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New linkages for protected areas: Making them worth conserving and restoring

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Summary

In an increasingly human-dominated world, more realistic, holistic and durable linkages need to be created for protected areas (PAs), for the sake of long-term conservation of species and habitats, and of ecosystem goods and services for serving the local people, as well as societies at large. In conjunction, geographical, physical and biological concepts of linkages need to be extended to embrace socio-economic factors. This paper develops a conceptual approach to this task, employing the nascent notions of 'emerging ecosystems', restoring natural capital and 'socio-ecological systems'. We employ the value notion generally used by economic science, i.e. monetary value, as a conceptual tool in the unavoidable assessment process that modern societies must undertake to decide about conserving or restoring PAs, and ecosystems in general. We also draw attention to the importance of ecological and socio-economic thresholds of irreversibility in the evaluation and decision-making process concerning PAs and their linkages with other PAs, and especially with the unprotected world beyond their boundaries.

Introduction

Recent studies from natural scientists underscore the need for integrating many factors in the selection, planning, financing and managing of parks and other protected areas (PAs) (Driver, Cowling, & Maze, 2003; Faith & Walker, 1996a, b).

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There is also much evidence that PAs should be intimately linked with adjacent landscape units and other PAs, to increase the survival chances of the plant and animal populations, and species of concern (Bennett, 2003). Under PAs, we include nature parks and all other areas designated for protection and delivery of ecosystem services, e.g. watershed and topsoil conservation or hydrological flow management, as well as cultural landscapes that may be under some form of protection (Aronson, Clewell, Blignaut, & Milton, 2006; Council of Europe, 2000; Farina, 2001; Moreira, Queiroz, & Aronson, 2006).

Yet, conservation planning to date has concentrated too exclusively on the representation of biodiversity patterns within a PA or a system of PAs (Cowling, Pressey, Lombard, Desmet, & Ellis, 1999). More attention is needed on the relationships of PAs with their surroundings, both human and non-human. Suggestions have been made for methods to identify high priority sites in terms of uniqueness, and of species vulnerability to threatening processes, mostly of human origin (Araùjo, Williams, & Fuller, 2002; Driver et al., 2003). Far less attention has been devoted to determining how PAs actually benefit, and/or impose costs on local people (Cunningham, 2001; Emerton, 1999). Bawa, Seidler, and Raven (2006), among others, call for abandoning overly general conservation models and paying much more attention to the home-grown models and modalities adopted by locally driven indigenous efforts, which incorporate both ecological and economic systems.

Historically, PAs served as 'set-asides' for biological conservation, separating selected elements of biodiversity (e.g. rare and endemic species) or striking and exceptional landscapes, from anthropogenic processes that threatened their existence in the wild (Terborgh, 1999). Today, however, we see that—especially in developing countries—the dynamics created by exponential growth in human population and migration, as well as by economic expansion and globalisation, have rendered this approach to PAs obsolete. Housing and commercial development, as well as growing and changing consumption patterns and energy demands, in rich and poor countries alike, increasingly compete with natural reserves for set-aside or protected land (Dobson, Rodriguez, Roberts, & Wilcove, 1997; Pierce, Cowling, Sandwith, & MacKinnon, 2002; Wackernagel, Schulz, & Deumling, 2002).

The obvious implication is that we—society—must take responsibility and make conscious coherent decisions about all life-support systems (Janzen, 1998, 1999). Prudence, and a human ecology perspective would seem to be good general guidelines (Gadgil, 1995; Hardin, 1985). With regard to PAs, and restoration sites, conservation planners, especially in developing countries, must take social, economic, cultural and political concerns into consideration (IUCN, UNEP and WWF, 1991; http://www.ser.org/content/Globalrationale.asp). Concurrently, politicians and decision-makers should base their actions not only on social references and pressures, but also on knowledge-based systems, including quantitative, ecological data and theory, even though this reduces their choices in unwelcome ways (Hollick, 1981; Pierce et al., 2002).

One conceptual aid that is relevant are *socio-ecological systems* (*SES*), a concept that has been proposed not necessarily to replace the notion of 'natural' ecosystems, but to complement it (Carpenter, Walker, Anderies, & Abel, 2001). This concept together with the nascent ones of 'emerging ecosystems (EEs)' (Hobbs et al., 2006; Milton, 2003) and restoring natural capital (Aronson et al., 2006) are used here to address the issue of building appropriate and broader linkages for PAs for improving current efforts to conserve and restore them. However, as yet, there are very few practical models of how to apply this concept to management and conservation practices.

Historical and geographical antecedents

Scientific prescriptions for PAs to fulfil their conservation role generally focus on theoretical and technical aspects of how they can meet the objectives of representativeness and persistence of species, ecosystems or landscapes (Williams et al., 2003). Experts in conservation planning have also used the equilibrium theory of island biogeography (MacArthur & Wilson, 1967) and associated biogeographical theory to help set targets for the size, shape and distances among PAs (Margules & Pressey, 2000). This body of theory suggests that bigger, closer and roughly circular PAs are to be preferred, and that PAs should ideally be linked by inter-connecting habitat "corridors" (Diamond, 1975; Wilson & Willis, 1975). Additionally, there is strong experimental support for the prediction that increased isolation, i.e. landscape and habitat fragmentation, reduces the persistence likelihood of many species (Davies, Margules, & Lawrence, 2000), which in turn further supports the need for biophysical linkages and networking among PAs. This approach fails to consider what is happening and what is done *outside* PAs.

As Harmon (1987) has shown, these ideas of configuration and linkages among PAs were developed mainly in conjunction with the national parks concept born in the USA, within a "boundless wealth" paradigm, and the Panglossian "expectation that the natural resources left outside [the PAs] were inexhaustible." The American national park ideal was for a century or more the model system of conservation for a large part of the world (West & Brechin, 1991). However, this model can be entirely inappropriate to deal with conservation problems in developing countries, where large, impoverished rural populations often live in close relation with PAs, and depend largely on them for their subsistence. In fact, natural resources outside the PAs in these countries are often all but exhausted.

Indeed, in most developing countries, large numbers of people live close to, or even within, PAs (Cunningham, 2001; Gadgil & Guha, 1995). In Latin America, about 50% of national parks have human occupation (Amend & Amend, 1992), which also applies in India, Madagascar, and Uganda. Therefore, it is crucial that people in or near parks perceive a connection between their own well-being and the PAs. Otherwise, they will consider these areas of little value and not care about degradation poaching, neglect and overexploitation. This is a complex issue—as people have often been excluded by park demarcation, while continuing to suffer crop and livestock damage caused by "protected" park animals (Cunningham, 2001; Emerton, 1999). Not surprisingly, the costs incurred and benefits (natural resources and tourism revenues) are greatest for people in or very near the boundaries of PAs (Fig. 1).

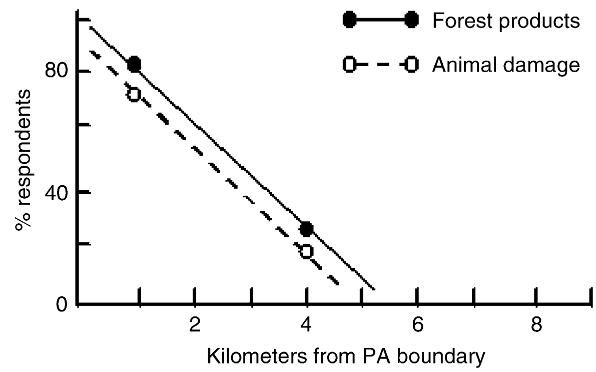

Figure 1. Examples of costs and benefits of PAs to neighbours, as a function of distance in a developing country. Mean percentage of people from communities adjacent to Bwindi-Impenetrable National Park, Uganda ($n = 978$), who were involved in collecting forest products or affected by crop-raiding animals prior to park closure, compared to those further away ($n = 1405$). Redrawn from Cunningham (2001); reproduced with permission of the publisher.

The European concept of national parks, among others, differs radically from the American model. In Great Britain, France, Italy and elsewhere in Europe, people and their activities have, over millennia, shaped and sculpted landscapes with a distinctly human touch. Wilderness as such does not exist there. As a result, European parks were not conceived as wilderness preserves like those of the United States, but rather explicitly cultural landscapes, where people and the land have become inseparable and considered worthy of protection as such. Another important difference is that in the UK, for example, government ownership of land is applied only for small areas within the designated parks. The parks of Western Europe have always incorporated the principles of eco-development involving sustainable resource use and rural development (Blacksell, 1982; Harmon, 1987; West & Brechin, 1991).

By contrast, parks in developing countries, with people on one side and 'Nature' on the other side of PAs boundaries, tend to resist expanding agricultural production. In Africa, for example, a holistic, comprehensive and gradual approach is needed to find socially and economically viable solutions to a variety of problems in national parks (S.J. Milton, J.N. Blignaut, personal communications).

Protected areas as emerging ecosystems

We consider PAs—especially those in developing countries—as belonging to the large and growing number of "EEs." The concept of "emerging" or "novel" ecosystems" (Hobbs et al., 2006; Milton, 2003) deals with land units partially or totally transformed by people that have an uncertain ecological and economic trajectory when management activities cease. These are ecosystems that develop after social, economic and cultural conditions change, allowing new biotic assemblages to colonise and persist for decades with positive or negative social, economic and biodiversity consequences.

The EEs concept is valuable for calling attention to the over-riding role that human activities will play in determining the direction and the dynamics of ecological communities and ecosystems in space and time (Sanderson, Jaiteh, & Levy, 2002; Vitousek, Mooney, Lubchenco, & Melillo, 1997). It is a logical extension of the "Flux of Nature" paradigm (Botkin, 1990; Pickett, Parker, & Fiedler, 1992) that is rapidly replacing the hoary ''Balance of Nature'' paradigm of past centuries. Figure 2 illustrates the idea that at any given moment in time, t_0, the value of an EE, V_0, including PAs, is determined by a large number of variables related to:

(a) biophysical characteristics of the ecosystem itself (number and diversity of species, degree of endemism, rates of production of different goods and services, climatic conditions, etc.);
(b) localisation (i.e. exposure and vulnerability to natural disasters, proximity to areas affected by climate change, distance to urban centres, etc.);
(c) existing scientific knowledge (regarding biophysical and hydrological functions, biodiversity, potential for future production of pharmaceutical products, etc.);
(d) relevant social aspects (such as proximity to human settlements, distance to main markets, presence of indigenous people within its borders, etc.);
(e) cultural aspects (such as the perceptions of different groups about the ecosystem and its characteristics and perceived beauty feelings and regarding animals' rights, religious beliefs regarding nature in general, etc.);
(f) economic aspects (e.g. relative scarcity of the goods and services provided by the ecosystem, income level of the local population, availability of resources for conservation, etc.); and
(g) institutional aspects (nature and the degree of protection of prevailing property rights, the existence and effectiveness of legislation, the degree of law enforcement, etc.).

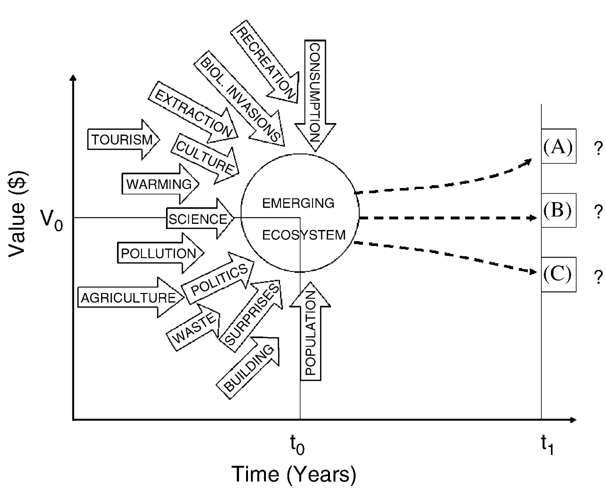

Figure 2. Multiple determinants of the monetary value of 'emerging ecosystems', including Protected Areas, and three possible trajectories for the future. Management, and aftercare planned in conjunction with restoration are absolutely critical to the process.

Economists usually use money as a unit of measure (as in Fig. 2), because it is universally well-known. However, a given number of units of any good or service could be used as an alternative unit of measure when valuing an ecosystem, in which case the value would be expressed in units of the chosen good.

The full value of a PA, cultural landscape or managed ecosystem is equal to that attached to it by individuals and society, and can be expressed not only in money terms but in different units of measure. Depending on the influences exercised by the numerous different variables determining the value of a PA or an EE its value can increase, remain constant or decrease over time. For example, if uncontrolled hunting, excessive extraction of products, continuous soil erosion, pollution of water resources, and deterioration of aesthetic amenities strongly impact a PA, its value will be pushed downward over time. In this case, the value of the PA will decrease from V_0, at time t_0, to a point like C, at time t_1, in Fig. 2.

Conversely, the recognised value of a PA will tend to increase if the predominant total effect of human activities is one resulting from variables such as the development and public diffusion of new scientific knowledge. These can emphasise the importance of life-preserving ecosystem functions, the development of new forms of alternative and sustainable tourism with their associated income flows for indigenous people (Figueroa, Bravo, & Alvarez, 2003; Pierce et al., 2002), the increase in the cultural appreciation of outdoor life and nature-related activities, and the rise in food and materials that can be sustainably extracted each year. Additional factors, in future, will include the potential increase in the ecosystem's annual rate of carbon sequestration and awareness about related beneficial climate and health effects. In these cases, the value of the PA will follow trajectory A.

Expanding the notion of linkages

People perceive costs and benefits of PAs in complex ways depending, on their education, social background, personal experience and specific knowledge about nature, ecology, hydrology, chemistry, psychology, etc. Thus, it is just as crucial for conservation advocates and planners to create effective linkages between PAs and the relevant social groups, as it is to create linkages among PAs. Conservation requires that all people concerned understand the short- and long-term benefits obtained, as well as the costs they must bear, from parks and PAs, in all their permutations. Well-grounded and nurtured connections between PAs and social groups will allow more informed social perceptions and, hopefully, better social decisions.

Monetary economic value and conservation assets ("natural capital") are not competing concepts (Boxall & Beckley, 2002). Indeed, as we confront global scarcity of, and competition for land, water and energy, we must face our liabilities and our vulnerability. With regards to biodiversity conservation, and ecosystem services in PAs, the two terms, monetary value and natural capital, should be seen as the two faces of a coin, since the first is nothing else than a quantitative measure of the latter, determined by individual and social preferences and expressed in commonly used units (money). Ecosystem goods and services, as well as biodiversity conservation in PAs, depend directly upon how well the PA networks actually serve people, especially the rural poor, who are often the most directly affected. In brief, the fates of rural people and of protected conservation areas are inextricably woven together. In fact, in the context of the EE model discussed here, it also embraces urban people and natural resources in general.

The nature, scale and duration of human activities are determined by individual and collective decisions, which result in the allocation of scarce resources to meet competing needs. The resources allocated reflect not only the desire for, but also the cost incurred in obtaining its benefits. Indeed, conservation of species or ecosystems provides satisfaction or benefits, just like any other good or service that meet human needs. Therefore, conservation, active management, and restoration, all compete with other human needs (e.g. food, health, education, shelter, etc.) for the allocation of scarce resources.

From an economist's perspective, it is easy to predict the outcome when desirable limited resources are available; those human needs most valued will be satisfied preferentially. Therefore, PAs will expand, remain the same or be reduced depending on the relative competing social needs. This applies also to the efforts invested in restoring or ameliorating landscape linkages (Figueroa in press; Milton, Dean, & Richardson, 2003).

An expansion in PA linkages and their added "value" is urgently required, particularly in densely populated countries like India or South Africa where little or no "empty" land exists, and where cultural landscapes and "SESs" have predominated for millennia. This is equally applicable to sparsely populated arid lands with extremely limited carrying capacities for people and livestock. Indeed, it probably applies in all developing countries (Amend & Amend, 1992).

Increasing the value of PAs and expanding conservation and restoration

Individuals and societies allocate their limited resources so as to obtain the maximum welfare possible. This implies satisfying the needs they choose to meet, or anticipated in the long-term. Therefore, to take decisions, they need to assess the relative value of resources in order to choose those that provide the highest value per unit of costs incurred in attaining them. Hence, the conservation, management, and, where needed, the expensive process of restoration, will be more likely chosen by individuals and societies who place a larger value on the importance of these areas.

As suggested the value of a PA or an EE corresponds to the sum of the values of the goods and services provided to individuals and to society. For those goods and services traded in explicit markets, such as minerals, timber, etc., it is possible to obtain their market prices. Indeed, calculating their respective monetary values is straightforward, since market prices provide much of the necessary information. However, many other ecosystem goods and services, e.g. clean air, erosion protection, carbon sequestration, aesthetic amenities and recreational facilities, do not have explicit market prices. Over the last four decades, economic science has developed techniques to evaluate these kinds of goods and services. Thus, in general, there are means available to calculate the value of PAs, with economic science showing the relative valuation assigned with respect to other goods and services (Campbell & Luckert, 2002; Pearce & Moran, 1998; Turpie, Heydenrych, & Lamberth, 2003).

For a PA to enjoy an increase in its recognised value, i.e. its SES value, people must perceive that the goods and services received are more valuable than in the past. For example, greater access, allowing increased, yet sustainable consumptive use value (wood, meat, fruits, etc.) would provide more benefits to local people, though management of resources may be required to prevent over-exploitation.

A more sustainable alternative for benefit sharing is to make the PAs a hub of rural economies. Such activities in and around parks can include night drives, guided tours, up-market hotels and restaurants. In addition, "tourism levies" on entrance fees can be used to upgrade roads and other services in the PA's neighbourhood, or agreements made to purchase locally grown or made products. When applied as an alternative or supplement to direct use of natural resources, these approaches provide greater incentives and tangible benefits for poor and wealthy neighbouring communities alike (Privett, Heydenrych, & Cowling, 2002).

Linkages to the landscape

Returning to the central subject of linkages, and addressing both conceptual and practical aspects, our generic EE in Fig. 3, including PAs in developing countries, is represented as a series of three concentric circles. The middle circle represents all ecological linkages of the ecosystem to its surrounding environment and, in the context of PAs, to other PAs in a conservation network. The outer circle represents socio-economic linkages. In addition, we develop the issue of thresholds to indicate the possible range of future trajectories for a given ecosystem. These depend not only on the values assigned to them, and the resources devoted to management, but also on the ecological—or economic—thresholds crossed in the past.

In Fig. 3, the EE at the present is poorly 'linked' in both ecological and socio-economic terms, as indicated by the dotted lines of the outer concentric circles. To follow the path to scenario (A), efforts and investments are made to ameliorate linkages, as indicated in the nearly reintegrated concentric circles, and the increased SES value. In scenario (B), where no special effort is made, an ecological or socio-economic threshold may be crossed, leading to a loss of SES values. In scenario (C), where two thresholds are crossed, SES values can be expected to drop sufficiently, so that people decide, at last, to intervene. At this point, however, it is doubtful to what extent, and how quickly, SES values can be recovered.

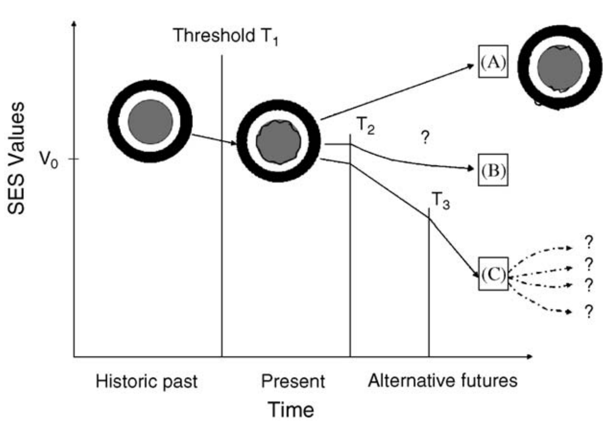

Figure 3. Typical past trajectory and three possible futures for an emerging ecosystem crossing one or more thresholds.

Creating landscape linkages in the real world

Today a variety of means exist that can be employed to create the necessary linkages between rural populations and PAs to increase conservation support and/or restoration efforts in developing countries. For example, information and communication technologies (ICT) provide cheap and readily available means to reach rural communities and create the needed interactions with conservation planners, government agencies, NGOs or private companies implementing conservation strategies or restoration programmes.

The creation and maintenance of appropriate linkages between PAs and social groups will be even more important the further these groups are from protected and rural areas in general. This is because people and interest groups living far away can be expected to be less aware of the goods, services and amenities that these areas provide for them. Hence, a larger effort may be needed to properly inform these people of the benefits they perceive from conservation and PAs and from natural capital restoration.

Creating effective linkages of PAs with interest groups will represent a challenging task for conservation governmental agencies in many developing countries, since their experience in carrying out community participation has been, in general, rather weak. Even worse, in some developing countries, rural populations and indigenous groups have become sceptical and distrusting towards community participation initiatives led by government agencies. Unfortunately, many past participatory programmes did not deliver what they promised when they were implemented. In Africa, Asia and Latin America, numerous examples of integrated rural development programmes, with ambitious community participation components, were embarrassing failures (e.g. Alpert, 1996; Emerton, 1999, 2000) in spite of having financial and technical support from multilateral and international organisations. The work carried out by certain NGOs with rural and marginal urban communities in some developing countries since the early 1990s could represent a valuable experience for creating effective social linkages with PAs in the future.

Conclusions

We have argued that the concepts of EEs and SES, which emphasise the constant evolution of ecosystems and increasing human influences, provide an appropriate framework to analyse and develop linkages of PAs in all their permutations. Therefore, within this analytical context, the values of PAs are predominantly determined by individual and social beliefs, perceptions, attitudes and actions; while the fate of PAs depends on human decisions. Thus, the conservation of species, habitats and ecosystem goods and services is impossible without active human management and investment in linkages. Likewise, restoring PAs and natural capital in general will be more acceptable economically and socially if such linkages are developed.

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References

Alpert, P. (1996). Integrated conservation and development projects – examples from Africa. *BioScience*, *46*, 845–855.

Amend, S., & Amend, T. (Eds.). (1992). *Espacios sin habitantes? Parques Nacionales de América del Sur.* Caracas: IUCN, Gland and Editorial Nueva Sociedad.

Araùjo, M. B., Williams, P. H., & Fuller, R. J. (2002). Dynamics of extinction and the selection of nature reserves. *Proceedings of the Royal Society of London B*, *269*, 1971–1980.

Aronson, J., Clewell, A., Blignaut, J., & Milton, S. J. (2006). Ecological restoration: A new frontier for nature conservation and economics. *Journal for Nature Conservation, 14*, 135–139.

Bawa, K., Seidler, R., & Raven, P. (2006). Reconciling conservation paradigms. *Conservation Biology, 18*, 859–860.

Bennett, A. F. (2003). *Linkages in the landscape: The role of corridors and connectivity in wildlife conservation*. Gtland, Switzerland, Cambridge, UK: IUCN.

Blacksell, M. (1982). The Spirit and Purpose of National Parks in Britain. *Parks*, *6*, 14–17.

Botkin, D. (1990). *Discordant harmonies: A new ecology for the twenty-first century.* Oxford, UK: Oxford University Press.

Boxall, P. C., & Beckley, T. (2002). An introduction to approaches and issues for measuring non-market values in developing countries. In B. M. Campbell, & M. K. Luckert (Eds.), *Uncovering the hidden harvest: Valuation methods for woodland and forest resources* (pp. 103–152). London: Earthscan.

Campbell, B. M., & Luckert, M. K. (Eds.). (2002). *Uncovering the hidden harvest – Valuation methods for woodlands and forest resources. People and Plants Conservation Series*. London, UK: Earthscan Publications.

Carpenter, S., Walker, B., Anderies, J. M., & Abel, N. (2001). From metaphor to measurement: Resilience of What to What? *Ecosystems, 4*, 765–781.

Council of Europe. (2000). *The European landscape convention*. Strasbourg.

Cowling, R. M., Pressey, R. L., Lombard, A. T., Desmet, P. G., & Ellis, A. G. (1999). From representation to persistence: Requirements for a sustainable system of conservation areas in the species-rich mediterranean-climate desert of southern Africa. *Diversity and Distributions*, *5*, 51–71.

Cunningham, A. B. (2001). *Applied ethnobotany: People, wild plant use and conservation*. London: Earthscan.

Davies, K. F., Margules, C. R., & Lawrence, J. F. (2000). Which traits of species predict population decline in experimental forest fragments? *Ecology, 81*, 1450–1461.

Diamond, J. M. (1975). The island dilemma: Lessons of modern biogeographic studies for the design of natural reserves. *Biological Conservation*, *7*, 129–146.

Dobson, A. P., Rodriguez, J. P., Roberts, W. M., & Wilcove, D. S. (1997). Geographic distribution of endangered species in the United States. *Science*, *275*, 550–553.

Driver, A., Cowling, R. M., & Maze, K. (2003). *Planning for living landscapes: Perspectives and lessons from South Africa*. Washington, DC: Center for Applied Biodiversity Science at Conservation International; Cape Town: Botanical Society of South Africa. Available at www.botanicalsociety.org.za/ccu or www.biodiversityscience.org

Emerton, L. (1999). *The nature of benefits and the benefits of nature: Why wildlife conservation has not economically benefited communities in Africa*. Report No. 9. Community Conservation Research in Africa: Principles and Comparative Practice. Global Environmental Change Programme, Economic and Social Research Council, United Kingdom.

Emerton, L. (2000) Using economic incentives for biodiversity conservation. IUCN Economics and Biodiversity Programme LAE@iucnearo.org

Faith, D. P., & Walker, P. A. (1996a). Integrating conservation and development: effective trade-offs between biodiversity and cost in the selection of protected areas. *Biodiversity and Conservation*, *5*, 431–446.

Faith, D. P., & Walker, P. A. (1996b). Integrating conservation and development: Incorporating vulnerability into biodiversity-assessment of areas. *Biodiversity and Conservation*, *5*, 417–429.

Farina, A. (2001). The cultural landscape as a model for the integration of ecology and economics. *BioScience*, *50*, 313–320.

Figueroa, E. B. (in press). Restoring natural capital: An economic perspective. In Aronson, J., Milton, S. J., & Blignaut, J. N. (Eds.), *Restoring natural capital: Science, business and practice*. Washington, DC: Island Press (to appear in 2007).

Figueroa, E., Bravo, C., & Alvarez, R. (2003). Biodiversidad y Turismo: Oportunidades para el Desarrollo Económico y la Conservación en Chile. In E. Figueroa, & J. A. Simonetti (Eds.), *Biodiversidad y Globalización: Oportunidades y Desafíos para la Sociedad Chilena* (pp. 241–302). Santiago, Chile: Editorial Universitaria.

Gadgil, M. (1995). Prudence and profligacy: A human ecological perspective. In T. M. Swanson (Ed.), *The economics and ecology of biodiversity decline* (pp. 99–110). New York: Cambridge University Press.

Gadgil, M., & Guha, R. (1995). *Ecology and equity. The use and abuse of nature in contemporary India*. New Delhi: Penguin Books.

Hardin, G. (1985). Human ecology: The subversive, conservative science. *American Zoologist*, *25*, 469–476.

Harmon, D. (1987). Cultural diversity, human subsistence, and the National Park Ideal. *Environmental Ethics*, *9*, 147–158.

Hobbs, R. J., Arico, S., Aronson, J., Baron, J. S., Bridgewater, P., Cramer, V. A., et al. (2006). Novel ecosystems: Theoretical and management aspects of the new ecological world order. *Global Ecology and Biogeography*, *15*, 1–7.

Hollick, M. (1981). The role of quantitative decision making methods in environmental impact assessment. *Journal of Environmental Management*, *12*, 65–78.

IUCN, UNEP and WWF. (1991). *Caring for Earth. A strategy for sustainable living*. Switzerland: Gland.

Janzen, D. (1998). Gardenification of wildland nature and the human footprint. *Science*, *279*, 1312–1313.

Janzen, D. (1999). Gardenification of tropical conserved wildlands: Multitasking, multicropping and multiusers. In *Proceedings of the National Academy of Science*, USA 96, pp. 5987–5994.

MacArthur, R. H., & Wilson, E. O. (1967). *The theory of Island biogeography*. New Jersey: Princeton.

Margules, C. R., & Pressey, R. L. (2000). Systematic conservation planning. *Nature*, *405*, 243–253.

Milton, S. J. (2003). Emerging ecosystems – A washingstone for ecologists, economists and sociologists? *South African Journal of Science*, *99*, 404–406.

Milton, S. J., Dean, W. R. J., & Richardson, D. M. (2003). Economic incentives for restoring natural capital: Trends in southern African rangelands. *Frontiers in Ecology and Environment*, *1*, 247–254.

Moreira, F., Queiroz, A. I. & Aronson, J. (2006). Restoration principles applied to cultural landscapes. *Journal for Nature Conservation, 14*, 217–224.

Pearce, D., & Moran, D. (1998). The economics of biological diversity conservation. In P. L. Fiedler, & P. M. Karieva (Eds.), *Conservation for the coming decade* (pp. 384–395). New York: Chapman & Hall.

Pickett, S. T. A., Parker, V. T., & Fiedler, P. L. (1992). The new paradigm in ecology: Implications for conservation biology above the species level. In P. Fiedler, & S. Jain (Eds.), *Conservation biology: The theory and practice of nature conservation, preservation and management* (pp. 65–88). New York: Chapman & Hall.

Pierce, S. M., Cowling, R. M., Sandwith, T., & MacKinnon, K. (2002). *Mainstreaming biodiversity in development*. Washington, DC, USA: The World Bank Environment Department.

Privett, S. D. J., Heydenrych, B. J., & Cowling, R. M. (2002). Putting biodiversity to business on the Agulhas Plain. In S. M. Pierce, R. M. Cowling, T. Sandwith, & K. MacKinnon (Eds.), *Mainstreaming biodiversity in development: Case studies from South Africa* (pp. 101–115). Washington: The World Bank.

Sanderson, E. W., Jaiteh, M., Levy, M. A., et al. (2002). The human footprint and the last of the wild. *BioScience*, *52*, 891–904.

Terborgh, J. (1999). *Requiem for nature*. Washington, DC: Island Press.

Turpie, J. K., Heydenrych, B. J., & Lamberth, S. J. (2003). Economic value of terrestrial and marine biodiversity in the Cape Floristic Region: Implications for defining effective and socially optimal conservation strategies. *Biologicial Conservation*, *112*, 233–251.

Vitousek, P., Mooney, H. A., Lubchenco, J., & Melillo, J. (1997). Human domination of ecosystems. *Science*, *277*, 494–499.

Wackernagel, M., Schulz, N. B., & Deumling, D., et al. (2002). Tracking the ecological overshoot of the human economy. In *Proceedings National Academy of Science*, USA, Vol. 99, pp. 9266–9271. http://www.pnas.org/cgi/doi/10.1073/pnas.142033699, accessed 31 December 2003.

West, P. C., & Brechin, S. R. (Eds.). (1991). *Resident peoples and National Parks: Social dilemmas and strategies in International Conservation*. Tucson, USA: The University of Arizona Press.

Williams, P. H., Moore, J. J., Kamden Toham, A., Brooks, T. M., Strand, H., D'Amico, J., et al. (2003). Integrating biodiversity priorities with conflicting socio-economic values in the Guinean-Congolian forest region. *Biodiversity and Conservation*, *12*, 1297–1320.

Wilson, E. O., & Willis, E. O. (1975). Applied biogeography. In M. L. Cody, & J. M. Diamond (Eds.), *Ecology and evolution of communities* (pp. 522–534). Cambridge, MA: Belknap.