kondolf 2006). Postcolonial societies have afforded rivers a limited range of uses, and systems are expressed as healthy only if conditions suitable for these uses are maintained. Disconnection in a river system, whether it is the presence of isolated pools in river channels or ephemeral tributaries, has been portrayed as an undesirable and unsustainable state (Kondolf et al. 2006). This contrasts with the recognition of variable and changing forms of connection and disconnection in many indigenous cultures (Smolyak 2001; James 2006). Recognition of the variable and changing patterns of connectivity across time and space, and between social and biophysical dimensions, is a core component of integrative river management.

Whether appraised in biophysical or social terms, landscapes, ecosystems, and communities can be relatively connected or disconnected. For river management to be successful and relevant it is important to recognize that biophysical disconnection may be natural and healthy at a given time and place. For this reason we use the term (dis)connectivity to refer to dynamic patterns of connection and disconnection. For example, disconnected and isolated parts of river systems may shelter distinct genetic populations of species and unique floristic and faunal attributes (Sheldon et al. 2002; Bunn et al. 2006). Likewise, human (dis)connectivity with rivers has often been mediated by cultural factors, particularly in indigenous societies through taboos, totems, and sacred sites that are the result of co-evolution with landscapes over many generations (Rose 1999; Townsend et al. 2004). However, in more recent history, biophysical disconnection has often been imposed through the construction of barriers, while social disconnection has resulted from appropriation and enclosure of riparian land or in response to rapidly developing perceptions by local communities of the river as polluted, unhealthy, or as bringer of damaging floods. Present day disconnection is often the legacy of an earlier focus on narrow and exclusive uses of the river for irrigation, discharge of effluent, or navigation. This type of disconnection is referred to by Ward (2001) as "geo-environmental disconnection," the product of technocentric efforts to forge landscapes for agriculture, industry, and recreation. Conversely, biophysical and social connections may have been imposed through the development of irrigation systems in semi-arid landscapes.

In this chapter, we argue that imposed, arbitrary (dis)connectivity based on a narrowly defined or exclusive use of water is unhealthy in river management and is ultimately unsustainable and unjust. Such (dis)connectivity reduces community understanding of, and concern for, our rivers, while allowing dominant and environmentally damaging uses and practices to continue. It also promotes feelings of inequity, distributing costs and benefits through top-down decisions or decrees. On the other hand, broad and holistic (dis)connectivity in its many forms strongly implies the convergence of social and biophysical perspectives and acknowledgment of a wider range of values as a vital step in the process of river repair. Transdisciplinary work on links between ecological and community health and well-being indicates that the healthy appreciation of the inherent diversity and variability of river systems is an integral part of healing our relationship to the natural world (Costanza and Mageau 1999; Connor et al. 2004). Based on these guiding principles, this chapter uses a transdisciplinary and place-based analysis of biophysical and social (dis)connectivity to:

1. Examine the broad links between connectivity and river health.
2. Describe the biophysical and social forms and changing patterns of connection and disconnection within a river system and between people and that system.
3. Analyze key themes in the interrelationship between the biophysical and social dimensions of (dis)connectivity through case examples.
4. Highlight implications of these themes in the development of just and sustainable approaches to the management of healthy rivers.

Connectivity and River Health

The view of river health outlined in chapter 7 focuses on external, biophysical, and verifiable indicators of river condition. Often such indicators are specific to particular disciplines and scales. However, given that rivers epitomize the links between landscapes and ecosystems (Jungwirth et al. 2002), a practical understanding of biophysical linkages is crucial in producing the "mature knowledge" that is increasingly required for effective integrated ecosystem management (Lake 2001; Dunn 2004). This is in itself a major challenge, since complex indicators of river condition, such as connectivity itself, have proved difficult to quantify both for conceptual reasons and because of scientific concern over valid descriptors and rigor of resultant data (Dunn 2004).

A complementary but distinct view of river health is one based on the idea of social connection, in which health is about people developing, maintaining, or losing interaction with the river. Integral to thinking about the way people connect and disconnect with rivers is the broad geographical notion of place as socially constructed (Massey 2005), and of a sense of place as individually interpreted rather than having one particular "essence." The notion of "place-identity" has been used to describe this social dimension of connection: "it can be said to represent the physical settings' importance for a person's identity. Research in place-identity suggests that an individual has more complex relations to the environment than simply living in it" (Wester-Herber 2004, 111).

Connection through place-identity is fundamental to community engagement in river management programmes, fostering a sense of commitment, building social capital, and allowing local knowledge a role in planning (Hillman et al. 2003; Thompson and Pepperdine 2003). Place-identity also highlights the need to think of health in terms of both biophysical indicators and human-nature relationships (Brierley et al. 2006b). However, the establishment of place-identity is a necessary prerequisite rather than an inherently sufficient condition for river health—there is no reason to argue that some good use-values create place-identity and others do not. Our point here is that without such connection, the relationships that sustain integrative river management cannot be forged and that narrow and exclusive values will prevail.

The nexus between biophysical and community health in major river rehabilitation strategies has been expressed in practice in several forms. For instance, the Loire Vivante (Living Loire) program aims to "lead people back to the river," while the Mersey Basin Campaign includes in its vision the goal of increasing the valuing of the river by its community. The Dutch policy of "Space for the River" aims to maintain flood protection in the face of increased design discharges, while at the same time conserving landscape, ecological, and historical features (Cals and van Drimmelen 2000). The Victorian River Health Strategy includes the objective of "maintaining the rivers' place in our collective history" with the overall aim that "our communities will be confident and capable, appreciating the values of their rivers, understanding their dependency on healthy rivers and actively participating in decision-making" (Victorian Government 2002, 3).

The next sections develop a conceptual framework for exploring forms of biophysical and social (dis)connectivity in river systems, providing a lens on this integrative approach and on a condition-connection notion of health.

Large-Scale Ecosystem Restoration

Five Case Studies from the United States

Edited by Mary Doyle and Cynthia A. Drew

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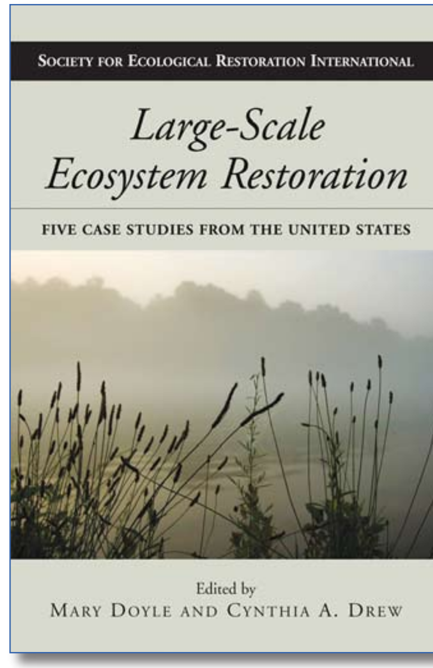

Contents

About This Excerpt

Representing a variety of geographic regions and project structures, the case studies in this volume shed light on the central controversies faced by large-scale ecosystem restoration from political, ecological, and economic perspectives. In this excerpt from chapter 6, Stephen Polasky introduces the complex interest groups with a stake in watershed management along the Platte River.

From chapter 6, "Navigating the Shoals: Costs and Benefits of Platte River Ecosystem Management," by Stephen Polasky

The presettlement Platte River was a wide, shallow, muddy, slow-moving river with complex, braided channels. Early attempts to use the river to transport furs and other goods met with frustration and failure, as boats frequently ran aground and had to be dragged over numerous sandbars. Nineteenth-century humorist Artemus Ward described the Platte as "a mile wide and an inch deep" and said that it would be a considerable river if turned on edge (Willoughby 2007). Attempting to navigate a successful plan for ecosystem management on the Platte is even more difficult than attempting to navigate a fully loaded boat on the river itself. Like the Platte's physical description, negotiations to reach agreement on how to manage water flows in the Platte River are wide-ranging; often slow moving; involve complex, interconnected sets of interest groups; and have an unclear (muddy) resolution. While the Platte River has not proved to be important for transportation, it is vitally important in other ways. The Platte flows through one of the most arid regions of the country, from central and eastern Colorado and Wyoming through western and central Nebraska, before emptying into the Missouri River at the relatively wetter, eastern end of Nebraska. In dry years, the Platte is the only significant source of surface water in much of its drainage basin: "The Platte River is a consistent source of relatively well-watered habitat . . . with its water source in distant mountains watersheds that are not subject to drought cycles as severe as those of the Northern Plains" (NRC 2004, 8). As such, water from the Platte River and the habitat along the river is in high demand by people and other species. What the river is good for and how it should be managed depend on one's point of view. There are numerous complex interest groups with a stake in watershed management along the Platte; it is useful to categorize them into three main groups: (1) urban water users, (2) agricultural water users, and (3) environmental interests. The first two groups have primary interests in water uses that require withdrawal of water from the river, while the environmental interests seek to maintain natural flow regimes and habitat along the river necessary to support species, especially three federally listed endangered species: whooping crane, interior least tern, and pallid sturgeon and one threatened species: piping plover.

Urban and Agricultural Water Use

Within the Platte River Basin along the Front Range of the Rocky Mountains are rapidly growing metropolitan areas from Denver to Cheyenne. The area is attractive because of its sunny weather and the nearby mountains' beauty and recreational opportunities. In Colorado, the population living in the South Platte Basin increased by 34 percent between 1990 and 2003, from 2.3 million to over 3 million (Thorvaldson and Pritchett 2005). Rapid population growth is expected to continue. Population within the South Platte Basin in Colorado is expected to grow by almost 2 million people (a 65-percent increase) between 2000 and 2030 (DiNatale et al. 2005). More people will mean more water demands for residential, commercial, and industrial uses. Total water demand in the South Platte Basin in Colorado is expected to increase by 630,000 acre-feet per year, roughly a 50-percent increase from 2000 to 2030 (DiNatale et al. 2005).

While urban water uses are growing rapidly, agriculture remains the dominant water user in the Platte River Basin. Water is the limiting factor for agricultural production in much of the western United States, certainly the case in the Platte River Basin. Rainfall in much of the basin averages between 10 and 20 inches per year, too little to support agricultural production without irrigation. ("Dryland" agriculture, which does not require irrigation, is practiced but requires fallow years between crop production years.) Only near the mouth of the Platte in eastern Nebraska is rainfall sufficient to support annual crop production without significant irrigation.

In the western states as a whole, agriculture accounted for over 75 percent of all surface water diversions in 1990 (CBO 1997). During the same year, municipal and industrial uses accounted for roughly 10 percent of total surface diversions (CBO 1997). In the South Platte Basin in Colorado, agricultural diversions are currently 3.4 times those of municipal and industrial uses (DiNatale et al. 2005). Approximately 2 million acres of land are irrigated in the Platte River Basin (NRC 2004). While agriculture uses the lion's share of the basin's water, the economic contribution of the agricultural sector to the overall economy is small. Agriculture contributed less than 1 percent of the total value of annual revenues from agricultural products in the South Platte Basin in Colorado (Thorvaldson and Pritchett 2005). Agriculture constitutes a higher percentage of the economy in Nebraska, 4.6 percent of total gross state product in 2006 (BEA 2007), because agricultural production is larger and the rest of the economy is smaller.

Water for agricultural use in the Platte River Basin is likely to decline over time because agriculture cannot compete economically or politically with water demand for urban uses, and overall water diversions will be limited by environmental concerns. In the South Platte Basin in Colorado, irrigated acres are expected to fall by between 133,000 to 226,000 acres from 2000 to 2030 (DiNatale et al. 2005).

As well noted by David Freeman (see Chapter 4, this volume), though diverting water from agriculture to other higher valued uses is supported by economic logic, the drying of agriculture will have negative consequences for small towns and rural areas dependent upon agriculture. Many small communities in the Great Plains have suffered from declining populations and stagnant economies for decades. Loss of water for irrigation will hasten the decline and literally drain much of the remaining vitality from these communities.

The recent surge of interest in the production of renewable energy from biomass crops (biofuels) might keep agriculture afloat in the region. The increased demand for corn from ethanol production helped push corn prices from around $2 per bushel in early 2006 to nearly $4 per bushel in early 2007 before prices fell back slightly (ERS 2007). The increased demand and higher price for corn make its production more profitable and water use for agriculture more valuable. Calls for even higher production of biofuels in the future could result in further increases in agricultural prices. In the State of the Union Address in 2007, President George Bush called for increasing renewable and alternative fuel production to 35 billion gallons by 2017. By way of contrast, ethanol production in 2006 was 4.85 billion gallons (RFA 2007. Ethanol production in the Platte River Basin, however, will likely be limited by water availability. In addition to water used for irrigation of corn and other crops, water for ethanol production amounted to 4.2 gallons of water per gallon of ethanol produced in 2005 (IATP 2006).

Old Fields

Dynamics and Restoration of Abandoned Farmland

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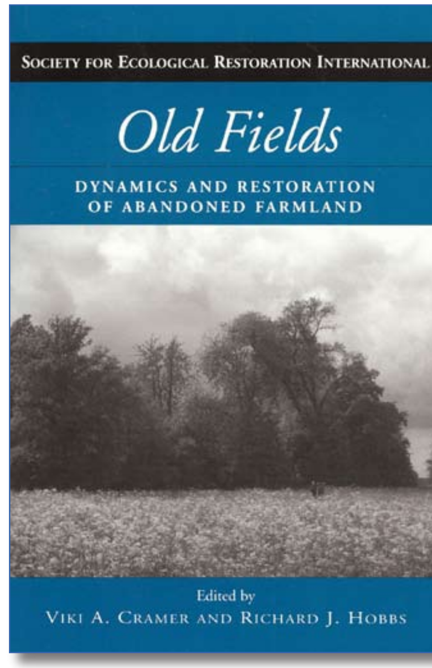

Contents

About This Excerpt

Land that has been transformed by agriculture is being abandoned in all sorts of ecosystems around the world as economics and lifestyles change. In *Old Fields*, leading experts synthesize past and current work on these abandoned lands, providing an up-to-date perspective on their ecological dynamics. In chapter 6, from which this excerpt is drawn, Karen D. Holl reviews ongoing debates in the successional literature, and evaluates three interrelated questions in the context of tropical old fields.

From chapter 6, "Old Field Vegetation Succession in the Neotropics," by Karen D. Holl

Factors Affecting Forest Succession in Old Fields

It is necessary to identify factors that affect the rate and direction of succession at a range of spatial scales in order to develop restoration strategies to accelerate or redirect successional trajectories. I categorize the factors affecting forest succession into three broad categories (Figure 6.2): the underlying abiotic gradients (rainfall, temperature, and soil), the composition of surrounding land use mosaic, and the type and intensity of past land use. I discuss in detail how each of these general factors affects the rate and direction of tropical old field succession.

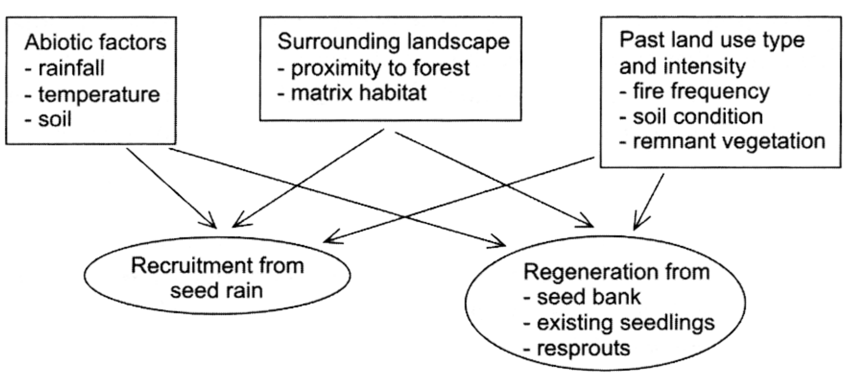

Figure 6.2: Factors affecting (boxes) different modes of regeneration (ellipses) of tropical old fields.

Underlying Abiotic Gradients

Much of the difference in the rate and direction of succession in neotropical old fields can be explained by anthropogenic factors, such as surrounding land use and land use history (discussed later), but these factors are overlain on a number of abiotic gradients, primarily rainfall, temperature, and soil type, which strongly influence forest recovery. Tropical forests fall along a rainfall gradient ranging from up to 7–8 m of rainfall evenly distributed throughout the year in the wettest sites to <1.5 m of rainfall with >6 months of dry season in the driest sites. The amount and seasonality of rainfall are primary evolutionary drivers for tropical forest plants, and they profoundly influence old field succession.

A number of tropical plant life-history traits that vary across the rainfall gradient strongly affect recovery. First, drier tropical forests have a higher percentage of wind-dispersed seeds (Ewel 1977; Janzen 2002; Vieira and Scariot 2006). For example, in a review of studies across a precipitation gradient, Viera and Scariot (2006) report 30%–63% wind-dispersed species in tropical dry forest and <16% wind-dispersed species in tropical moist or wet forest. Dry forest recovery may therefore be less dispersal limited than moist tropical forests (Janzen 2002). Second, although resprouting after disturbance is a common plant adaptation throughout the tropics, resprouting is more common in tropical dry forests, where water limitation favors an increased energy investment in roots (Ewel 1977; Vesk and Westoby 2004; Vieira and Scariot 2006). Third, seedling mortality due to desiccation is much higher in dry forests and varies a great deal interannually, depending on rainfall fluctuations (Vieira and Scariot 2006), which makes establishment from seed less predictable. Likewise, seedling growth may be slower due to lack of water. There is some evidence that dry forests may be more resilient because of lower structural complexity, greater wind dispersal, and more frequent resprouting; nonetheless, it is impossible to generalize about the rate of recovery of all forests across a rainfall gradient, given the many other factors that may influence the rate of recovery.

A second underlying, large-scale, climatic gradient is temperature. Temperature, however, is generally considered a less important driving factor in tropical forests compared to rainfall. The main effect of temperature on recovery is in higher elevation systems such as cloud forests, where the slow growth rate of trees can increase the time of recovery (Ewel 1980). Additionally, Zarin et al. (2001) found that growing season degrees (growing season length × growing season temperature) was a significant predictor of biomass across numerous old fields in the Brazilian Amazon, an area of little topographic variation, suggesting that temperatures may be an important predictor of recovery at larger scales.

A final important abiotic gradient is soil type. Much of the tropics are covered by oxisols and ultisols, which have low nutrient levels and high acidity. Some areas have more fertile, volcanic soils, such as andisols and inceptisols, although these soils commonly have low available phosphorus. Some authors (Moran et al. 2000; Zarin et al. 2001) have noted that over large spatial scales, differences in soil texture and fertility more strongly affect the recovery of biomass than previous land use (discussed later). For example, Moran et al. (2000) studied forest recovery across a range of land uses (pasture, swidden, mechanized agriculture) at five locations in Colombia and Brazil. He found that interregional variations in biomass and stand height were best explained by soil fertility, whereas within single locations previous land use explained most of the variation. Soil patterning at smaller scales can also affect species distributions. For example, Herrera and Finegan (1997) found that the different abundances of two common tree species, *Vochysia ferruginea* and *Cordia alliodora*, reflected differences in exchangeable acidity, slope, and magnesium.

Surrounding Landscape Mosaic

Gradients of climate and soil provide a template on which various anthropogenic factors act to influence neotropical forest recovery. A primary anthropogenic factor affecting succession in tropical old fields is the mosaic of surrounding land cover types, such as remnant forest, complex agroforests, shifting cultivation, pasture, or intensive agriculture (reviewed in Guariguata and Ostertag 2001; Holl 2002; Chazdon 2003, 2007). Although swidden agriculture (shifting cultivation) is important along some agricultural frontiers (Finegan and Nasi 2004), the intensity and scale of agriculture in the tropics is generally increasing. Therefore, the many regenerating sites embedded within agricultural landscapes are increasingly isolated from sources of seeds for recolonization.

Many studies in secondary growth habitats in the tropics demonstrate that seed rain and seedling establishment, particularly of large, animal-dispersed species, decline rapidly with increasing distance from the forest edge both in wet and dry forests (e.g., Aide and Cavelier 1994; Harvey 2000; Zimmerman et al. 2000; Mesquita et al. 2001). The scale over which this decline occurs ranges from within a few meters of the pasture edge up to 100 meters. Needless to say, many abandoned old fields are >100 meters from the forest edge, a distance at which there is generally minimal seed rain unless there is woody vegetation to attract seed dispersers. This isolation may be mediated by land use types, such as agroforestry, that facilitate movement of some seed dispersers within the agricultural matrix (Finegan and Nasi 2004; Harvey et al. 2004; Kupfer et al. 2004). Given that the area under secondary succession is increasing, successional forests may be the primary source of seeds of early successional species in many recovering areas (Finegan and Nasi 2004). Nonetheless, there have been few studies comparing seed rain or vegetation recovery as a function of different surrounding land uses.

The lack of seed dispersal into abandoned old fields can affect both the rate of recovery as well as the successional trajectory, and a number of authors have recorded lower numbers of individuals and species establishing in old fields farther from forest (Mesquita et al. 2001; Chinea 2002; Ferguson et al. 2003). For example, Hooper et al. (2004) found that community composition varied substantially with distance from the forest edge in abandoned pastures in Panama. However, most of these past studies on the effect of distance to forest edge on vegetation community composition have been carried out over relatively short time periods (<5 years) and longer-term studies are critical to furthering our understanding of successional trajectories.

The surrounding landscape also has substantial effects on the faunal communities, which may affect not only seed dispersal, but other common tropical plant community mutualisms, such as pollination and herbivory. There is growing evidence that changes in mammal assemblages due to hunting, isolation, or fragmentation (Dirzo and Miranda 1990; Chapman and Chapman 1995) can be detrimental to the recruitment of many neotropical trees. Large-seeded species that are animal dispersed may be particularly at risk of disappearing from fragmented areas due to loss of dispersers (Cordeiro and Howe 2003). In addition, disruption of these animal communities can cause profound changes in seed fate (Dirzo and Miranda 1990), secondary seed dispersal (Forget 1993), and seedling recruitment (Benítez-Malvido 1998). Despite the likely impacts of the surrounding land use mosaic on seed predation and seedling herbivory, it has received much less study (Holl and Lulow 1997; Duncan and Duncan 2000; Jones et al. 2003) and is a ripe area for future research on vegetation dynamics in tropical old fields.

A Guide for Desert and Dryland Restoration

New Hope for Arid Lands

David A. Bainbridge

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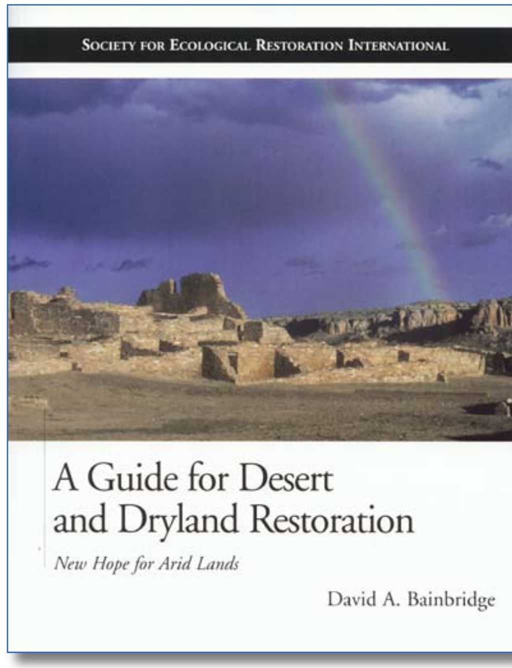

Contents

About This Excerpt

Is there hope for reversing and repairing arid lands? Dryland degradation and desertification now affect almost a billion people around the world. In *A Guide for Desert and Dryland Restoration*, David Bainbridge, who has worked in deserts for twenty-five years, offers practical, field-tested solutions to this critical problem. Written for restoration practitioners, land managers, ranchers, farmers, educators, landscapers, and foresters, this book is meant to be used in the field and on the ground to improve land and water management. In this excerpt from chapter 1, Bainbridge offers an overview of the problem of desertification and forecasts the elements of effective restoration efforts.

From chapter 1, "Desertification: Crisis and Opportunity"

The Task Ahead

Dryland restoration is needed in almost every place humans have been active past the gatherer–hunter stage (Bainbridge 1985b). Areas now considered to be desert-like in many cases were complex and productive ecosystems but were gradually or quickly destroyed by poor management. South-eastern Spain provides an excellent example of a human-made desert, and it is but one of many (Latorre et al. 2001).

The semiarid and arid areas of the world make up about 35 percent of the global land area and account for 15 percent of the world's population. In 1980 about 450 million people suffered losses in income and quality of life from degraded drylands (Table 1.2). Today these lands affect the daily lives of more than 850 million people, and every year another 1–1.5 million hectares are completely lost to production through desertification (Dregne 1986; United Nations Environment Programme 2005). By addressing the underlying causes of dryland deterioration, understanding the history of abuse and change, and applying the best restoration techniques we can begin to reverse these changes.

Globally more than 60 percent of the rangeland, 60 percent of rainfed croplands, and 30 percent of irrigated croplands are at risk for further degradation (Figure 1.6). Poor use of fragile resources has limited the ability of dryland residents to make a living, reduced their quality of life, destroyed communities, led to conflicts over land and water, reduced health and life expectancy, and severely affected natural systems and biodiversity (Lean and Hinrichsen 1992).

Land degradation in one area often affects other areas through increased flooding, reduced stream flow, and dust and sand deposition. Recent studies have even suggested a link between dustfall resulting from desertification in Africa and coral reef dieback in the Caribbean. The dust from the growing deserts in

Table 1.2: Areas affected by severe or very severe desertification (percentage)

	Rangeland		Cropland: Rainfed		Cropland: Irrigated	
	Very Severe	Severe	Very Severe	Severe	Very Severe	Severe
Africa	0.4	53.3	0.7	6.5	—	1.2
Asia	0.7	44.0	1.4	8.5	1.8	6.3
Europe	1.1	46.0	0.4	14.6	0.9	3.9
Australia	4.4	8.4	<0.1	1.0	1.1	7.0
North America	2.1	59.0	0.2	1.0	1.0	3.5
South America	3.9	47.2	0.6	2.6	0.7	3.7

western China has circled the globe, with largely unknown affects on ecosystems where the dust, fungal spores, and nutrients are deposited.

Restoration and improved management of these resources are essential in reversing the process of degradation and desertification. Restoration may include a wide range of interventions, from surface shaping to soil amendments, tillage and weed removal, seeding, planting, irrigation, and aftercare. Low-cost revegetation efforts can help restore productivity to degraded grazing lands. On severely damaged sites the establishment of any species may be difficult, and even the growth of weeds may be considered a success. *Rehabilitation* and *reclamation* are used to describe efforts to return sites to use in stable condition but often with introduced species and less complexity than a restoration project. Ecological restoration tries to restore ecosystem function (how things work) and structure (how things look) to match undisturbed reference sites. This can be very difficult because our understanding of these dryland systems is limited and the environmental variability is very high, characterized by pulses rather than steady progress even in undisturbed and "stable" conditions.

Restoration Efforts

Work on dryland restoration began in earnest in about 1980, although projects in the Southwest were started as early as 1900 and many were attempted in the 1930s (Griffiths 1901; Cox et al. 1982). Without a solid scientific base and without controlled experiments and research, progress was limited. Many flawed and inappropriate approaches were used over and over because managers were ignorant of what other people had done in earlier trials. Research efforts were also hampered by a very limited understanding of arid and semiarid ecosystems (Hall 2001). Most researchers and land managers had come from more humid areas where vegetation did recover naturally, and few studied the lessons from other drylands around the world.

Although restoration is desirable for biological, economic, social, and aesthetic reasons, it can rarely be justified with current incomplete economic accounting practices. This has also hampered research and limited trials and demonstrations to short-term studies. Funding for long-term research has been rare and support for integrated research rarer still. Why study the effects of spending $5,000 an acre for a potential grazing return of less than $50 a year?

Restoration research also provides us with an often humbling opportunity to test and refine our application and understanding of ecological processes and theories (Bradshaw 1992b). Although the roots of the science of restoration ecology can be traced back to the late 1800s in Europe, Aldo Leopold and his work at the University of Wisconsin during the Great Depression laid the foundation for environmental restoration in America. Although some excellent work was done on recovery after disturbance in the 1970s and 1980s (Vasek et al. 1975; Lathrop 1983b; Prose and Metzger 1985), modern interest and commitment to research and publication improved rapidly after the Society for Ecological Restoration (SER) International held its first annual conference in 1989 in Oakland, California. This conference included a few papers on dryland restoration. In 1993 I led the first SER workshop on dryland restoration at Red Rock Canyon State Park in California with Ray Franson and Laurie Lippitt (Figure 1.7). This hands-on training is critical in improving understanding and management.

Many other projects were started about this time throughout the Southwest, and I have learned much from their successes and failures. My colleagues, staff, and students have also contributed a great deal to my understanding. This research and testing have led to improved techniques and approaches for desert restoration, but it will never be easy.

A multidisciplinary approach is very desirable for restoration work. Ideally a team will be able to provide insight about not only the plants, soil microorgan-

Desertification

Figure 1.6: World desertification map, with severity indicated by shading.

isms, insects, animals, reptiles, and birds but also the current and historic national and international events that have determined the economic incentives and pressures that influence land managers. Restoration begins with a clear understanding of environmental history, current conditions, and the decision-making environment and leads through planning, funding, and implementation to maintenance and monitoring.

Setting appropriate restoration targets often is a critical issue and is discussed in more detail in Chapter 5. Current laws often require land to be restored only to the condition it was in before the latest insult and injury. This could mean restoration to alien grasses and weeds in much of the western United States. Others think, as I do, that restoration should consider conditions before overgrazing began, when the land was managed by native people. This may mean before 1800 in California and before 1700 in New Mexico. We must remember that this means a return to a different management scheme, not a time when there was no human manipulation of the environment. Most of these lands were occupied and managed for thousands of years, often successfully, before the arrival of the Europeans. If we want to go back before human intervention in land use, we would be back 25,000 years or more in California and perhaps 250,000 years in Australia.

The native people of what is now called southwestern North America were intelligent, skilled, and knowledgeable applied ecologists who actively managed the land and shaped its ecosystems (Lawton and Bean 1968; Shipek 1990; Anderson 1996). Although they did not have bulldozers, tractors, and transnational seed companies they were skilled in the use of fire; kept animals; hunted some species very effectively; selected, transported, and planted seeds for annual and perennial crops; and transplanted trees and shrubs. A better knowledge of their management practices can explain many biological mysteries, and it will also help improve our land management and protection of rare and endangered species and ecosystems (Figure 1.8).

Restoration Is Possible

Successful restoration requires a systematic, holistic view of the interactions between humans and the environment through time. The most appropriate resto-

Figure 1.7: The first SER desert restoration class, hosted by Red Rock Canyon State Park, California. Hands-on learning is best for restoration training.

A)

B)

Figure 1.8: Lessons from the past: (a) Terraces and (b) reservoir at Mesa Verde, Colorado, demonstrate proven methods for soil and water conservation in arid lands.

ration approach for a given site depends on the type of disturbance, the degree of disturbance, the causes of the disturbance, the available budget (both time and labor), and the goals and speed of recovery desired. The effort should always include consideration of ecosystem structure and function.

Traditionally structure has often been favored (how many plants, what kinds) over function (water flow and retention, nutrient cycling), but repairing function usually is more important. Like many others, I originally began with a focus on returning specific plants, but after a few years it became clear that restoring function was more important. If ecosystem function is restored it will hasten recovery and ensure that the environment continues to improve after the restoration team leaves.

The cost for restoration can range from a few hundred dollars to $20,000 per acre or more. As the investment increases the rate of recovery will increase, but even large expenditures are no guarantee of success in these extreme environments. Everything has to be done correctly at the right time, and water usually has to be provided for initial establishment.

The cost of failing to restore damaged drylands is high, biologically, socially, and economically. Yet until recently the value of the ecosystem services provided by nature has not been counted, and an economic justification for restoration was lacking. The functions of flood control, water purification, oxygen production, and dust control do matter, and they do have value. When Robert Dixon compared the cost of a big flood in Tucson with the cost of dryland restoration, which would have lessened or prevented the flood, the restoration would have cost less. As nature's services are more completely and clearly valued, restoration will become more common as an economic investment. It will always remain an important activity for beauty, biodiversity, recreation, and quality of life. Restoration also provides invaluable opportunities to test our theory and knowledge about how ecosystems function (Bradshaw 1992b).

The magnitude of the task globally can be calculated from the vast areas of land needing treatment. A modest restoration program to repair just 10 percent of the desertified rangelands in the United States could cost $36 billion a year, about 12 percent of the current budget for the Department of Defense. To treat 10 percent of the 3.6 billion hectares of degraded drylands in the world each year would cost about $360 billion dollars, about 4 percent of the annual gross domestic product of the United States. Sadly, the countries that are worst affected are least able to pay, with many struggling to pay immense debt services on loans from rich countries. In many cases their total external debts are more than three times their total foreign exchange earnings. The ongoing effort to relieve some of this pressure is critical because these high debt service requirements often drive resource mismanagement.

RESTORING THE PACIFIC NORTHWEST

Restoring the Pacific Northwest

The Art and Science of Ecological Restoration in Cascadia

Edited by Dean Apostol and Marcia Sinclair

Foreword by Eric Higgs

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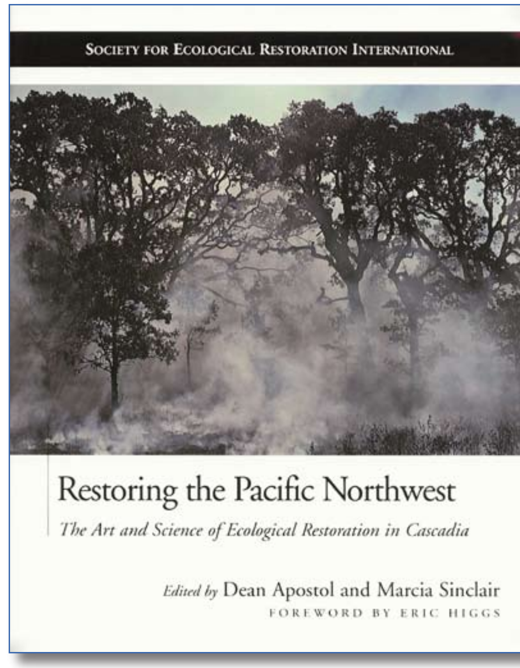


Contents

About This Excerpt

The Pacific Northwest, stretching along the west coast of North America from northern California through British Columbia to southeast Alaska, is a "hotspot of boggling diversity in restoration projects," according to Eric Higgs, chair of the board of directors of the Society for Ecological Restoration International from 2001 to 2003. *Restoring the Pacific Northwest* brings together fifty-seven experts and practitioners to showcase nine seminal habitat types, six distinct restoration approaches, and more than three dozen case studies. It is an essential handbook and encyclopedic overview for restorationists and practitioners around the world. This excerpt from chapter 17 offers key concepts in traditional ecological knowledge.

From chapter 17, "Traditional Ecological Knowledge and Restoration Practice," by René Senos, Frank K. Lake, Nancy Turner, and Dennis Martinez

Key Concepts in TEK Restoration

Kincentricity

A key concept in the indigenous world view is kincentricity , or a view of humans and nature as part of an extended ecological family that shares ancestry and origins (Salmón 2000, Martinez 1995). The kin or relatives include all the natural elements of an ecosystem; indigenous people sometimes refer to this interconnectedness as "all my relations. " Kincentricity acknowledges that a healthy environment is achievable only when humans regard life around them as kin.

Kinship with plants and animals entails familial responsibilities; it tells us why we are on this earth and what our ecological role or niche is vis-à-vis our relatives in the natural world. T o put it another way, it tells us that we are a legitimate part of nature, that we have responsibilities within nature, and that in exercising those responsibilities we are as "ecological" or "natural" as any other species. The indigenous land ethic holds that we can have a positive restoration effect in the very act of using natural resources. Kincentric ecology entails direct interaction with nature to promote enhanced ecosystem and cultural functioning. This is what sustainable practices are all about.

Our responsibility for participating in the "recreation of the world" (as tribes on Klamath River in northwestern California call it) is never finished. Periodic intervention by humans in nature has long been part of ecosystem dynamics in the Pacific Northwest. Given the myriad ecological catastrophes we now face, the need for active restoration will only increase. There are no finished restoration projects. Nature has self-healing powers, but these engage only after a specific harmful disturbance (e.g., a dam or invasive species) is removed or modified. Humans also can lend a hand by restoring missing species, modifying structure, and so forth. As the SERI *Primer* notes, continuing management is necessary to "guarantee the continued well-being of the restored ecosystem thereafter" (Society for Ecological Restoration 2004:6).

Pioneering Western ecologists and restorationists have come to similar conclusions with respect to kincentricity, starting with Aldo Leopold (1949) and his land community ethic, which posited that an individual is a member of a community of interdependent parts and that each citizen is ethically bound to maintain cooperative relations with the biotic community. Restorationist Stephen Packard discovered that recovering degraded Midwest landscapes required corps of dedicated volunteers to restore several thousand acres of tallgrass prairie and oak savanna (Stevens 1995). This ambitious endeavor was not possible until people invested in the r home places. Environment al philosopher Andrew Light (2005) has explored the personal, moral, and environment al dimensions of making amends to our kin through the act of restoration and considers ho w restoration provides a venue for ecological citizenship. Kincentricity provides a basis for considering restoration as a process of engagement with nature, a way to sustain or repair relations with the living world. In doing so we develop viable cultural, economic, and ecological practices that support and nurture our shared environment.

Reference Systems

When addressing the needs of restoration today, whether at the landscape, habitat, or species level, it is important to recognize indigenous peoples as an influence in shaping and maintaining the historical condition of many different ecosystems. In this sense, the effect of past indigenous management practices should be considered part of the reference ecosystem, or more generally as providing a set of reference processes to guide a restoration effort. In the Northwest, reference conditions influenced by indigenous land use practices of a pre-European era are the benchmark.

Any reference condition or design of future desired conditions must account for humans' use and management of the environment. The scale and intensity of human use and management of the environment are important to successful ecocultural restoration and to establishing a sustainable relationship to place. Our reference window is at least as large as 10,000 years, or the postglacial Holocene period, with particular attention paid to the last 4,000 years, during which a gradual cooling trend occurred that most resembles present climatic and ecological conditions (Figure 17.2). The last 10,000 years also is the period during which humans have exerted the most influence on North American ecosystems (Egan and Howell 2005).

The use of reference ecosystems in restoration is not without controversy. It can be expensive and time-consuming to use multiple disciplines and indirect proxy lines of ethnographic and scientific evidence to

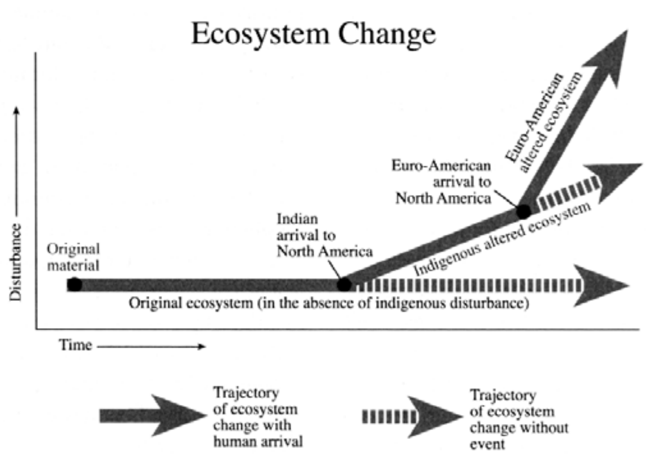


Figure 17.2: Ecosystem trajectory showing that humans represent an ecological force on the landscape.(From Lewis and Anderson 2002. Copyright © 2002 by the University of Oklahoma Press, Norman. Reprinted with permission. All rights reserved.)

establish a reasonable probability of accuracy in identifying a site at a specific point in history (see Egan and Howell 2005 for technical information regarding reconstructing historical ecosystems).

Some restoration scientists and practitioners question the value of using historical baselines to guide restoration. Not only can it be difficult to reconstruct historical ecosystems because of severe changes in some environments, but why go back to an arbitrary point in history? Why choose, for example, a time before European contact as the reference? Why not just try to improve the function of degraded ecosystems? Isn't nature constantly changing?

However, TEK does not advocate that we stop change and freeze ecosystems in a particular time frame or that we recreate a snapshot in time. After all, present conditions are also a snapshot in time. What we really need to do is connect the past with the present in order to reveal what kind of trajectory an ecosystem may be on and then nudge that trajectory just enough to restore key functions. History and function, then, are inseparable. Both Higgs and Martinez have written and spoken about balancing historical fidelity or authenticity with ecological functionality. Instead of fixing a snapshot in time, we need to rerun a moving picture, played out within boundaries determined by historical trends in disturbance regimes, including the kinds, intensities, and frequencies of disturbance with which an ecosystem has evolved.

For example, forest stand-level restoration projects are subject to constraints imposed by the greater landscape scale. Although a reference ecosystem can guide initial restoration efforts, these will be modified by larger landscape considerations (e.g., fragmentation, fire hazard, exotic species invasion, species losses), or even larger phenomena such as climate change. Anchoring the reference model in real historical time will help us to recover key features of ecosystem structure, composition, and processes within natural variability , with a look to designing the future desired condition.

Building sound reference models for ecocultural restoration requires the best Western science and the best of TEK, not one or the other . Each has the potential to reinforce the other and to compensate for inherent methodological limitations by considering history and function, quality and quantity, long term and short term, culture and ecology, economy and environment.

Successional Theory and Disturbance

Traditional ecological knowledge complements contemporary knowledge of fire ecology by providing information about historical and contemporary applications of fire by indigenous people, including fire effects on wildlife and vegetation in different environments. Indigenous knowledge of fire ecology includes but is not limited to variations in fire frequency, intensity, severity, and specificity of areas burned in different ecosystems or plant communities by indigenous people or by lightning ignitions. TEK provides knowledge about fire effects and ecosystem responses and about how physical and biological processes such as hydrology and forest succession respond to fire over time (Lewis and Anderson 2002).

Integrating multiple knowledge systems to understand the effects of fire on the remaining post- treatment vegetation or soils can lead to greater accomplishment of objectives. Thinning and spring season pile burning may be intermediate steps that help prepare the site for the reintroduction of fall season low-intensity burns that emulate Indian fire (Williams 2000). Ethnographic information and TEK may be instrumental at each treatment step, especially when one is considering restoration effects on wildlife, food plants, or nontimber forest products, resources that hold high social and ecological value to local communities (Anderson 2001).

Restoring and maintaining biocultural diversity of the landscape through integrated restoration planning involves an interdisciplinary as well as a multicultural approach. Fuel reduction projects that incorporate Indian fi re will have higher levels of success in restoring and maintaining biodiversity , which in turn will support cultural diversity and local communities. This premise may hold true especially with native cultures that historically and currently rely on fire and fire-dependent landscapes for their sustenance and cultural survival (Boyd 1999). A community forestry approach can help local communities cope with likely future changes caused by climate change and intensified demands of natural resources.

Defining Scale

Issues of ecological and social scale are important considerations in restoration work. Any restoration program directed toward a given geographic area must carefully define the scale at which it will operate. For example, will projects focus on a single species or population, a particular habitat, or an entire watershed? Will restoration engage the collaboration of an individual, a community, or a national organization or institution?

Indigenous ways of understanding and relating to the environment provide useful models for framing restoration efforts at the appropriate scale. In coastal Pacific Northwest environs, individual families traditionally were responsible for a particular resource base at a specific location (e.g., shellfish beds). Villages were organized around specific places along stream reaches, and an affiliated tribal group (distinguished by common linguistics) managed a given bioregion (J. James, personal communication, 2001). Appropri-

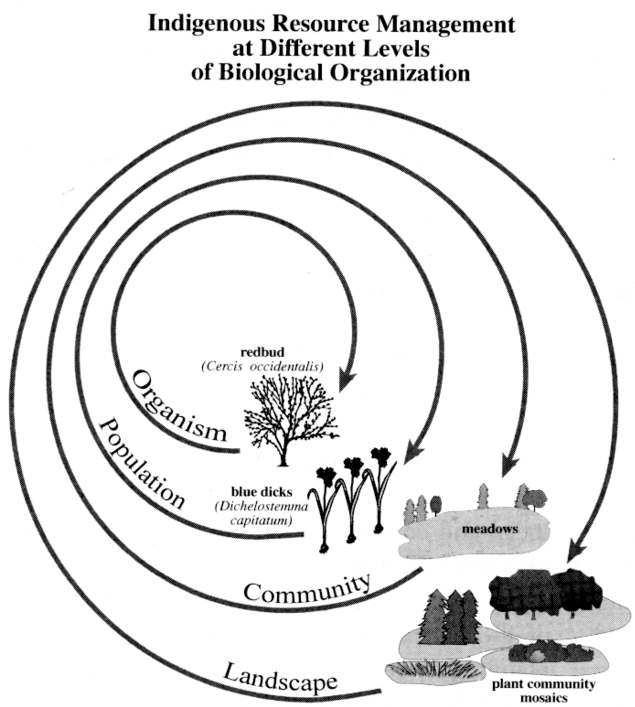


Figure 17.3.

Scale of potential management effects. (From Lewis and Ander son 2002. Copyright © 2002 by the University of Oklahoma Press, Norman. Reprinted with permission. All rights reserved.)

ate technologies and resource management practices were ritualized to maintain healthy functioning of the system at all social and ecological scales.

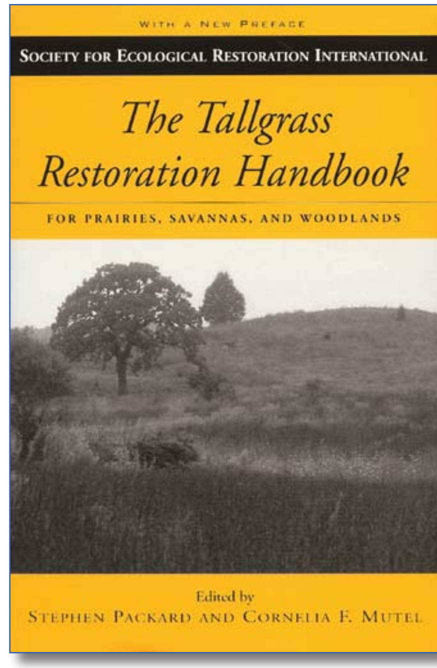
Defining the scale of operation provides context for our individual actions linking with others' actions across or up in scale. TEK provides an operational framework that addresses the integration of the various ecological and social scales, situated within temporal scales (Figure 17.3; Berkes et al. 2000). The perspective of scale can also be reflective in that the strengths and weaknesses of TEK and Western science are evaluated in the context of the restoration program or projects being planned, implemented, or monitored.

The Tallgrass Restoration Handbook

For Prairies, Savannas, and Woodlands

Edited by Stephen Packard and Cornelia F. Mutel

Foreword by William R. Jordan III



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Index, appendix

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Part II. Goals and Plans

Part III. Seeds and Planting

Part IV. Management and Monitoring

Part V. Protecting, Restoring, and Monitoring Animals

Appendixes

About This Excerpt

In the 1930s, researchers at the University of Wisconsin-Madison Arboretum launched experiments aimed at learning how prairies work, and the field of ecological restoration was born. *The Tallgrass Restoration Handbook* offers the excitement and perspective of the trial-and-error, hands-on work of prairie restoration. Written by practitioners for practitioners, this is a practical manual of the art and science of prairie, savanna, and oak woodland restoration. In this excerpt, Virginia Kline introduces us to the living community that makes up this "sea of grass and orchards of oak."

From chapter 1, "Orchards of Oak and a Sea of Grass," by Virginia M. Kline

The Tallgrass Prairie

What Is a Prairie?

French explorers called it *prairie*, taken from a French word meaning "meadow," and early settlers adopted that name for this unfamiliar New World grassland, for which there seemed to be no appropriate word in English. Those who experienced the prairie first-hand had no need for a precise definition of the term; that would come much later, after a fledgling science acquired its own new name: ecology. John T. Curtis, the first ecologist at the University of Wisconsin, undertook with his students the task of delineating and characterizing each of the major biotic communities of Wisconsin. Based on their extensive field studies, Curtis's book *The Vegetation of Wisconsin* was published in 1959. In it Curtis defined a prairie as an open community, dominated by grass, and having less than one tree per acre. In setting this limit, he cautioned that this was an arbitrary distinction for the convenience of those studying a continuum of vegetation. Nature seldom draws lines; one community is likely to blend into the next.

Plants of the Prairie Community

Prairies are rich in species, and the grasses, composites, and legumes are especially well represented. (See appendix A for a listing of tallgrass prairie vascular plants.) The particular group of species present depends on geographic location, since some species' ranges are limited to certain areas within the prairie region; it also depends on local topography and soil. Prairies near the northern limits of the prairie region differ in composition from those farther south, while within the same geographic area, prairies on high rocky hill slopes, sand terraces, deep silt loam soils, and poorly drained lowlands also differ from one another. In Wisconsin, characteristic grasses of the drier prairies include little bluestem, prairie dropseed, and side-oats grama. Sites with deep silt loam soils are dominated by big bluestem and Indian grass, while wetter sites have blue joint grass and prairie cord grass. (See Figure 1.1.) Disturbance and fire history influence composition as well; for example, some species require soil disturbance, such as that provided by bison wallows or animal burrows, to get started, and some species do best when fires are frequent or occur at a particular season.

Prairie plants grow close together, sharing available resources in time and space. Some species flower early in the season, some in midsummer, some in fall. Thus not all the species have their most rapid growth phases at the same time; instead they take turns. Early growers tend to be short, and height tends to increase over the growing season, culminating in the tall grasses in early fall. However, some species that bloom later than the tall grasses, including goldenrods, asters, and gentians, are shorter than the grasses, taking advantage of the increased light levels as the grass leaves turn fall color. Beneath the surface, roots of different shapes, sizes and depths divide the space.

Adaptations of the Plants

Each species is adapted to the extreme temperatures, drought, wind, high light intensity, fire, and grazing that are part of the prairie plant's environment. Some of the morphological adaptations easily observed on the prairie include finely divided or narrow vertical leaves to prevent overheating by the sun and offer less resistance to the wind, and leathery or waxy leaves to reduce water loss. Unseen are the extensive root systems that make up two-thirds of the total plant biomass—an adaptation that helps maintain a favorable water balance and allows rapid regrowth after fire or grazing. Buds located at or below the ground surface are important for resprouting, as is the ability, in grasses, to regrow from nodes low on the plant.

Although we often consider wind and high light levels as factors that plants must withstand, wind and light are important resources as well. Many prairie species take advantage of wind for pollen and seed dispersal. Many have a large amount of leaf surface (even though individual leaves may be small) to take advantage of the high light intensity, thereby increasing productivity. Many of the plants, including the warm-season grasses, carry out photosynthesis using a distinctive chemical pathway that is advantageous under the hot and dry conditions frequently encountered in prairies. This C4 pathway (so named because the first product formed is a four-carbon molecule) allows a high rate of photosynthesis at high temperatures and a higher efficiency of water use. Tallgrass prairies are among the most productive vegetation types in the world.

Fire

Fire is an important process in prairies, where it is part of a positive feedback system, the growth of prairie grass providing excellent fuel for fire and fire in turn stimulating the growth of prairie grass. Lightning can ignite a prairie fire, but during the centuries

Figure 1.1: Prairie grasses.

(Note: There are as yet no popular guides to the identification of most prairie or woodland grasses and sedges. Yet these plants are crucial to restoration. The best way to learn them is to master the technical keys and to find local botanists who can coach you. The drawings and captions here will introduce a few of the most widespread species.)

a. Little bluestem (*Schizachyrium scoparium*): Seed heads throughout top half of the stems. Leaves folded (v-shaped) in bud. Base of stem very flat. Mesic to dry soil. Thigh high.

b. June grass (*Koeleria macrantha*): Long, compact, erect seed heads. Dry, often sandy soil. Shin high.

c. Big bluestem (*Andropogon gerardii*): "Turkey-foot" seed heads. Stems often multicolored (with blue, purple, red, green, yellow, and orange). Base of stem roundish. Leaves rolled in bud. Mesic soil. Head high.

d. Indian grass (*Sorghastrum nutans*): Seed head featherlike. Distinctive auricles (lobes that hug stem at base of leaf). Mesic soil. Head high.

e. Porcupine grass (*Stipa spartea*): Needle-sharp seeds measure five to eight inches long. Dry soil. Waist high.

f. Blue joint grass (*Calamagrostis canadensis*): Wispy seed heads on three- to four-foot stems. Papery ligule around stem at base of leaf. Forms solid stands in wetlands; spreads by runners. Waist high.

g. Switch grass (*Panicum virgatum*): Open seed heads. Hairy where leaf meets stem. Wet-mesic soil. Chest high.

h. Canada wild rye (*Elymus canadensis*): Plant is pale blue-green at flowering time. Nodding heads of bristly seeds, which spread and recurve as they dry. Wet-mesic soil and woods edges. Waist to chest high.

i. Prairie dropseed (*Sporobolus heterolepis*): Very narrow leaves in dense clumps. Spherical seeds in open heads that rise well above leaves. Strong scent of buttered popcorn or hot wax. Mesic to dry soil. Seed heads waist high.

j. Prairie cord grass (*Spartina pectinata*): Coarse, rough-edged leaves tapering to very fine tips. Rows of brushlike seed heads. Wet soil. Over head high.

preceding European settlement, fires set by Native Americans were much more important. These people used fire for a variety of reasons, which can perhaps be summarized as managing their habitat. They burned frequently, possibly as often as every year.

Where the climate is suitable for trees and shrubs, fire is critical to prevent woody invasion of prairie. Fire played a major role in maintaining the mosaic of prairie, oak savanna, and oak forest that characterized the eastern boundary of the prairie. Fire also increases the vigor of many prairie species, and the year of a burn is likely to be associated with taller plants, especially the grasses, and a greater abundance of flowers. The stimulation is due to removal of the insulating litter of grass stems and leaves by the fire, which allows the soil to warm up earlier in the spring and thus increases the length of the growing season. Today managers of prairies often use carefully timed burns to help control unwanted exotic weeds as well.

Great Basin Riparian Ecosystems

Ecology, Management, and Restoration

Edited by Jeanne C. Chambers and Jerry R. Miller

Foreword by James A. MacMahon

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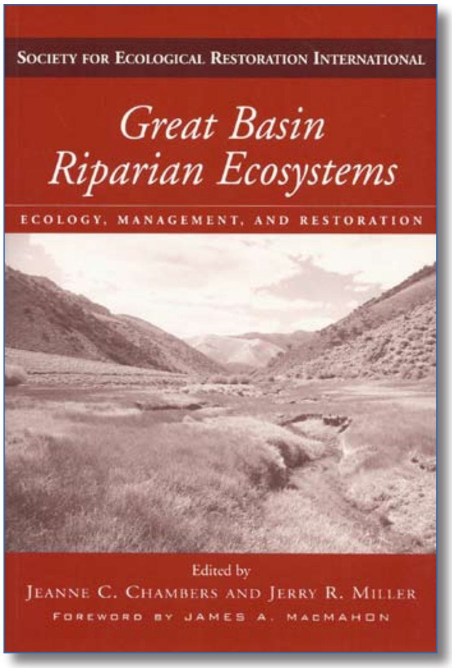


Contents

About This Excerpt

"*Great Basin Riparian Ecosystems* is not just about the ecology and restoration of Great Basin riparian areas, but also the geomorphology and hydrology of the associated uplands.... The most important lesson in these pages is that restorationists cannot escape history. The geomorphic history is critical to understanding the roles of humans, livestock, and climate change in stream channel incision, and restoring to presettlement conditions may actually result in an unstable state in a nonequilibrium landscape. The lessons recounted here have broad implications for other systems."

— Edith B. Allen, former editor of *Restoration Ecology*

From chapter 9, "Process-Based Approaches for Managing and Restoring Riparian Ecosystems," by Jeanne C. Chambers, Jerry R. Miller, Dru Germanoski, and Dave A. Weixelman

Processes Structuring Great Basin Riparian Areas

The processes currently structuring riparian areas in the central Great Basin are strongly influenced by past climates. The paleoecological records collected as part of the EM Project, as well as previous investigations, indicate that a major drought occurred in the region from approximately 2500 to 1300 YBP (Miller et al. 2001; Chapter 2). During this drought, most of the available sediments were stripped from the hillslopes and deposited on the valley floors and on side-valley alluvial fans (Chapter 3). As a consequence of this hillslope erosion, streams are now sediment limited and have a natural tendency to incise. In fact, geomorphic data indicate that over the past two thousand years, the dominant response of the streams to disturbance has been incision. The most recent episode of incision began about 500 to 400 YBP, which predates Anglo-American settlement of the area in the 1860s.

The tendency of a stream to incise depends on the sensitivity of the watershed to both natural and anthropogenic disturbance. Analysis of central Great Basin watersheds indicate that stream incision is closely related to watershed characteristics, including geology, size, and morphology, and to valley segment attributes like gradient, width, and substrate size (Chapter 4). In these semiarid ecosystems where precipitation and, thus, streamflow is highly variable both between and within years, most incision occurs during episodic, high-flow events. Since the initiation of the EM Project in 1994, high-flow events capable of producing significant incision have occurred in 1995 and 1998 (Chambers et al. 1998; Germanoski et al. 2001). Watersheds that are highly sensitive to disturbance respond to more-frequent, lower-magnitude runoff events than watersheds that are less sensitive to disturbance, The combined geomorphic and hydrologic characteristics of the watersheds determine the composition and pattern of riparian vegetation at watershed- to valley-segment scales (chapter 7). Thus, watershed attributes that characterize basin sensitivity to disturbance also have good predictive value for vegetation types and associations.

A number of the watersheds are characterized by prominent side-valley alluvial fans that influence both stream and riparian ecosystems. The fans reached their maximum extent during the drought that occurred from about 2500 to 1399 YPB (Miller et al. 2001; Chapter 3). Watersheds with well-developed fans often are characterized by stepped-valley profiles and, consequently, riparian corridors that exhibit abrupt changes in local geomorphic and hydrologic attributes. Because of the relationships among geomorphic characteristics, hydrologic regimes, and riparian vegetation, the fans also influence ecosystem patterns within the riparian corridors (Korfmacher 2001; Chapters 4 and 7). Many of the fans currently serve as local base-level controls that determine the rate and magnitude of upstream incision.

Riparian ecosystems located immediately upstream of alluvial fans often are at risk of stream incision through the fan deposits. Many of the fans have multiple knickpoints (a short, oversteepened segment of the longitudinal profile of the channel) and are subject to stream incision due to high shear stress associated with knickzone migration during high-flow events. Similarly, ecosystems located in watersheds with pseudostable channels can be degraded during catastrophic incision via groundwater sapping.

Because the stream channel serves as a groundwater discharge point, it represents the base level to which the hydraulic gradient of the groundwater system is adjusted. Stream incision lowers this base level for groundwater discharge, resulting in declines in water table levels and subsequent changes in the composition and structure of riparian vegetation (Castelli et al. 2000; Wright and Chambers 2002). Meadow complexes are presently the ecosystems most susceptible to degradation not only because they are often located upstream of alluvial fans, but also because they are subject to processes such as groundwater sapping (Chapter 5).

The rate and magnitude of stream incision in central Great Basin watersheds have been increased by anthropogenic disturbances. Because most of the streams have been prone to incision for the past two thousand years, separating changes attributable to ongoing stream incision from those associated with anthropogenic disturbance can be exceedingly complex (Chapter 3). In the cases of roads, diversions, and livestock or recreational trails, the point of initiation of stream incision often can be identified and the local effects of the disturbance on the stream reconstructed. The most direct evidence that anthropogenic disturbance has influenced stream incision in the central Great Basin is derived from ongoing studies on the effects of roads on riparian areas. Increasingly, roads are identified as major causes of stream incision and riparian area degradation across the United States (USDA Forest Service 1997; Forman and Deblinger 2000; Trombulak and Frissel 2000). In the central Great Basin, several causes of "road captures" have been documented and many others observed where streams have been diverted onto road surfaces during high flows (Lahde 2003). This diversion has resulted in increased shear stress and stream power and, ultimately, stream incision (Lahde 2003). Once initiated, stream incision often continues to occur as a result of knickpoint migration.

Assigning cause and effect to more diffuse anthropogenic disturbances such as overgrazing by livestock is more difficult. In general, overgrazing by livestock can negatively affect stream bank and channel stability, and localized changes in stream morphology often have been associated with overgrazing by livestock in the western United States (see reviews in Trimble and Mendel 1995; Belsky et al. 1999). However, data that clearly demonstrate the relationship(s) between regional stream incision and overgrazing by livestock have not been collected for the central Great Basin. In reality, it may never be possible to precisely distinguish the amount of channel incision caused by climate change from that due to anthropogenic disturbance. From a restoration and management standpoint, it is important to recognize that because particular types of streams are prone to incision, they have greater sensitivity to both natural and anthropogenic disturbances.

Anthropogenic disturbances have effects on the central Great Basin riparian ecosystems that are unrelated to stream incision. In semiarid rangelands, like those in the western United States, the general degradation of riparian areas has been attributed primarily to overgrazing by livestock (see reviews in Kauffman and Krueger 1984; Clary and Webster 1989; Skovlin 1984; Fleischner 1994; Ohmart 1996; Belsky et al. 1999). Livestock grazing influences riparian ecosystems by (1) removing herbage, which allows soil temperatures to rise and results in increased evaporation; (2) damaging plants due to rubbing, trampling, grazing, or browsing; (3) altering nutrient dynamics by depositing nitrogen in excreta from animals and removing foliage; and (4) compacting soil, which increases runoff and decreases water availability to plants. Research conducted on Great Basin meadow ecosystems (Weixelman et al. 1997; Chapter 7) and elsewhere shows that these effects can cause change in plant physiology, population dynamics, and community attributes such as cover, biomass, composition, and structure. In addition, overgrazing by livestock can result in local decreases in water quality as a result of increased sediment from road damage and road crossings (M. Amacher, unpublished data). Disturbances due to recreation, such as campsites, and vehicle and foot traffic, have increasingly widespread effects but have been poorly documented.

The cumulative effects of climatic perturbation and anthropogenic disturbance in the central Great Basin have multiple consequences for streams and their associated riparian ecosystems. There have been major changes in channel pattern and form and many streams have been isolated from their floodplains (Chapters 3 and 4). Surface water and groundwater interactions have been altered (Germanoski et al. 2001; Chapter 5), and declines in water tables have caused changes in plant species composition and vegetation structure (Wright and Chambers 2002; Chapter 7). Overgrazing by livestock and other anthropogenic disturbances have caused additional degradation of these ecosystems. The net effects of these changes have been a decrease in the real extent of riparian corridors and a reduction in habitat quantity and quality for both aquatic and terrestrial animals.

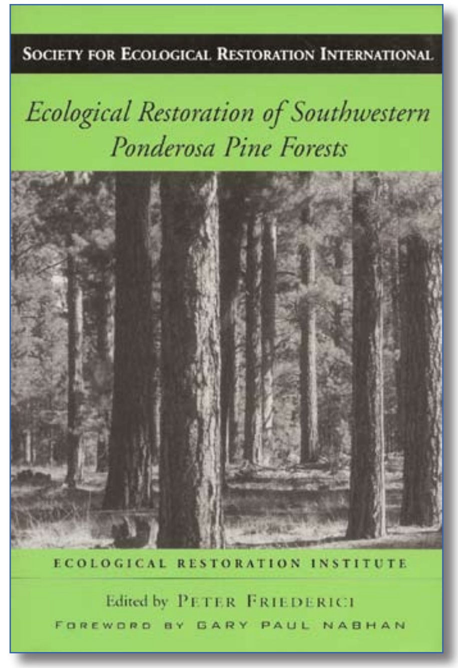
Conceptual Basis for Management and Restoration

The importance of developing a conceptual basis for managing and restoring degraded ecosystems is gaining increasing recognition (Allen et al. 1997; Williams et al. 1997; Whisenant 1999; National Research Council 2002). For degraded riparian ecosystems like those in the Great Basin, such a conceptual basis must consider both the type and characteristics of degradation, and the recovery potential as determined by the underlying physical and biotic processes.

In the central Great Basin many of the streams and, consequently, riparian ecosystems are currently in a nonequilibrium state. Because of the drought that occurred between 2500 and 1300 YBP and the erosion of available hillslope sediments, the streams are sediment limited and exhibit a tendency to incise. Some of the streams have adjusted to the new set of geomorphic conditions, in other words, they have reached their maximum depth of incision under the current sediment and hydrologic regime. Others are still adjusting and will continue to incise because of heterogeneous channel profiles and the lack of hillslope sediments. Due to the resulting changes in stream processes and surface and groundwater relations, riparian ecosystems have crossed thresholds. For our purposes, threshold crossings occur when the system does not return to the original state following disturbance (Ritter et al. 1999). Threshold crossings can be defined based on the limits of natural variability within systems. For streams and riparian ecosystems that have crossed geomorphic and hydrologic thresholds, returning the system to a predisturbance state is an unrealistic goal. Thus, it is necessary to base concepts of sustainability and approaches to management on current, and not historic, stream processes and riparian ecosystem conditions.

As outlined in Chapter 1, the goal of restoration and management activities in the central Great Basin is sustainable stream and riparian ecosystems. Sustainable ecosystems, over the normal cycle of disturbance events, retain characteristic processes including rates and magnitudes of geomorphic activity, hydrologic flux and storage, biogeochemical cycling and storage, and biological activity and production (modified from Chapin et al. 1996 and Christensen et al. 1996). Sustainable stream and riparian ecosystems also exhibit physical, chemical, and biological linkages among their geomorphic, hydrologic, and biotic components (Gregory et al. 1991). Thus, for the purposes of this volume, managing and restoring riparian areas is defined as reestablishing or maintaining sustainable fluvial systems and riparian ecosystems that exhibit both characteristic processes and related biological, chemical, and physical linkages among system components (modified from National Research Council 1992). Inherent in this definition is the idea that sustainable ecosystems provide important ecosystem services. In the central Great Basin, ecosystem services from riparian areas include an adequate supply of high-quality water, habitat for a diverse array of aquatic and terrestrial organisms, forage and browse for native herbivores and livestock, and recreational opportunities.

Ecological Restoration of Southwestern Ponderosa Pine Forests



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Contents

About This Excerpt

With years of drought parching the southwestern region of the United States, a growing human population moving into drastically altered ponderosa pine forests, and severe wildfires occurring regularly in the region, *Ecological Restoration of Southwestern Ponderosa Pine Forests* is a timely and much-needed volume with information that is also relevant outside this region. In this book, Peter Friederici brings together practitioners and thinkers from a wide variety of fields to synthesize what is known about ecological restoration in ponderosa pine forests. In this excerpt, Max Oelschlaeger argues that all participants in these forests must see themselves as part of the "forest story" for restoration to be successful.

From chapter 6, "Ecological Restoration as Thinking Like a Forest," by Max Oelschlaeger

Ecological Restoration in Ecosystem Context

Frank Golley (1993) argues that the "ecosystem concept" militates against the theoretical separation of evolved human systems, or culture (carried by symbolic codes), from evolved biological systems, or nature (carried by genetic codes). That separation, he continues, is not only conceptually untenable, but dangerous to the sustainability of cultural-natural systems.

The muddle that characterizes restoration of southwestern ponderosa forests is precisely a case in point. Why? Because these ecosystems are increasingly and predominantly artifactual, shaped more by human agency than naturally evolved, more the consequence of cultural narratives than of genetic codes. The evolving narrative of ecological restoration recognizes their anthropogenic nature. It also dreams of restoring these forests to health by uncoupling ourselves from them, by taking actions that incrementally nudge the forests onto trajectories of recovery.

This evolving conversation, whatever its uncertainties, is a high-minded effort—ecocentric rather than narrowly anthropocentric. Ecological restoration presupposes that biophysically evolved systems and creatures have intrinsic value. The community of restorationists is beginning to think and act in ways first conceptualized by Aldo Leopold (1949). He argued that governing land management practices were fundamentally flawed, and that new ways of thinking that address the evolved reality of ecosystems and the deleterious outcomes of ecologically myopic human actions were essential to long-term ecosystem integrity, stability, and beauty. In particular, Leopold distinguished views that classify natural systems as resources to be economically exploited from those that categorize them as something more than raw materials for human purposes—even if they can be economically exploited. In his essay "Thinking Like a Mountain," Leopold shows how ranchers think about montane ecosystems as cow factories, loggers think about them as wood factories, and hunters think about them as deer factories. But none thinks like the mountain itself, a naturally evolved system with a temporal dimension and evolved structure that elude the native range of human perceptions and established conceptual schemes. Leopold's intent is to make clear the difficulty humans face in thinking ecologically. In fact, without expanding the temporal scale of perception and escaping conventions that overdetermine their thinking, humans cannot think ecologically.

For those caught in the muddle, the challenge of restoration might be described as coming to "think like a forest." And maybe, just maybe, accepting that challenge might help us begin to understand why stakeholders are conflicted. Why? Because humans have very little experience, almost none, in thinking like a forest. Since the advent of Neolithic culture, we big-brained two-leggeds have primarily thought about the natural world, whether landscapes or species, from the perspective of a self-interested species. In the modern world, that self-interest has played out through narrow, fundamentally economic judgments; forest policies and sciences have primarily served the ends of exploitation.

Public-lands forests would doubtless be in even worse condition without the efforts of Gifford Pinchot, who stopped a long history of laissez-faire exploitation. But the ethical credo behind progressive resource conservation was—and remains—largely economic. Fire exclusion and suppression grew out of the same ideology that led to the damming of wild rivers. Just as free-flowing rivers were conceptualized as wasting water, so forest fires were viewed as destroying valuable timber, and the grasses that naturally carried frequent, low-intensity fires through the pines were seen as fodder that could grow beef and mutton. Today's multi-use statutory framework (often termed "multi-abuse" by environmentalists) is firmly rooted in utilitarianism. Forest science itself has been mesmerized by classical physics and its quest for predictive knowledge, and by market economics, which views forests as raw materials only.

Forest science fails to recognize its roots in the scientific revolution of the 1600s and the notion that through science humans would become the masters and possessors of nature. The resulting "conspiracy of optimism" (Hirt 1994) held that by applying science to management and policy we could exclude fire, harvest timber, eradicate insects, graze the grasses, provide recreational opportunities— and still have healthy, flourishing forests. But forest science was until recently blind to the evolved role of low-intensity ground fires in ponderosa pine ecosystems, ignorant of the unsustainability of commercial forestry in the arid Southwest, indifferent to the adverse consequences of grazing, and oblivious to the harmful consequences of some recreational uses.

Politically the forests have been held in the lock of the "iron triangle" (Chapter 5) made up of commercial interests, western congressional representatives and senators, and entrenched land management bureaucrats, including forest scientists. It assured a stream of timber, employment, and subsidies for the commercial interests, campaign contributions and reelection for members of Congress, and funding for management agencies—but none of these stakeholders were thinking like a forest. Rather, all considered the forest a commodity, raw material to fuel an economic engine.

The situation finally changed during the 1990s, as commercial forestry on public lands dramatically declined. Yet environmental advocates continue to worry, understandably, about the reestablishment of commercial forestry. Today's greatest danger to ponderosa forests is not the sawyer but crown fire, but proposals for multigenerational, landscape-scale ecological restoration projects that would incrementally lessen and ultimately eradicate the danger of crown fire are viewed by some advocacy groups as a Trojan Horse. Given the history of these forests, this position is not incomprehensible.

Yet as I listen to the diverse community of ecological restorationists, I hear hope that the basic composition, structure, and function of ecosystems driven far from their natural dynamic equilibrium might be restored. Who among us—other than profiteers looking for a quick buck and the first stage out of town—could be against such a goal? We have learned that the anthropogenic ecosystems to which we are so closely coupled are, whatever previous intentions, subject to catastrophic, costly, and irreversible changes. I have been somewhat surprised when people whom I respect for their untiring defense of southwestern forests claim that ecological restoration is a fast road to Hell—to forest conditions worse than those that exist now. This proves, at least, that "high-mindedness" comes at the cost of cognitive dissonance. But cognitive dissonance can be productive by inviting a deeper and richer understanding of the tangle of issues concerning restoration.

Sweet Reason

Human beings are generally not risk-takers. We tend to like things settled, sure and certain. Politicians, especially, do not like risk. Land managers do not like risk. They need to be in control of the forest. Scientific researchers attempt to minimize risk of various types of methodological mistakes. Environmentalists do not like risks, either, especially leaders of many environmental organizations, for whom compromise has become a dreaded word. And all of us, when it comes to ethical judgment, hate even a whiff of relativism. What we like are evaluative judgments that a particular goal is clearly better than any other goal. Thus, we too often accept utilitarian arguments.

Yet the deepest thinkers of our time, like the Nobel Prize winner Ilya Prigogine, argue that the end of certainty is at hand (Prigogine 1984, 1997). Fallibilism rules. No judgments of policy, science, or ethics are anything more or less than fallible. Remarkable advances in understanding the biological basis of human nature have trailed in the wake of theoretical developments such as Gödel's proof and Heisenberg's uncertainty principle. As a result, some say we have moved beyond both objectivism (the belief that humans can have sure and certain knowledge that is good for all people in all places at all times) and relativism (the belief that humans can never make knowledge claims that escape the particularity of time and place, person and class, or political ideology, nationalist fervor, and religious dogma).

One constructive alternative to either objectivism or relativism is the conjecture that humans learn as they go. We are, simply, fallible learners. Learning how to think like a forest is an exercise in fallibility that perhaps begins with the recognition that most of what we thought we knew about forests is not defensible. Clearcutting, for example, defended by several generations of timber managers as scientifically grounded and thus as rational forest management, is clearly a failed economic strategy that is hard on western forests and associated human communities (Langston 1995; Power 1996). Similarly, fire exclusion within ponderosa pine forests (though not necessarily at urban–wildland interfaces) is unecological and economically wasteful, leading to ever-increasing loss of ponderosa forests to crown fire, destruction of watersheds, loss of critical habitat for endangered species, dramatically increasing expenditures to fight fires, loss of property, and even the loss of human life (US-GAO 1998).

That said, are we learning to think like a forest? Maybe, only maybe. We remain collectively at the sheer beginning of a learning curve that slopes up steeply and only gradually flattens out. The scientific process of conjecture and refutation—or fallible learning—is farthest along in answering question one: What are the causal factors that have led to present conditions and that might be manipulated to heal ponderosa forests? These factors are addressed in Parts II and III of this volume. However, stakeholder groups, including researchers, are beginning to realize that science, however noble in its quest for truth, is not the final solution (Chapter 5). Neither the science that exists today, nor future advances, will ever be complete. Scientific inquiry itself is increasingly understood as existing within and partially framed by complicated historical, social, political, and economic contexts (Sarewitz 1996). Further, scientific inquiry inevitably has multiple public dimensions, including a commitment to open and continuing public inquiry, comment, and even participation (Lee 1993; Wildlands Project 2001–2002).

Enter question two; namely, policy—What ameliorative actions are appropriate, over what time frame, at what cost?—and the policy process— What processes are appropriate in establishing forest policy? Many commentators remark on the paradox of the public-lands West, which at one and the same time manifests an inspiring "geography of hope" (Stegner 1987) while yielding first and typically a politics of exploitation (Wilkinson 1992) that gives way finally to a politics of stalemate (Kemmis 1990). As so-called new Western historians have made clear (Limerick 1987), the post-settlement West was overdetermined by ideologies little suited to its landscapes, and thereby subjected to short-term exploitation. Only recently has the policy process started to change, with a variety of experiments in inclusive, collaborative processes. Given the diversity of opinion among stakeholder groups, though, and the inability of various experiments in collaborative process to overcome these differences, it is likely that the federal government will continue to pour tens of billions of dollars into the black hole of fire exclusion and suppression (Chapter 4). As far as the policy process that might establish landscape-scale ecological restoration projects is concerned, fallible learning remains a dream.

Finally, question three, that of intention: What reasons legitimate a proposed policy, policy process, or scientific claim? Clearly, given the muddle that exists and the increasing frequency of crown fires, the status quo is not a legitimate alternative. Prolonging the muddle might serve the agendas of some stakeholder groups, but such stalemate blocks thinking like a forest.

Policies for ecological restoration are informed by science on the quantitative side and by ethics on the qualitative side. Science itself is inevitably value-laden, and the legislative framework that established and governs our public lands—including such legislation as the Wilderness Act and Endangered Species Act—is itself the result of so-called citizen choices (Sagoff 1988), or affirmations by the American people of what counts and what does not. Though narrowly focused, largely self-interested economic motives continue to play a large role in the management of our national forests and parks, their establishment did reflect America's collective commitment to intrinsic values of the natural (or seminatural) world.

Ecological restoration necessarily serves these larger purposes (Bateson 1979; Golley 1993). Thus, restoration is conflicted partly because it implicitly contains definitions of our humanity—notions of whom we are, where we are going, and why we should go there, or in effect a moral map (Jordan 2003).

Assembly Rules and Restoration Ecology

Bridging the Gap between Theory and Practice

Edited by Vicky M. Temperton, Richard J. Hobbs, Tim Nuttle, and Stefan Halle

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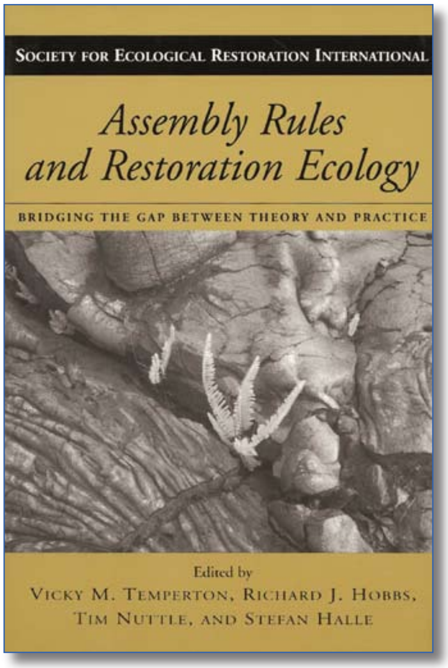


Contents

About This Excerpt

How are ecosystems assembled? How do the species that make up a particular biological community arrive in an area, survive, and interact with other species? How can we return a degraded area to a functioning ecosystem that can serve as a habitat or perform useful ecosystem services? *Assembly Rules and Restoration Ecology* sets restoration ecology in a firm theoretical framework that is both readable and rigorous. In this excerpt from chapter 5, Richard J. Hobbs and David A. Norton consider the interaction of aspects of ecological filters in determining how well restoration projects succeed.

From chapter 5, "Ecological Filters, Thresholds, and Gradients in Resistance to Ecosystem Reassembly," by Richard J. Hobbs and David A. Norton

The way in which biological communities are assembled from the regional pool of biodiversity has received increasing study, and the question of whether recognizable "assembly rules" exist has been examined (see Chapter 3). There is now evidence to suggest that assembly rules may be found for both experimental and natural communities. However, there is also evidence to suggest that different assemblages result from different starting conditions, order of species arrival or introduction, and type and timing of disturbance or management.

Other chapters in this book discuss the ongoing debate over how the biotic elements of ecosystems assemble, particularly following disturbance, and examine whether this process is relevant to ecological restoration. In Hobbs and Norton (1996), we proposed some general principles that might help build a broad framework for restoration ecology. In this chapter, we explore various aspects of ecosystem dynamics and restoration to develop such a framework further. Specifically, we look at how a general framework can serve as a basis for better understanding what can be achieved in restoration at a particular site. We start by reviewing recent developments in ecological concepts and examining how they radically alter the way in which we should consider ecosystem dynamics and restoration. In particular, we reflect on the importance of alternative stable states in ecosystems and the drivers that force transitions between states. We look at these state and transition approaches in a restoration context, particularly in relation to restoration thresholds. We then consider the concept of biotic and abiotic filters and examine whether it meshes with the ideas of states, transitions, and thresholds to assist in developing a comprehensive framework for ecosystem restoration.

New Ecosystem Paradigms

Ecological concepts have changed dramatically over the past 20 years (Pickett and Ostfeld 1995, Pickett et al. 1992). It is now increasingly recognized that ecosystems are complex, dynamic entities that vary at multiple scales in both space and time. The complex dynamic behavior of ecosystems often makes it difficult to understand how they function and to predict the outcomes of any given event or management activity (Hobbs 1998, Hobbs and Morton 1999, Pahl-Wostl 1995). Following Hobbs and Morton (1999), we now outline some of the main changes in ecological thinking that are relevant to understanding ecological assembly and restoration.

The Flux of Nature

Ecologists historically viewed the natural world as a fundamentally stable place in which each species had its ordered position within a community and in which any disturbance would result in an ordered successional progression leading back to the original climax state (Christensen 1988). Ecological communities were considered to be organized, patterned collections of coevolved species that incompatible species could not easily penetrate (Simberloff 1982). Ecologists now speak of this view as the equilibrium paradigm. In recent years, we have seen this notion of organization and stability give way to the view that the natural world is dynamic. In this paradigm, ecosystems are characterized more by instability than by permanence. Ecosystems are also marked by frequent disturbances that continually push them in alternative directions instead of causing them to return to their original condition. We now see ecosystems as characterized by more or less unpredictable individualistic species responses rather than by predictable and correlated species responses.

The *nonequilibrium paradigm* just described recognizes that the natural world is an uncertain place in which disturbances are constantly causing alterations in the composition of assemblages and in the spatial patterns of the environment (Fiedler et al. 1997, Pickett et al. 1992, Pickett and White 1985, Sousa 1984). This view does not suggest that ecological equilibria do not exist but rather that they are scale-dependent and embedded in nonequilibrial conditions. Nevertheless, the nonequilibrium paradigm does imply that predictable endpoints to the successional process following disturbance are rare, that multiple stable states may exist, and that some quasi-stable states can persist for long periods.

Multiple Stable States

Disturbance inevitably sets in motion some form of succession. Current discussion of ecosystem development following disturbance asks whether concepts of individualistic assembly and multiple endpoints are more appropriate than classical successional theory (Young et al. 2001). The course of succession can be difficult to predict, because the direction the ecosystem or assemblage takes is contingent upon the particular circumstances of the disturbance and the nature of the biophysical conditions that precede and follow it. The notion of contingency implies that history matters in patterns and processes of community change. As a consequence, the endpoint of a successional process is not a predictably uniform outcome; instead, several states are possible, depending on the contingent circumstances (Hobbs 1994, Noble and Slatyer 1980). A general feature of many ecosystems is the potential for the system to exist in a number of different states, depending on both past and present biotic and abiotic factors.

Patchiness and Landscape Ecology

Recognition of the importance of spatial and temporal variability, together with the increased availability of suitable tools for analyzing it, is at the root of the emerging discipline of landscape ecology. The understanding and management of the natural world depends as much on the analysis of flows of resources across ecosystems as it does on the detailed study of individual places. One of the principal issues underpinning landscape ecology is recognition of the vital importance of patchiness (Levin 1989, Ostfeld et al. 1997, Turner and Gardner 1991). Patchiness focuses on the spatial matrix of ecological processes and emphasizes the fluxes of materials and organisms within and between different parts of the landscape.

Thresholds in Restoration

The recent changes in the way ecological systems are viewed have a number of lessons for restoration ecology. If ecosystems are nonequilibrium, patchy, and liable to exist in a number of different states, then restoration goals need to take these facts into account. Aiming for a single endpoint may not be valid or may constrain the restoration endeavor too much. Furthermore, it seems likely that for many ecosystems, restoration thresholds exist as a result of human activities that prevent the system from returning to a less degraded state without the input of management, restoration, and aftercare effort (Aronson et al. 1993a; Hobbs and Harris 2001, Whisenant 1999).

Whisenant (1999) has recently suggested that two main types of such thresholds exist, one caused by biotic interactions and alterations and the other by abiotic alterations, transformations, or inherent limitations (Figure 5.1). If the system in question has been degraded mainly as a result of biotic changes (such as grazing-induced changes in vegetation composition), restoration efforts should focus on biotic manipulations that remove the degrading factor (for example, the grazing animal) and adjust the biotic composition (for example, replanting lost species). If, on the other hand, the system has become degraded as a result of changes in abiotic features (such as soil erosion or changed hydrology), restoration efforts must focus first on removing the degrading factor and repairing the physical and/or chemical environment (for example, to reinstate a particular hydrological regime). There is little point in focusing exclusively on biotic manipulation and ignoring the abiotic problems. In other words, system functioning should be corrected or maintained before questions of biotic composition and structure are considered. Taking system function into account provides a useful framework for initial assessment of the state of the system and subsequent selection of repair measures (Ludwig et al. 1997, Tongway and Ludwig 1996). Where function is not impaired, restoration can then focus on composition and structure.

Note that the threshold model (see Figure 5.1) suggests that the biota simply respond passively to the abiotic environment; that is, biotic thresholds always come after abiotic ones. In some situations, however, the presence of a keystone species or "ecosystem engineers" may be required to change the abiotic environment. Hence, biotic manipulation may be required in some cases to overcome an abiotic threshold.

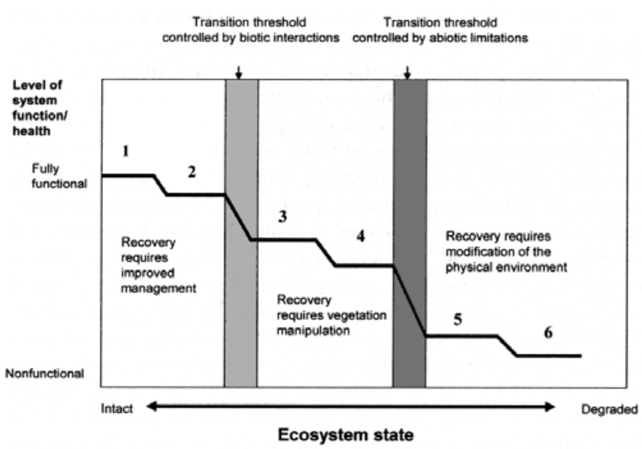
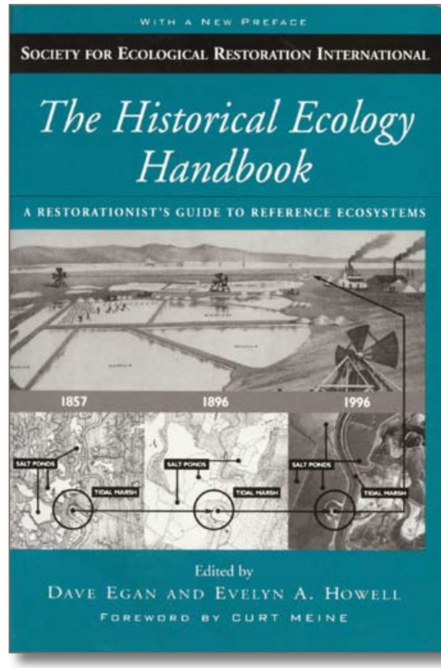


Figure 5.1: Conceptual model of transitions between undegraded and degraded ecosystem states and the presence of biotic and abiotic thresholds (after Whisenant 1999).

The Historical Ecology Handbook

A Restorationist's Guide to Reference Ecosystems



Edited by Dave Egan and Evelyn A. Howell

Foreword by Curt Meine

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Tables, figures, boxes, index

Contents

About This Excerpt

A fundamental aspect of the work of ecological restoration is to rediscover the past and bring it into the present. Dave Egan and Evelyn Howell have assembled twenty-three experts who describe techniques focusing on culturally derived evidence (documents, maps, photographs, land surveys, oral history) and biological records (woodlot surveys, tree rings, pollen, packrat middens, opal phytoliths, animal remains, records of changes in soil and hydrology). This excerpt discusses tree rings as a source of proxy information that can be used in reconstructing and understanding the history of past cultures, landscapes, and environments.

From chapter 8, "Using Dendrochronology to Reconstruct the History of Forest and Woodland Ecosystems," by Kurt F. Kipfmueller and Thomas W. Swetnam

Tree rings are an important source of long-term proxy information that can be used in reconstructing and understanding the history of past cultures, landscapes, and environments. Dendrochronology—or tree-ring dating—relies on the practice of cross-dating, which is the assignment of exact calendar dates to each annual ring through the matching of patterns of growth and other tree-ring characteristics. The best-known applications of dendrochronology (e.g., Stokes and Smiley 1968; Fritts 1976; Baillie 1995; Schweingruber 1996). The tree-ring dating of ancient cliff dwellings and pueblos of the southwestern United States, for example, is widely known and celebrated in both scientific and popular literature (Douglass 1929; Baillie 1995). Likewise, the unique insights derived from tree-ring reconstructions of past climatic variations are commonly referenced in assessments of climatic change and its implications at global scales (Oldfield 1998; USGCRP 1999).

A somewhat less well-known application of dendrochronology is in the field of historical ecology (or paleoecology). Although the potential of tree rings for ecological research was recognized by the founder of modern dendrochronology, A. E. Douglass (1920), it is only recently that the subdiscipline of dendroecology has received much attention by dendrochronologists or ecologists (e.g., Fritts and Swetnam 1989; Schweingruber 1996). In 1986, for example, an international conference on the ecological aspects of tree-ring research resulted in seventy-nine conference papers (Jacoby and Hornbeck 1987), and a recent tree-ring conference with the broad theme of "Environment and Humanity" (Dean, Meko, and Swetnam 1996) contained about thirty papers (out of a total of eighty-two) that directly addressed ecological topics. Moreover, the utility of tree rings in addressing important ecological questions is gaining recognition among the larger community of ecologists. This trend is exemplified by recent W. S. Cooper Awards (for "outstanding contributions in geobotany, physiographic ecology, or plant succession") presented by the Ecological Society of America to the authors of four papers that made extensive use of dendrochronological techniques (Hupp 1992; Fastie 1995; Arsenault and Payette 1997; Lloyd and Graumlich 1997).

A primary reason for this expanded interest in dendrochronology is that both ecologists and natural resource managers have become increasingly aware of the fundamental importance of historical perspectives. This appreciation stems, in part, from recognition of the ubiquity of nonequilibrium dynamics arising from historical processes, such as climatic change and aperiodic disturbances (Sprugel 1991). To understand how ecosystems arrived at their current configurations, we must know about past events and trajectories of change (Brown 1995; Christensen et al. 1996). This interest in history is reflected in an increasing demand for reconstructions of past ecosystem processes and structures that would be useful in defining the "historical or natural range of variability" (Morgan et al. 1994; Kaufmann et al. 1994; Landres, Morgan, and Swanson 1999; Stephenson 1999; Swetnam, Allen, and Betancourt 1999).

Dendrochronology is particularly suited to historical-ecological research because trees tend to be long-lived and the variations in characteristics of their annual rings can be used to reconstruct long and detailed histories of the surrounding environment. Another reason is that cross-dating—the most important principle and practice of dendrochronology—facilitates a multidisciplinary and multiple-lines-of-evidence approach that is very effective in historical reconstruction (Swetnam, Allen, and Betancourt 1999). Cross-dating is the matching of tree-ring characteristics within and among trees across a range of temporal and spatial scales for the purpose of exactly dating individual rings and the structures and elements contained within the rings. Cross-dating is most commonly due to regional climatic variation (e.g., drought and wet years) that cause synchronous changes in tree growth processes (Douglass 1941; Stokes and Smiley 1968; Fritts 1976). Hence, by carrying out tree-ring cross-dating, most dendrochronologists are at least indirectly involved in the study of climatic variability, even though their primary focus may be on the study of ecological events and processes (e.g., births and deaths of trees, wildfires, insect outbreaks, and construction of ancient dwellings and human demography).

The use of dendrochronological principles and techniques affords several important advantages over other dating techniques. The primary advantage of dendrochronology is accuracy of dating. The use of dendrochronological principles ensures that the assignment of annual dates to a series of tree rings is exact. This accuracy in turn facilitates the establishment of connections between the temporal occurrence of events and the timing of other changes to the physical system. Without this level of accuracy, dendrochronologists cannot establish important relationships between tree growth and events in the surrounding environment.

Second, tree-ring data represent an extraordinary natural archive of ecological variation over long periods of time. Depending upon the physical environment and rate of decay, tree rings can be used to reconstruct past events or changes in ecological systems spanning centuries and, in some cases, millennia. This is especially true when a large number of samples are cross-dated, as opposed to simple ring counting of a few samples. Simple ring counting is limited temporally by the longevity of the tree species being sampled. Though this may be very long in some species that attain great age, such as bristlecone pine (*Pinus longaeva*), the cross-dating of remnant material (i.e., sample from logs, stumps, and standing dead trees) can extend the record of change considerably. The bristlecone pine chronology developed by cross-dating currently exceeds nine thousand years, almost twice as long as the oldest currently living individual (the oldest living bristlecone pine that we know of is about forty-eight hundred years old). Remnant material has also been an integral part of development of fire histories in the southwest (e.g., Baisan and Swetnam 1990) and of documentation of changes in forest communities (LaMarche 1973; Lloyd and Graumlich 1997; Donnegan and Rebertus 1999).

Another benefit of dendrochronology is that it can assist in identifying potential mechanisms of variability. For instance, tree-ring-based fire history reconstructions indicate that fires occurred in many mountain ranges of the southwestern United States during 1748 (Swetnam and Baisan 1996; Swetnam and Betancourt 1998) (figure 8.1). Study of historical documents and climate reconstructions from tree rings and corals indicate that the El Nino–Southern Oscillation was probably a climatic mechanism for those events (Swetnam and Betancourt 1990). An unusually strong El Nino event occurred in 1747. The regional climate signal is clearly evident in the growth characteristics of precipitation sensitive conifers collected throughout the region, with a wide ring forming in 1747 followed by a narrow ring during the dry year of 1748 (figure 8.1). Similar relationships between climate and natural disturbances exist in the case of insect-caused epidemics, as we will describe later.

Ex Situ Plant Conservation

Supporting Species Survival in the Wild

Edited by Edward O. Guerrant Jr., Kayri Havens, and Mike Maunder

Foreword by Peter H. Raven

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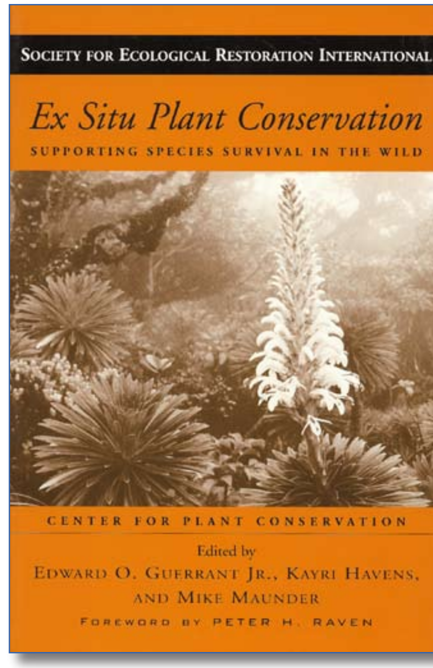


Contents

About This Excerpt

This book outlines the role, value, and limits of ex situ conservation and updates the best management practices for the field. Plant conservation practitioners at botanic gardens, zoos, and other conservation organizations, as well as students and faculty in conservation biology, managers of protected areas, and the international community concerned with species conservation will find *Ex Situ Plant Conservation* a vital tool. The following excerpt from chapter 12 discusses the profound modifications to which ex situ populations can be subjected as a result of the horticultural or storage environment.

From chapter 12, "Population Responses to Novel Environments: Implications for Ex Situ Plant Conservation," by Brian C. Husband and Lesley G. Campbell

The primary goal of ex situ plant conservation is to establish and maintain seed or growing collections of wild species outside their natural habitat for the direct or indirect purposes of species recovery in situ. Such programs typically involve three stages: collection from natural populations, establishment and maintenance of seed or growing material off site and, when appropriate, use of ex situ plant material for in situ reintroduction efforts. Viewed in this way, from initial collection to final end use, the success of an ex situ conservation program depends on its ability to adequately represent the species of interest in the ex situ population and to preserve the utility of the population for future recovery efforts. The challenge for conservationists, then, is to determine the genetic and demographic factors that affect the implementation and long-term utility of an ex situ conservation program.

In this chapter, we identify the critical genetic and demographic factors influencing ex situ conservation by examining relevant evolutionary principles. We begin by considering the historical role of evolutionary biology in plant conservation and the need for its greater use in ex situ conservation. We argue that the challenges facing ex situ conservation programs can be viewed within the conceptual framework of populations in abruptly changing environments. Following on this theme, we characterize the selective environment that a transplanted population will experience and the demographic consequences that such an environmental shift may impose. We then explore the genetic and demographic factors that may influence the success of such a transplantation or colonization event. Finally, we discuss the implications of our findings for ex situ conservation programs and suggest steps that increase the chances for success.

Evolutionary Biology in Ex Situ Conservation

Evolutionary biology has contributed much to the development of conservation, particularly as it relates to the protection of extant populations in situ. For example, the theoretical and empirical literature regarding the ecology and genetics of small populations (Franklin 1980; Lande 1988, 1993; Barrett and Kohn 1991; Rieseberg 1991; Schemske et al. 1994; Lynch et al. 1995; Newman and Pilson 1997; Fischer and Matthies 1998a, 1998b; Bataillon and Kirkpatrick 2000) has helped to quantify the risks of extinction facing many threatened species. In addition, knowledge of the organization of genetic variation (Frankel and Brown 1984; Brown and Briggs 1991) has led to the refinement of collection and management strategies used in national programs of plant conservation (Falk 1991; Center for Plant Conservation 1991). Unfortunately, the evolutionary biology of ex situ conservation for wild species has been less well explored and therefore has played a smaller role in conservation practice. As a result, many ex situ collections have involved a small number of threatened taxa whose representation in collections is narrow (Brown and Briggs 1991; Maunder et al. 1999). The genetic base on which many reintroduction programs are founded is likely to be equally narrow.

To determine the potential role of evolutionary biology for guiding offsite collections and restorative plantings, we conducted a survey of recent ex situ plantings and in situ reintroductions for several North American species at risk, available through online searches (Table 12.1). In total, 50 cases were identified; although the list is far from exhaustive, it shows that 79 percent of all plantings were based on propagules from only a single source population, and 50 percent of these plantings were based on fewer than 10 source individuals (Table 12.1). Clearly, collections of threatened species, for either ex situ or in situ conservation, are necessarily constrained by the limited availability of source material (Brown and Briggs 1991). Indeed, many of the species listed in Table 12.1 are not known from more than a single population. It is especially important for this reason that conservation programs consider the genetic (e.g., genetic variance, population differentiation, inbreeding) and ecological (e.g., number of individuals, habitat characteristics) attributes of organisms to ensure that plantings, both on and off site, are viable in the long term and can meet specified recovery targets. Interestingly however, in our limited survey, 85 percent and 26 percent of these cases did not state explicit genetic or ecological criteria, respectively, to guide their programs. When these criteria were considered, they were most often applied to decisions regarding sampling or collecting the original material but rarely in decisions regarding storage or establishment of plantings off site and subsequent reintroduction efforts.

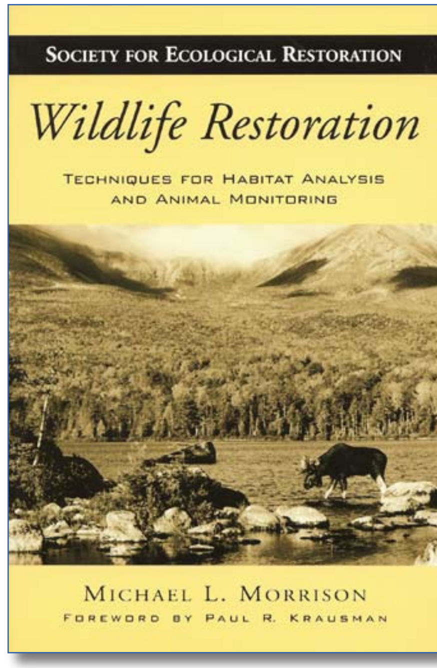
Developing scientifically informed programs of ex situ conservation will involve more than simply applying the theory that is applicable to in situ conservation because their goals and procedures can be quite distinct. In contrast to in situ methods, ex situ conservation involves sampling material from existing populations. Sampling error and loss of genetic diversity through genetic drift are likely to be important in both aspects of conservation and in fact may be exaggerated in ex situ efforts by the joint effects of small source populations and finite samples. Currently, there are several excellent discussions in the literature concerned with the biology and the conservation risks associated with small populations (Soulé and Simberloff 1986; Barrett and Kohn 1991; Lande 1993).

Perhaps what is most distinct about ex situ conservation, in comparison with in situ, is that target plants are being removed from their native location and introduced and maintained in a new environment whose abiotic and biotic conditions certainly are different from that of the original population. Although the specific environment of the ex situ collection will vary widely among cases, the expectation of an abrupt shift in the ex situ environment will apply equally to seed collections and actively growing plant material. Whereas in one case (in situ) the conservationist is attempting to facilitate survival and growth of an extant population, in the other he or she is potentially adding a new source of endangerment, namely maladaptation. Because the biological features of in situ and ex situ populations are inherently different, so too are the best management practices necessary for success. In particular, the conservation biologist is faced with the challenge of collecting a sample that is genetically representative and maintaining it in an environment that may be anything but ecologically representative of the native habitat. Furthermore, ex situ collections are not an end in themselves but rather a means toward the goal of long-term viability for populations in situ. Therefore, ex situ collections often must be managed to simultaneously maintain their short-term viability off site and their long-term utility in restoration or reintroduction efforts (Guerrant 1996).

A conceptual framework and some practical guidelines have been developed for ex situ conservation of crop plants (Brown and Marshall 1995; Schoen and Brown 1995). This application of evolutionary biology may not be completely transferable to wild species because the goals and the target species for the latter are somewhat different. For crop species, ex situ collections are more permanent and serve primarily as sources of single characters or individual genes that plant breeders will transfer into locally adapted stocks. For wild organisms, ex situ programs operate on a short to long-term timescale and often originate from a smaller number of plants. In addition, the primary goal of the ex situ collection is to facilitate demographic viability of the species in the wild, not genetic preservation, so the ex situ collection must provide a source of propagules that can be assembled into whole, functioning populations. Moreover, conserving wild species off site is more complex because of their diverse and complex life histories, variable mating systems, and lower storage tolerance (Brown and Briggs 1991).

Wildlife Restoration

Techniques for Habitat Analysis and Animal Monitoring



Michael L. Morrison

Foreword by Paul R. Krausman

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Tables

Contents

> **About This Excerpt**
>
> In this practical book, the inaugural volume of the *Science and Practice of Ecological Restoration* series, Michael L. Morrison provides ecologists, restorationists, managers, and students with a basic understanding of the fundamentals of wildlife populations and wildlife/habitat relationships. He argues that the success of a restoration project should be judged by how wildlife species respond to it. In this excerpt from chapter 1, Morrison discusses three avenues to recovery of wildlife populations.

From chapter 1, "Populations"

Three Roads to Recovery: Breeding, Reintroduction, and Translocation

Although habitat restoration is the primary focus of this book, in this section I review the use of captive breeding, reintroduction, and translocation in restoring animal populations. Throughout the world, these techniques have been used to assist with the restoration of numerous rare and endangered species. The goal of this section is to introduce readers to these three options for restoration programs.

The process of restoration often creates a system of subdivided populations of animals. Metapopulation biology, therefore, has direct application to the design and management of restoration projects. Metapopulation dynamics includes local population extinction, local population establishment or reestablishment, and movement or linkage among the various local populations. Metapopulation management, if properly applied, can reduce the probability of permanent extinction in local populations and help maintain genetic variability. Captive breeding facilities can maintain essentially fragmented populations. Thus captive propagation and restoration planning depend on a knowledge of metapopulation dynamics (Bowles and Whelan 1994).

Captive breeding and reintroduction are not the ideal means of achieving recovery of rare populations. Captive propagation is expensive, and reintroduction is problematic. Factors such as rates of gene flow among subpopulations, effective population size, mutation rates, and social structure must all be considered when planning restoration and reintroduction. Small populations might have been subject to past population bottlenecks, and genetic manipulation might be required to recover declining populations or to restore or maintain evolutionary potential. Maintenance of genetic variation and evolutionary potential are concerns for rare or isolated populations as well as captive populations. Captive breeding may lead to loss of genetic variation through random drift, to genetic adaptation through selection to the captive environment, and thus to inadequate adaptation for reintroduction to a restored location (Bowles and Whelan 1994).

Ramey et al. (2000) cite five key issues you must address when considering the augmentation of populations:

- Are there two lines of evidence (genetic, demographic, behavior) supporting the hypotheses that a severe population bottleneck has occurred?

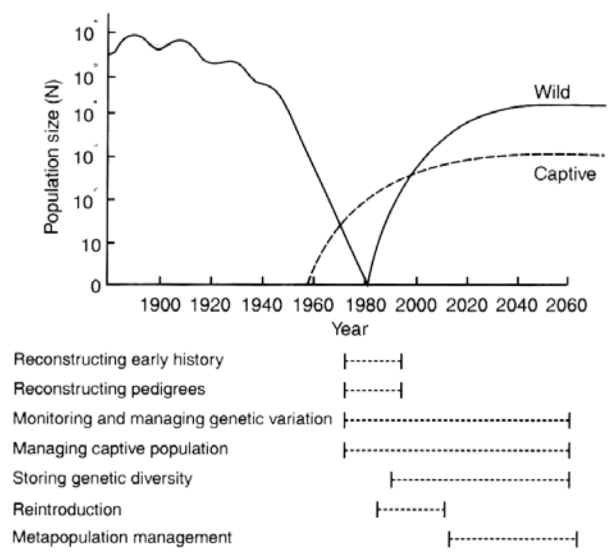

Figure 1.5: Chronology of a captive breeding and restoration program. (From Mace et al. (eds.), "Conserving Genetic Diversity with the Help of Biotechnology—Desert Antelopes as an Example," Figure 1. Pages 123–134 in H. D. M. Moore et al. (eds.), Biotechnology and the Conservation of Genetic Diversity. Copyright 1992, The Zoological Society of London. (Reprinted by permission of Oxford University Press.)

- Would the introduction of additional animals degrade resource conditions, driving the wild animals to more rapid extinction?
- Was the population bottleneck due to a disease outbreak (or other specific occurrence) and can you eliminate the source of the problem?
- Are there habitat patches nearby to establish a population (or metapopulation) of larger size rather than a single, isolated population?
- How should the sex and age composition of an augmentation be structured?

These questions must be considered before you begin an animal restoration project. Moreover, there is no reason to proceed with reintroduction if habitat and niche conditions are not appropriate.

Issues

Captive breeding and reintroduction have been successful and indeed may be the only alternative to extinction in the wild. A restoration and conservation program that includes captive breeding and reintroduction involves numerous overlapping steps (Figure 1.5). Here I review some of the issues you should consider when planning for captive breeding, reintroduction, or translocation. I draw heavily from the summary of genetic considerations discussed by Lacy (1994).

GOALS. The goal of captive breeding programs is to support survival of the species, subspecies, or other defined unit in the wild. According to DeBoer (1992), meeting this goal requires:

- Propagating and managing highly endangered taxa, with prescribed levels of genetic diversity and demographic stability, for defined periods of time, to prevent extinction. Using captive programs as part of conservation strategies that manage captive and wild populations to ensure survival of these taxa in the wild— which means using captive populations to reestablish, reinforce, or re-create wild populations.
- Developing self-sustaining captive populations of rare or endangered taxa for education programs that benefit the survival of conspecifics in the wild (Figure 1.6).

Regardless of the species, there are certain prerequisites to meeting these goals:

- At the level of the individual, sufficient longevity and physical, physiological, and psychological well-being should be assured in the captive situation. This involves species-specific zootechnical, medical, and biological knowledge and research.
- At the level of breeding pairs and groups, sufficient reproduction should be assured to guarantee continuity over generations. This entails species-specific knowledge and research on reproductive biology, ethology, and related topics.
- At the level of population, the preservation of a genetic population structure should be assured that resembles the wild one as closely as possible.

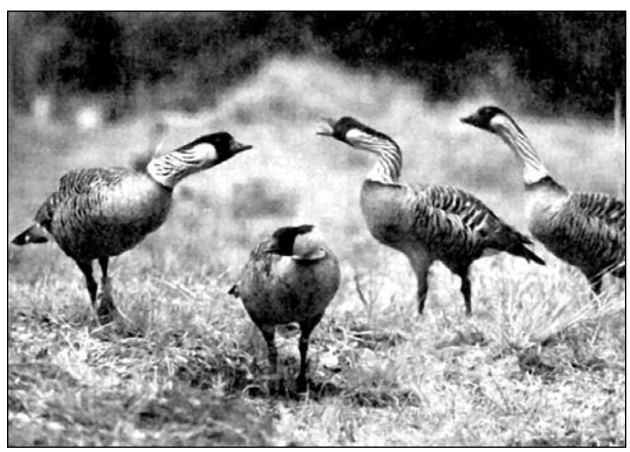

Figure 1.6: The Hawaiian goose, or nene, is being raised at the Zoological Society of San Diego's Keauhou Bird Center, Hawaii, for reintroduction into the wild. (Photo courtesy of Zoological Society of San Diego.)