RESTORING THE HEALTH AND WEALTH OF ECOSYSTEMS

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Abstract

Before we can set about restoring the health and wealth of ecosystems, we need to have a clear idea of what we actually mean by the health and wealth of ecosystems. What is a healthy ecosystem? What is the wealth of an ecosystem? How well do we understand ecosystem structure and function, and can we predict the outcome of any particular manipulation of an ecosystem or its components?

Further, how do we then go about restoring ecosystem health and wealth? What do we want to achieve, and how do we go about it? Are the results from one ecosystem or one location applicable to other ecosystems and other locations?

These questions are explored in the context of recent developments in the science of restoration ecology, and relate these to practical examples from Western Australia and elsewhere. Restoration ecology has frequently been billed as a testing ground for ecological theories, by providing practical case studies within which theory has to be applied. However, until recently, restoration ecology itself has rested on a relatively poorly developed conceptual framework, and has interacted little with other branches of science.

Recent attempts to develop a conceptual framework for restoration ecology have focussed on the changing paradigms in ecology. The concepts of resilience, stability and alternative stable states are central to defining goals for restoration projects, as are concepts relating to the naturalness of ecosystems. In addition, the development of ideas relating to community assembly rules have important implications for restoration ecology. Current debate in ecology over the ecosystem function of biodiversity also has relevance to restoration ecology, as do developments in landscape ecology and design. The need for restoration at broad scales is also apparent in many parts of the world - in other words, we need to move from looking at restoration of particular sites to restoration across whole landscapes. Synthesis of these disparate trends can provide a broadly-based framework on which restoration ecology can develop from a largely site- and situation-specific discipline to one which provides useful and generalizable treatments for the widespread ecosystem degradation apparent today.

There is a clear need for closer linkages between restoration ecology and other fields so that synergies and reciprocal testing of ideas can be encouraged. Similarly, synergistic relationships between researchers and managers are essential for the further development of restoration ecology and its successful application in real world problems.

ECOSYSTEM HEALTH AND WEALTH

This proceedings takes the concepts of ecosystem health and wealth as its theme, extending the usage of the terms health and wealth from humans to ecosystems. While these terms may provide useful metaphors for the discussion of the state of ecosystems, we need to consider exactly what we mean by them. In humans, health relates to the physical and mental well-being of the individual, while wealth usually relates to material well-being in terms of accumulated assets and savings. However, these definitions are not universal and wealth can also relate not only to monetary wealth but also to wealth in terms of family, companionship, personal fulfillment and the like.

There are a number of widely used indicators for both health and wealth. Indicators of health include the kinds of things that are measured on a regular check up with the doctor – i.e. blood pressure, cholesterol levels and so on, but can also include more complex tests and diagnoses which aim to detect more subtle or complex medical conditions. Health can be compromised in many different ways, including by the effects of disease organisms, damage to tissues and bones through mechanical and other impacts, and more subtle or long term effects of diet, smoking and so on. Indices of wealth include measures of net assets, annual income and so on. In both cases the indices are selected to be easily-measured parameters which provide a summary of the current state of the individual. Additionally, further indices are used to summarize the situation for whole communities or countries. For instance, the GDP is supposed to provide a summary statistic indicative of the economic well-being of a country.

How, then, does this relate to ecosystems. Are there readily-obtainable measures of ecosystem health and wealth? Ecosystem health has recently been put forward as an effective means of discussing the state of ecosystems (Costanza et al. 1992; Cairns et al. 1993; Shrader-Frechette 1994). Central elements of ecosystem health are the system's vigour (or activity, production), organization (or the diversity and number of interactions between system components) and resilience (the system's capacity to maintain structure and function in the presence of stress (Rapport et al. 1998a). Attempts have also been made to produce readily-measurable indices of ecosystem health for a number of different ecosystems (e.g. (Shaeffer & Cox 1992; Mageau et al. 1995; Shear 1996; Yazvenko & Rapport 1996). There has, however, been considerable debate over the utility of the ecosystem health concept, with several authors suggesting that it is difficult to quantify and an inappropriate descriptor for use in management decisions (e.g. (Calow 1992; Suter 1993; Wicklum & Davies 1995). Despite this, the concept has gained momentum and provides an easily understandable metaphor for discussing the state of ecosystems, on which attempts to develop interdisciplinary responses to environmental problems can be based (Rapport et al. 1998a,b).

The wealth of ecosystems is generally viewed as the diversity and persistence of living things within the system. This diversity is generally measured at the species level, but can also be considered in terms of functional groups, guilds or other groupings, and is also taken to include genetic, ecosystem and landscape diversity (Noss 1990). The question of how best to measure "biodiversity" surfaces repeatedly, since it is virtually impossible to quantify the total biodiversity of any given system or location, given that this includes a multitude of invertebrates and micro-organisms, many of which occur in the soil and most of which are either difficult to identify or as yet un-named. This has given rise to the search for effective indicators of biodiversity (e.g. Saunders et al. 1998). A further issue is that ecosystems differ naturally in their diversity, and species richness *per se* is not necessarily of prime importance. This has led some commentators to suggest that biological "integrity" is the more important issue – i.e., the presence of all the appropriate elements of the system and the occurrence of all processes at appropriate rates (Angermeier & Karr 1994). Clearly, the concepts of integrity and health are closely linked, and I discuss their relationship further below.

NEW PARADIGMS IN ECOLOGY

Discussions of ecosystem health and integrity, and how these relate to ecosystem restoration, are best undertaken in relation to recent changes that have taken place in the way ecologists view natural systems. Ecology has undergone something of a quiet revolution during the past 20 years (Pickett et al. 1992; Pickett & Ostfield 1995; Hobbs & Morton 1999). Concepts that were considered firmly established in previous decades have undergone considerable revision, even reversal. Here I briefly outline some of the more important of these changes, based on material presented by Hobbs and Morton (1999).

The flux of nature

Previous generations of ecologists operated largely on the assumption that the natural world was fundamentally a stable place, a collection of communities in which each species had its ordered position and in which any disturbance would result in an ordered successional progression leading through subclimax phases back to the original climax (Christensen 1988). Ecological communities were considered to be organized, patterned collections of co-evolved species, into which incompatible species could not penetrate (Simberloff 1982). Ecologists now speak of this era as the period of the *equilibrium paradigm*. In recent years we have seen this notion of organization and stability give way to a vision of flux. Most ecologists have come to the view that the natural world is characterized more by instability than permanence, by frequent disturbance that continually pushes ecosystems in alternative directions instead of causing them to return inevitably and regularly to their original condition; more by unique specific responses than co-ordinated, predictable, tightly constrained combinations of species, as individualistic responses outweigh tendencies towards regularly occurring communities. Awareness that the natural world is an uncertain place in which disturbances are constantly causing alterations in composition of assemblages and in spatial pattern of the environment has led now to the *nonequilibrium paradigm* (Pickett et al. 1992; Fiedler et al. 1997). This paradigm does not hold that ecological equilibria are non-existent, but rather that they are scale-dependent and embedded in nonequilibrial conditions. Nevertheless, the nonequilibrium paradigm does imply that predictable end-points 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 train some form of succession. It is apparent now that the course of the succession is difficult to predict, because the direction which 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 brings history to the fore: history very much matters in patterns and processes of community change. As a consequence, the end-point of many successional processes is not a predictably uniform outcome; instead, several states are possible, depending on the contingent circumstances (Noble & Slatyer 1980; Hobbs 1994). Depending on the frequency of the disturbances, these multiple states may be stable for long periods of time, and distinct thresholds may exist which limit the transition from one state to another. The differences among outcomes of successional events in seemingly similar assemblages or ecosystems may well follow broadly interpretable patterns, but the itineraries are not easily predictable at the outset of the journey. This has important implications for restoration ecology, since it implies that the outcome of particular restoration is not entirely predictable and that there may be barriers or thresholds to restoration which require management input to overcome (Hobbs & Norton 1996; Yates & Hobbs 1997).

Patchiness and landscape ecology

Recognition of the importance of spatial and temporal variability, together with the increased availability of suitable tools for analysing it, has galvanized landscape ecology. Its re-emergence springs from realization that understanding and management of the natural world depends as much on the analysis of flows of resources across ecosystems as it does on the study of quadrats. But perhaps the principal issue underpinning landscape ecology is recognition of the vital importance of patchiness (Turner & Gardner 1991). Patchiness does not yet possess a complete or unified theory, but is a rapidly developing conceptual tool (Levin 1989; Ostfeld et al. 1997). Patchiness focuses on the spatial matrix of ecological processes, and emphasizes the fluxes of materials and organisms within and between parts of the landscape. It is a form of spatial heterogeneity in which boundaries are discernible, and in which units appear as contrasting, discrete states of physical or ecological phenomena Ostfeld *et al.* 1997). An array of patches constitutes a mosaic at whatever scale is appropriate for investigation (although it is important to note that multiple scales may be important). The study of patch dynamics promises to provide a valuable framework in which to understand and manage the landscape mosaic and to conduct landscape-scale restoration, although there is still much work to be done in this area. Although the need for landscape-scale restoration is clear, we are only beginning to develop methods for setting landscape-scale goals and priorities for action (McIntyre & Hobbs 1999).

Prediction

The nonequilibrium paradigm sees ecosystems as probabilistic rather than deterministic; inherently, therefore, most ecologists believe that ecosystems are characterized by uncertainty rather than by predictability. Because of the overwhelming importance of this uncertainty, ecologists have invested considerable intellectual energy in trying to comprehend *environmental stochasticity* - correlated variability in chance events caused by patches in a landscape experiencing a similar environment, including both physical and biotic features - and *catastrophes* - correlated variability of large magnitudes that occur at a low frequency. We cannot avoid the lack of predictability; consequently, there is a need to identify the bounds or conditions under which decisions can be made in the face of uncertainty. Risk analysis, and adaptive management through more detailed involvement of managers in research and development, are the principal routes by which ecologists are struggling to work with unpredictability. Although admission of the extreme difficulty of prediction has initially caused ecologists to be concerned that their science is fuzzy, a focus on uncertainty and risk analysis is common to many people in the social, political and economic spheres (Graham & Wiener 1995), and quantitative risk assessment is widely used in engineering and technology. Hence, ecology is not necessarily difficult, in this sense, for decision-makers to comprehend. This emphasises the point made above, that we cannot be too prescriptive about the outcome of any given restoration project.

Human beings and ecology

Recognition of the inevitability of disturbance, and of its profound ecological consequences, leads inevitably to the inclusion of humans as primary agents of flux in ecosystems (McDonnell & Pickett 1993; Hobbs 1997; Vitousek et al. 1997). Ecology is now beginning openly to extend its interest from supposedly "natural" systems, in order to include human-dominated systems. Anthropogenic disturbance can now be incorporated into ecology in the same way as any natural disturbance, rather than being considered as distracting noise. An implication of this for restoration is that it may no longer be appropriate to consider some idealised pristine ecosystem as a valid goal for restoration. This is discussed further below.

Biodiversity and ecosystem function

How does the wealth of an ecosystem (i.e. its biotic diversity or integrity) affect the health of the ecosystem (i.e. its functioning)? After a period of neglect, the question of how biotic diversity and ecosystem function are related is now considered one of the fundamental questions in ecology. The early neglect of this question can be traced to the fragmentation of ecology into distinct branches, most notably with a split between organism-centred population and community approaches and the material flux approach of ecosystem ecology (Jones & Lawton 1995)("things" versus "stuff": (Pickett et al. 1994). A large international program organized by SCOPE recently examined the question in detail, both from a theoretical point of view and in terms of what we know from examples from a variety of ecosystem types (Schulze & Mooney 1993; Mooney et al. 1996). The societal relevance of the question has also recently been explored in the context of "ecosystem services" and "how the diversity of life sustains us" (Baskin 1997; Daily 1997).

Ecologists are thus currently in the middle of a flurry of activity surrounding the question of the role of biodiversity in ecosystem function, but are also apparently in a bit of a muddle at the same time. Part of the problem has been a failure to define exactly what question is being asked. "The ecosystem function of biodiversity" is a ridiculously broad term, and both "ecosystem function" and "biodiversity" can be interpreted in numerous ways. Ecosystem function can refer to the primary functions of water, carbon, energy and nutrient cycling, or it can refer to the myriad of processes which go to make up these cycles, including biotic interactions. It can also be interpreted in more utilitarian way to mean "ecosystem services" for particular human purposes, such as the supply of fresh water, disease prevention etc. Similarly, biodiversity incorporates all levels of biological organization from genes to landscapes, although it is frequently interpreted simply as "number of species". Species can also be grouped in a number of different ways, and attempts are being made to define sensible groupings which have functional significance(Smith & Shugart 1996; Woodward & Cramer 1996). Without defining exactly what aspect of ecosystem function one is trying to relate to what element of biodiversity, it is unlikely that useful questions can be asked.

A further problem has been the lack of consideration of the impact of different *kinds* of species. Experimental work has concentrated almost exclusively on the number of species rather than the mix of different types of species. For instance, (Tilman & Downing 1994; Tilman et al. 1996) constructed grassland plant communities by randomly drawing species from a total species pool. Other ecological work on community assembly rules suggests that there may be readily-defined reasons why certain plant assemblages develop in response to particular environmental and biotic factors which act to "filter" species from the regional species pool (Keddy 1992; Weiher & Keddy 1995). It is also clear that individual species vary greatly in terms of their functional importance (e.g., in their quantitative contribution to particular processes), and a variety of terms have been derived for species which strongly influence system structure or function: e.g. "keystones" (Mills et al. 1993; Paine 1995; Stone 1995), "drivers" (Walker 1992), and "ecosystem engineers" (Jones et al. 1994). Huston (1997)questions the assumption that species diversity can be divorced from the effects of species identity. Indeed, the debate needs to focus more on the importance of particular elements of biodiversity rather than the importance of biodiversity per se.

An allied question is the degree of functional redundancy inherent in natural communities (Walker 1992). In practice, however, the perception of redundancy depends on the timescales and functions considered. Apparently functionally-similar species are likely to respond to environmental variation or disturbance differently and hence may increase the resilience of the system (Main 1992; Walker 1995; Hobbs & Mooney 1996). Functional redundancy thus provides “fail safe” or “back up” capacity. Recent accounts of the relationship between biodiversity and ecosystem function take greater cognizance of these dual questions of the functional significance of particular biotic elements and the importance of functional redundancy in conferring system resilience (Chapin et al. 1997). Future research on these issues will not only provide a better understanding of how systems work, but will also allow assessment of which system components are functionally the most important to system integrity or persistence, and hence which components are essential both to retain in existing ecosystems and to introduce into reconstructed systems.

GOALS FOR ECOLOGICAL RESTORATION

The term "ecological restoration" covers a wide range of activities involved with the repair of damaged or degraded ecosystems (Jordan et al. 1987; Berger 1990; Baldwin et al. 1994; Harris et al. 1996). An array of terms has been used to describe these activities including restoration, rehabilitation, reclamation, reconstruction, and reallocation. Generally, restoration is used to describe the complete reassembly of a degraded system to its undegraded state, while rehabilitation describes efforts to develop some sort of functional protective or productive system on a degraded site. In addition, some authors also use the term "reallocation" to describe the transfer of a site from one land-use to a more productive or otherwise beneficial use. Majer (1989), Aronson et al. (1993a), Jackson et al. (1995) and others discuss terminology and ideas on what comprises restoration ecology. Unfortunately, a stable terminology has been slow to develop and the above terms are frequently used interchangeably and differently by different authors. Here I will follow Hobbs &Norton (1996) and use the term restoration to refer broadly to activities which aim to repair damaged systems, although the other terms are used as above in particular examples.

Restoration ecology involves a number of interconnected activities, discussed in more detail below. Here I concentrate on the development of goals for restoration, since setting adequate and realistic goals is an essential component of effective restoration activities. Hobbs & Norton (1996) suggest that ecological restoration is usually carried out for one of the following reasons:

1. To restore highly disturbed, but localized sites, such as mine sites. Restoration often entails amelioration of the physical and chemical characteristics of the substrate and ensuring the return of vegetation cover (Collins et al. 1985; Bradshaw 1987; Ward et al. 1990).

2. To improve productive capability in degraded production lands. Degradation of productive land is increasing worldwide, leading to reduced agricultural, range, and forest production. Restoration in these cases aims to return the system to a sustainable level of productivity, e.g., by reversing or ameliorating soil erosion or salinization problems in agricultural or range lands (= rehabilitation, Aronson et al., 1993a).

3. To enhance nature conservation values in protected landscapes. Conservation lands worldwide are being reduced in value by various forms of human-induced disturbance, including the effects of introduced stock, invasive species (plant, animal, and pathogen), pollution, and fragmentation. In these cases, restoration aims to reverse the impacts of these degrading forces, e.g., by removing an introduced herbivore from a protected landscape. In many areas, there is also a recognized need to increase the areas of particular ecosystem types - for instance, attempts are being made to increase the area of native woodlands in the United Kingdom, in order to reverse past trends of decline and to increase the conservation value of the landscape (Ferris-Kaan 1995).

4. To restore ecological processes over broad landscape-scale or regional areas. In addition to the need for restoration efforts within conservation lands, there is also a need to ensure that human activities in the broader landscape do not adversely affect ecosystem processes. There is an increasing recognition that protected areas alone will not conserve biodiversity in the long term, and that production and protection lands are linked by landscape-scale processes and flows (e.g., hydrology, movement of biota). Methods of integrating conservation and productive use are thus required, as for instance in the Biosphere reserve and core-buffer-matrix models (Hobbs 1993; Noss & Cooperrider 1994; Morton et al. 1995). Restoration in this case entails (1) returning conservation value to portions of the productive landscape, preferably through an integration of production and conservation values and/or (2) ensuring that land uses within a region do not have adverse impacts on the region's ecological

processes.

Ecological restoration thus occurs along a continuum from the rebuilding of totally devastated sites, to the limited management of relatively unmodified sites (Hobbs & Hopkins 1990). The specific goals of restoration and the techniques used will obviously differ between these different cases. In general terms, however, restoration aims to return the degraded system to some form of cover which is protective, productive, aesthetically pleasing, or valuable in a conservation sense (Hobbs & Norton 1996). A further tacit aim is to develop a system which is sustainable in the long term. It should, however, be recognized, that goals for restoration are likely to change as societal values and attitudes change. For instance, in many parts of the world, forest ecosystems are now being valued more for non-production values such as those relating to biodiversity and recreation (Williams 1989; Aplet et al. 1993; Dargavel 1995), and hence these factors increasingly have to be built into management and restoration strategies.

Within these broad general aims, more specific goals are required to guide the restoration process. Ecosystem characteristics which may be considered when considering restoration goals include (from Hobbs and Norton, 1996):

1. Composition: species present and their relative abundances
2. Structure: vertical arrangement of vegetation and soil components (living and dead)
3. Pattern: horizontal arrangement of system components
4. Heterogeneity: a complex variable made up of components 1-3.
5. Function: performance of basic ecological processes (energy, water, nutrient transfers)
6. Species interactions: includes pollination, seed dispersal etc.
6. Dynamics and resilience: succession and state-transition processes, recovery from disturbance

This set of characteristics is complex, and often individual components are considered as primary goals. For instance, restoration of a mine site may aim to replace the complement of plant species present prior to disturbance, while other situations may have the restoration of particular ecosystem functions as a primary aim (e.g.. bioremediation of eutrophication in lakes, or the manipulation of vegetation cover to modify water use).

Unfortunately, restoration goals are often poorly defined, or stated in general terms relating to the return of the system to some pre-existing condition. The definition of the characteristics of this condition has proved problematic, since it assumes a static situation. As noted above, ecologists increasingly consider that natural systems are dynamic , that they may exhibit alternative (meta-)stable states , and that the definition of what is the "natural" ecosystem in any given area may be difficult (Sprugel 1991). Indeed, the concept of "naturalness" has been the subject of much recent debate (Elliot 1982; Maser 1990; Gunn 1991; Cowell 1993; Elliot 1994)

The question of relating restoration efforts to particular reference ecosystems has also been debated (Pickett & Parker 1994; Aronson et al. 1995). For instance, which time period do you use for defining the baseline ecosystem? A common trend in non-

European countries is to use the conditions prior to European colonization as a baseline (e.g., Anderson 1991), despite the fact that the characteristics of these pre-European ecosystems are often poorly documented and that irreversible ecosystem changes may have occurred in the meantime. On the other hand, the problem can also arise where the existing ecosystem is taken to be "natural" and used as a baseline, but is, in fact, an artifact of current management practices. An example of the dilemma confronting those wishing to establish baseline or reference systems is presented by the changes in New Zealand ecosystems over the past 1300 yr. At each stage, the ecosystems probably existed in a more or less metastable state. Should the baseline be pre-Maori, pre-European, or present-day?

Much of the ensuing confusion could be avoided by the careful enunciation of specific restoration goals - for instance, restoration of productive capacity can be assessed relative to the productive capacity of similar, undegraded land. Restoration of compositional, structural, and other ecosystem properties can be related to the known range of those properties, either in recent history if temporal data are available, or within similar less degraded ecosystems in the area. A prerequisite for this approach is the development of a set of easily measurable indicators or ecosystem response variables which can be monitored as the restoration proceeds. (Aronson et al. 1993a,b) have developed a set of "vital ecosystem attributes" which they suggest can be used to assess the status and trajectory of a system, and hence could be used to set restoration goals more rigorously. In addition, Hobbs and Norton (1996) suggest using these attributes within the framework of ecosystem health, as discussed above. Assessment of system trajectories toward the recognized range of conditions can then be used as a measure of the success in achieving restoration goals. This provides a feedback loop which checks progress against the established goals.

CONCLUSION

The changing face of ecology discussed above is important when restoration goals are being considered. So, too, is an increasing recognition that social and economic considerations are as, if not more, important to the success of restoration projects than are biophysical parameters. Without socially acceptable goals, based on a clear vision of what we would like our rural and urban landscapes to be like, effective restoration is impossible. Elsewhere, I have explored the need for three levels of activity in relation to restoration, namely the development of a vision (where do we want to go?), a strategy (how do we get there?) and appropriate tactics (which tools do we need to get there?) (Hobbs & Saunders 1999). The restoration and management goals we set need to reflect broader community aspirations, with ongoing dialogue as to what these aspirations could or should be.

Slocombe (1998) recently identified a set of desirable characteristics for ecosystem management goals, which are worth repeating here in the context of ecosystem restoration. He suggested that goals should:

1. Imply and reflect specific values and limits (normative)
2. Reflect “higher” values and ethical principles and rules (principled)
3. Reflect the wide range of interests, goals and objectives that exist (integrative)
4. Work with, not artificially reduce, complexity (complex)
5. Accept and recognize the inevitability of change (dynamic)
6. Synthesize a wide range of information and knowledge (transdisciplinary)

7. Be applicable to a wide range of ecosystem types and conditions (applicable)
8. Involve actors, stakeholders and public (participatory)
9. Be explainable and implementable in a consistent way to different people and groups (understandable)
10. Be inherently tentative and evolving as conditions and knowledge change (adaptive)

If we can increasingly agree that this set of characteristics is desirable, we may be increasingly able to agree on what to restore where and on the best way of going about it. In that way, valuable time and resources can go to the real job of restoring the health and wealth of our ecosystems in the most effective way possible.

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