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Review

Plant conservation: old problems, new perspectives

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Abstract

Nature conservation has changed from an idealistic philosophy to a serious technology
(J. Harper, 1992)

A review is given of the major conceptual changes that have taken place during the last 50 years in our understanding of the nature of plant conservation and of the principal methodological advances in undertaking conservation assessments and actions, largely through the incorporation of tools and techniques from other disciplines. The interrelationships between conservation and sustainable use are considered as well as the impact of the development of the discipline of conservation biology, the effects of the general acceptance of the concept of biodiversity and the practical implications of the implementation of the Convention on Biological diversity. The effect on conservation policy and management of the accelerating loss or conversion of habitats throughout the world and approaches for combating this are discussed.

Keywords: Conservation biology; Conservation strategies; Conservation tools; Convention on Biological Diversity; Management; Sustainable development

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1. Introduction

The past 50 years have witnessed a major evolution in our understanding of conservation and its interrelationship with the elusive goal of sustainable development. This period has also been marked by two antagonistic trends. On the one hand, there has been the rise of environmentalism and the establishment of conservation-orientated institutions and organisations and the negotiation and implementation of a series of treaties and other instruments that affect the ways in which most countries address these issues. On the other hand, the scale and intensity of human interactions with the environment have led to progressive and widescale habitat loss and degradation and fragmentation, with subsequent loss of species and genetic variability. As Wood et al. (2000) comment 'The race to save biodiversity is being lost, and it is being lost because the factors contributing to its degradation are more complex and powerful than those forces working to protect it.'

As a consequence, we are still faced with the old problems of how to address the conservation and management of protected areas, habitat loss, species loss, species recovery, but now with an increasing sense of urgency, leading to debates on how to set priorities, questioning of previous orthodoxy (even the role of protected areas) and searching for new approaches and tools for diagnosis and decision making in conservation and management.

In this introductory review, we explore these issues, offer some definitions and a route map for the way ahead. This special issue of *Biological Conservation* gathers contributions written by participants at the II Workshop for the Conservation Biology of Plants held in Madrid. The spirit of this workshop was to assess current trends in plant conservation brought about by the integration of new approaches, techniques and methodologies, often incorporated from other disciplines. Not all the topics we raise here are covered directly by the papers in this special issue of *Biological Conservation* but we have included them so as to provide a more or less comprehensive overview.

2. Conservation, conservation biology, biodiversity and ecology

2.1. Conservation

The term 'conservation' is an ambiguous one and has had different meanings to different people and constituencies over time. The ambiguity of the term conservation also derives from its having two roots (Jordan, 1995)—one in resource management and the other in natural history. The concern with resource management, which considered that biological resources had to be managed in such a way that was not wasteful and ensured that they did not become exhausted or extinct, is a long-standing one and was clearly enunciated by the North American forester Gifford Pinchot who equated conservation with the systematic exploitation of natural resources (Pinchot, 1947). Natural history roots of conservation are still strong and expressed through concern at the loss of species and of the degradation or loss of the 'wilderness' aspect of our natural landscapes. It is this populist perception of conservation, with its focus on familiar values such as known and well-loved habitats or cherished species, that is still the main source of public support for conservation action.

One of the greatest changes in our perception of conservation stems from an increasing realisation of the amount of dynamism that exists in natural systems. This is reflected in Leopold's (1949) evolutionary-ecological view of the ecology of living world as a complex and integrated system of interdependent processes and components, which foreshadowed current preoccupations with the maintenance of ecosystem health. This has led to another perspective that regards conservation as essentially management for change (cf. Luken, 1990), and emphasises the dynamics of the ecosystems, species and populations that we wish to conserve. Pickett et al. (1992) suggest that 'one does not conserve vegetation, which is a thing, but rather one is attempting to conserve a dynamic'. To this has to be added the impending problems of global change—both demographic and climatic—that hang over us like a sword of Damocles.

The context of conservation changed significantly following the UN Conference on the Environment at

Rio in 1992, itself the culmination of a remarkable series of major international initiatives in the preceding 30–40 years. The subsequent ratification by most of the world's governments of the *Convention on Biological Diversity* (CBD) marked a turning point and has placed the subject of biodiversity firmly on the political agenda. At a stroke, conservation ceased to be an optional extra and became official, global and national policy. But what kind of conservation and how it was to be implemented is still under debate. The CBD did not in itself resolve the issues of conservation of biological diversity—indeed the term conservation is not even defined in the Convention—rather, it raised a debate that still continues on how it may be interpreted by the Parties. It is significant that the Convention refers to 'conservation of biological diversity' and to 'sustainable use of its components' as separate matters, although the idea that conservation and sustainable use are necessarily linked is now widely accepted. The most plausible explanation for this is that it reflects the concerns of developing countries who wished to place emphasis on the use of the components of biological diversity (in a sustainable manner) and did not wish to see a shift in emphasis towards the 'preservationist' aspect.

Today, different proponents of conservation put more emphasis on one or other of these approaches although the 'resourcist' view of nature dominates at present, especially following the publication of the *World Conservation Strategy* (IUCN/UNEP/WWF, 1980), which defined conservation as 'the management of human use of the biosphere so that it may yield the greatest sustainable benefit to present generations, while maintaining its potential to meet the needs of and aspirations of future generations. Thus conservation is positive, embracing preservation, maintenance, sustainable utilisation, restoration and enhancement of the natural environment'. The resource and natural history concerns can be combined into a single definition such as the following adapted from Jordan (1995): '*conservation is a philosophy of managing the environment in such a way that does not despoil, exhaust, or extinguish it or the resources and values it contains*.' This allows for the more strictly preservation aspects, such as the preservation of flagship species, ex situ storage of germplasm, and the protected areas approach to be covered as well as the sustainable use of resources and the maintenance of environment health.

There is also an increasing tendency today to accept a broader integrative or holistic view that combines elements of them all. An example of this can be seen in the Global Strategy for Plant Conservation adopted by the Conference of the Parties of the Convention on Biological Diversity at its sixth meeting in April 2002 as well as in the European Plant Conservation Strategy developed by the Council of Europe and Planta Europa. Targets both in the Global Strategy and the European Strategy address five major objectives that, as a whole, integrate the earlier-mentioned holistic view: (1) understanding and documenting plant diversity; (2) conserving plant diversity; (3) using plant diversity sustainably; (4) promoting education and awareness about plant diversity and (5) building capacity for the conservation of plant diversity.

2.2. Conservation biology

> Conservation biology is the new, multidisciplinary science that has developed to deal with the crisis confronting biological diversity (Primack, 1993)

Of course, conservation is more than a concept and much effort has been devoted to resolving the scientific, technical, sociological and economic issues involved in implementing effective conservation action. This has manifested itself to a large degree in the rise of the discipline known as conservation biology that has been described as 'a recent response of the scientific community to the wave of global environmental change that is threatening to extinguish a very large fraction of the world's biological diversity' (Soulé and Kohm, 1989). It is considered to have its origin in the First International Conference on Conservation Biology held in San Diego, California in 1978 that resulted in the publication in 1980 of a book edited by Soulé and Wilcox, *Conservation Biology: An Evolutionary-Ecological Perspective*. The Society for Conservation Biology was founded in 1985, and was 'dedicated to promoting the scientific study of the phenomena that affect the maintenance, loss, and restoration of biological diversity'.

Conservation biology is a synthetic, multidisciplinary science that evolved since the 1980s, feeding on a variety of other areas of biology, notably ecology, demography, population biology, population genetics, biogeography, landscape ecology, environmental management and economics. It has also spawned new areas such as conservation genetics (Avise and Hamrick, 1996), metapopulation ecology dynamics and biology (Hanksi, 1999; Hanski and Gilpin, 1997), restoration biology and ecology (Jordan et al., 1987), fragmentation biology and patch dynamics. The papers in this volume explore some of these aspects.

Conservation biology is often termed a crisis discipline, in that it arose in response to the dramatic loss of biodiversity that was being documented across the world and the need to take steps to anticipate, prevent and repair this situation. In the words of Soulé and Kohm (1989), 'Conservation biologists view their main task as providing the intellectual and technological tools that will anticipate, prevent, minimize, and/or repair ecological damage'.

2.3. Biodiversity

The development of the concept of conservation biology is closely interrelated with that of biodiversity. Like conservation biology, the notion of biological diversity, later contracted as biodiversity, developed in the 1980s although its origins go back much earlier (Heywood et al., 1995). Again, like conservation biology it is a synthetic discipline (Heywood, 1994), serving as a focus where many disciplines and activities meet and interact. Indeed the disciplines feeding biodiversity are broadly the same—evolutionary biology, taxonomy, ecology, genetics.

The concept of biological diversity, or biodiversity for short, has risen into prominence in a remarkably short space of time. It is perhaps the dominant notion in environmental thinking, planning and action and the signing and entry into force of the *Convention on Biological Diversity* has placed it on both the international and political stages. Yet there is still considerable debate as to what biodiversity is, whether it is a meaningful concept or just a passing fashion, whether it can be treated as a rigorous discipline and whether it merits all the attention that it seems to attract.

The basic problem is the attractive simplicity of the idea: biodiversity as the 'variety of life' is the kind of general intuitive definition that most people share. It is when we attempt to apply more precise or rigorous definitions that we face difficulties: we then find that a variety of different notions or viewpoints about biodiversity have developed in the past 15 years. Gaston (1996a) suggests that these can be broadly grouped under three headings: those that regard biodiversity as a concept; those that regard it as a measurable entity; and those that regard it as a social or political construct.

The most frequently employed definitions distinguish between the major components or levels of biodiversity that can be recognised—ecosystems, species and genes—and these are recognised in the definition given in Article 2 of the Convention on Biological Diversity, as follows:

> 'Biological diversity' means the variability among living organisms, from all sources, including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems.

This has been modified in the definition adopted in the Global Biodiversity Assessment (Heywood, 1995: Box 1.2-1) as genetic, ecological and taxonomic diversity together with cultural diversity and this gives us a useful framework in which to plan assessments. The three main components have a hierarchical nature and overlap. It should be noted that the population is the unit common to all three levels/components. As Gaston (1996b) observes, such schemes serve mainly to clarify the breadth of the concept and are useful human constructs, like many of the devices we use in biology and especially taxonomy.

The Global Biodiversity Assessment also notes that strictly speaking the word diversity refers to the quality, range or extent of the differences between the different biological entities in a given set. Thus in total it would amount to the diversity of all life and is a characteristic of nature or life itself, not an entity or a resource. On the other hand, the term has come to be widely used to refer to the set of diverse organisms themselves, i.e. not the diversity of life but all life itself (Heywood, 1995). This recalls the dual use of the term 'taxon' for both the concept of a taxonomic group and actual named entities.

A major recent contribution to biodiversity theory has come from a book *The Unified Neutral Theory of Biodiversity and Biogeography* by Hubbell (2001), already hailed as a major conceptual breakthrough, that presents a new, general neutral theory to explain the origin, maintenance, and loss of biodiversity. It must be noted, however, that the author defines biodiversity 'to be synonymous with species richness and relative species abundance in space and time'. This theory is constructed on the foundation of the equilibrium theory of island biogeography of MacArthur and Wilson (1963, 1967). Including speciation and changing the neutrality assumption enables this new theory to predict not only species richness on islands and on the mainland, but also the relative abundance of species, species–area relationships, and phylogeny under ecological drift, random dispersal, and random speciation. The theory considers that ecological communities are open, nonequilibrium assemblages of species largely thrown together by chance, history, and random dispersal. As a consequence, ecological communities are seen in perpetual taxonomic nonequilibrium, undergoing continual endogenous change and species turnover through repeated immigrations and local extinctions.

2.3.1. Biodiversity indicators

One of the aims of subdividing the concept of biodiversity into its various components is to facilitate its measurement and submit it to rigorous comparative study. It would seem that biodiversity itself cannot be measured or reduced to a single measure (Norton, 1994) and we have in fact to choose which aspects of biodiversity we plan to measure. This will depend on our purpose, and since biodiversity has many constituencies, a whole range of different measures will be used although it is not always obvious what the information needs of different users of biodiversity are. Such measures are also necessary if we are to be able to derive indicators or indices of biodiversity that will allow us to monitor what is happening to biodiversity in space and time. Indeed one of the current concerns of the Parties to the Convention Biological Diversity is which indicators to select to measure changes in biodiversity (UNEP/CBD/SBSTTA, 1997) and IUCN proposed a core set of issues with a menu of indicators to the SBSTTA (IUCN, 2000b). Previously, about the only widely reported biodiversity indicators are lists of endangered species, statistics on the amount of 'wilderness' areas left and the percentage of land afforded some degree of protection (Hammond et al., 1995). In a review of recent approaches to biodiversity indicators, Hansson (2001) distinguishes between policy indicators and those used for management and monitoring. He discusses single species vs. community indicators, statistical indicators and functional indicators, and refers to the suggestion by Noss (1990) that hierarchy theory should be applied in the selection of indicators. It is quite clear that to address the whole of biodiversity and its composition, structure and function, many different indicators need to be applied at the different levels of organisation and as Delbaere (2002) notes, 'Despite the efforts that have been made to develop sound indicator sets and monitoring schemes, there is still a big discrepancy between the scientific development and policy requirements'.

2.3.2. Biodiversity and ecosystem functioning

A related topic to indicators is the question of ecosystem functioning. This has been the subject of intensive study, including a programme launched in 1991 by the Scientific Committee on Problems of the Environment (SCOPE) to assess the state of our knowledge of the role of biodiversity, in all its dimensions, in ecosystem and landscape processes. It was designed to answer the two overarching questions: (1) does biodiversity 'count' in system processes (e.g. nutrient retention, decomposition, production, etc.), including atmospheric feedbacks, over short- and long-term time spans, and in face of global change (climate change, land-use, invasions)? and (2) how is system stability and resistance affected by species diversity, and how will global change affect these relationships? Most of these studies have been based primarily on observational data, detailed summaries of which are given by Mooney et al. (1995a,b, 1996), and the International Biosphere Geosphere Programme has now taken on the task of bringing experimentation and more organised observations to this research field. Some research has been used to demonstrate benefits to ecosystem function arising from higher levels of biodiversity and this supports the suggestion that certain levels of biodiversity are essential for the functioning and/or sustainability of an ecosystem although this view is not shared by all ecologists (Grime, 1997).

One of the results of the ecosystem functioning programme was 'the capacity to predict which species will cause the greatest system impacts, and hence the greatest ecosystem services changes, when added or deleted' (Mooney et al., 1995a). This addressed the issue of identification of whether there are 'keystone species', i.e. species that have a uniquely important effect on the ecosystem, or more generally, whose impacts on its ecosystem are greater than would be expected from their relative abundance or biomass, and whose removal would lead to dramatic changes, a concept first formalised by Paine (1969). The fact is that we are still very uncertain as to which species play a key role in the maintenance and proper functioning of most ecosystems (Lawton, 1994), and the concept has been relaxed as 'only those species having a large, disproportionate effect, with respect to their biomass or abundance, on their community' (Power and Mills, 1995).

Since many ecosystems around the world are currently undergoing dramatic loss or changes in species composition due to the influence of human activity, it would be a matter of considerable importance for biodiversity conservation if we had a better understanding of the ways in which ecosystems function and of which of their component species played a key role. This would provide a scientific underpinning for conservation measures and help us in focusing our efforts and resources. It could, however, be a two-edged sword in that it could be used to suggest which species are 'disposable' or redundant in an ecosystem and whose loss could then be tolerated.

2.3.3. Human influences on biodiversity

Cultural biodiversity and human interactions at all level with biodiversity are also included as an additional dimension to the organismic, genetic and ecological levels. In fact in *Global Biodiversity Assessment* (Heywood, 1995), a recurring theme is the interaction between humans and the environment. Humans are seen as the dominant influence on biodiversity and the scale of transformation, management and utilisation of ecosystems in the past two to three centuries is enormous—so much so that no part of the world can be considered as truly 'undisturbed'. The Assessment also makes it clear that humans have to be seen not so much as the problem as part of the solution and focus is given to socio-economic strategies for the sustainable use, conservation and sharing of the benefits of biodiversity. A much fuller assessment, *Cultural and Spiritual Values of Biodiversity* was later published (Posey, 2000) as a complement to the GBA.

2.4. Ecology

> Ecology is fuzzy. It does not fit into the literal pigeonholes of conventional science (Tyson, 2001)

It might be expected that ecological science is the major underpinning of conservation science and technology but as Harper (1992) has commented, it 'is still too immature to provide all the wisdom that it must'. It is arguable, he suggests, that the desire to conserve nature will in itself force the discipline of ecology to identify fundamental problems in its scientific goals and methods. Indeed there have been major changes in approaches to ecology in recent years, especially since the 1980s, coinciding with the rise of biodiversity and conservation ecology and feeding like them on the developing evolutionary theory. The previously current 'holistic' or 'equilibrium' paradigm' (Simberloff, 1982) with its emphasis on ecological systems as closed, self-regulating systems in their natural state, in equilibrium at their most mature state, subject to successional changes that will restore the original state and equilibrium when disturbed and degraded by outside forces, has been replaced by a new paradigm characterised by a recognition of the dynamic and changing nature of communities and ecosystems, the importance of process rather than end point, a shifting scale of focus, and the inclusion of humans. The new paradigm has been termed the 'non-equilibrium' paradigm by Pickett et al. (1992). Gone is the emphasis on the stable state of ecosystems that are closed and self-regulating, with humans as separate, and the notion of 'the balance of nature', to be replaced by an emphasis on dynamism, multiple pathways of vegetation change, the openness of ecosystems, recognition of the integral role of humans, and the metaphor of' 'the flux of nature' (Pickett et al., 1992).

The change in ecological paradigm discussed above has significant consequences for conservation, especially the recognition of the very considerable dynamism shown by communities and by their component species, which can lead to considerable species turnover and even local loss (especially of rare species) even in areas that have been set aside for conservation (Huntley, 1999). A combination of the natural dynamics of populations and the dynamics of succession and other factors, such as spatial and environmental heterogeneity and disturbance regimes, may well lead to considerable changes in the composition and structure of ecosystems over even short periods of time, unless management intervention modifies or steers it to some predetermined state. As Condit et al. (1992) have noted, 'no community of species achieves, let alone remains in, static equilibrium. Species continuously wax and wane in relative abundance; they even go extinct locally and reimmigrate.' This dynamism has considerable implications for conservation management and practice.

3. Conservation strategies in a dynamic world

3.1. Setting goals and values

The dynamism and sometimes rapid species turnover raises such issues as to which state or stage of the cycle of vegetation one wishes to preserve—maintenance of the status quo, a return to 'how it was' in some earlier period or what? The hands-off approach to conservation is a recipe for change (albeit sometimes cyclical) and risks, at least in the short term, the loss of elements of the ecosystem, such as particular species or combinations, whose presence may have been one the principal reasons for wishing to conserve the ecosystem in the first place. Whether or not the structure and composition of a forest or other biotope is constant and will return to its original composition in time after disturbance—the so-called equilibrium theory—or whether diversity is maintained at any one location by a balance between local extinction and immigration—the non-equilibrium theory is still being debated (Condit et al., 1992; Primack, 1992; Primack and Hall, 1992). Even the notion of a 'representative sample' within a complex ecosystem such as a forest is, according to Chazdon (1996) perhaps just as illusory as a pristine forest.

One of the main justifications for in situ conservation (and at the same time a failing of ex situ conservation) is that it allows evolutionary change to continue in the component species and populations but in addition to endogenous evolutionary (and ecological) change, a whole series of exogenous factors are also involved (Loreau et al., 1995). The diversity of species in a community or region can only be explained if abiotic factors, biological interactions, such as competition, predation, parasitism and mutualism, and their various indirect effects—ecosystem processes, temporal and spatial variability of the environment, regional processes and historical contingency and evolutionary processes—are all taken into account.

Vegetation and exogenous dynamics also raise questions such as: how effective are large vs. a series of small protected areas in maintaining biodiversity? how large should protected areas be to maintain rare species? how much change in composition occurs even after strict protection occurs? what are the rates of extinction on the ground as opposed to what is predicted by models or theories? what is the role of changes in the frequency of common species on forest structure? how different can a forest or other ecosystem become in floristic composition before it appears to have changed physiognomically or structurally?

3.2. Time and space scale of concern

Much light on many of the questions posed earlier can be shed if we take into account an appropriate time and space scale of concern. Temporal and spatial scales in combination are the key to the evaluation of natural change, both in terms of evolutionary change and ecological dynamics, and human impacts on nature. The assessment of a conservation goal, an ecological trend or an anthropogenic perturbation wholly depends on the temporal and spatial scales in which we operate.

Therefore, the 'contradictory' goal of conserving or preserving a biota that is dynamic and ever-changing can only be solved when appropriate temporal and spatial scales are set (Callicot, 1997). From this perspective, human environmental impacts should only be allowed when they tend to disturb the biotic community at spatial and temporal scales that are similar to natural disturbances (Callicot, 1997).

Many of the problems of conservation actions and policies are related to conflicts between actions and processes occurring at different scales. Such is the case of the time periods needed to investigate the life history of an endangered species, or to implement a species recovery plan with regard to the terms of research funding programmes, or conservation actions of the administration, which are tightly dependent on political terms of office. In a similar way, management and economic considerations often restrict the size of protected areas or restoration projects when these should be much larger if purely biological considerations were taken into account.

3.3. Conservation targets and cost-effective biodiversity planning

It is widely accepted today that the primary strategy for nature conservation is the establishment and maintenance of a system or network of protected areas. But as Huntley (1999) points out, in a changing world this is a necessary but not sufficient condition of the successful conservation of biodiversity. Successful conservation during the coming centuries of change accelerated anthropogenically will, he suggests, require that species are afforded protection wherever in the landscape they find themselves. Thus, recent research shows that the surroundings of an area may be just as important as the reserve itself (Perfecto and Vandermeer, 2002).

Some conservationists believe that efforts to expand and strengthen the global system of protected areas should be redoubled and at the same time dismiss the whole concept of sustainable development of resources as a misguided effort (e.g. Brandon, 1997; Kramer et al., 1997; Soulé and Sanjayan, 1998). Soulé and Sanjayan (1998) argue in support of such an expansion that 'lands outside strictly protected areas in the tropics, not to mention those in many temperate-zone nations, will be greatly diminished in their capacity to sustain native species and ecosystems by 2050, by which time human populations may have more than doubled.'

While there can be few who would not welcome a strengthening of the world's protected area systems, in the light of the continual loss of biodiversity throughout the world, such an exclusive policy is somewhat shortsighted as it ignores the realities of the world in which we live. Also, it puts too much emphasis on a single approach to biodiversity conservation—one that is, moreover, not without serious risks—and dismisses the contribution that areas that are not reserved can make to the maintenance of biodiversity.

Not only does the greater part of biodiversity exist outside any kind of formal protection but a great deal of biodiversity is also found even in agricultural systems, especially those of traditional small farmers. Many native plant species have benefited from agricultural activities concomitant with forest removal, especially those growing in more open types of landscapes or those taking advantage of anthropogenic circumstances such as trampling by livestock in pastures (de Blois and Bouchard, 1995). Little attention has been paid to the importance of the mosaic of farmland habitats for the conservation of native plant species. A number of studies have documented the importance of noncrop habitats as refuges for plant species typical of once-dominant regional vegetation (e.g. Jobin et al., 1996; Boutin and Jobin, 1998).

Even within nature reserves or protected areas, Huntley (1999) makes the point that 'the current paradigm of conservation management set against a static environment must be replaced by an approach that incorporates a realisation of the dynamic character of the environment and of the species assemblages'. He suggests that in future, the emphasis should be on the types of physical habitat that the reserve is able to offer.

The approach to protected area management has in fact changed considerably during the past 20 years. It is often stated, for example, that there has been a move away from the 'fortress' concept of a protected area to a more participatory approach, with the neighbours of protected area being treated as partners in conservation. It could be argued that the former was more of a notional concept than one that was enforced in practice: as Spinage (2002) has commented, 'the ... designation "fortress conservation" is patently absurd. The most that the majority of national parks can muster is a weak policing'. Anyhow, we have moved on from the earlier *laissez faire* ethic that dominated conservation philosophy in earlier decades towards acceptance of a much more interventionist approach on the one hand and the acceptance of a broader range of options and techniques for conservation on the other (e.g. Marrero et al., 2003).

One of the most important causes of loss of plant diversity is habitat fragmentation. Fragmented landscapes influence movement and dispersal of organisms, rates of gene flow, and invasion by exotic competitors, among many other factors. In the short term, as a result of fragmentation, individuals become more inbred and may have lower viability and fecundity due to inbreeding depression (see Oostermeijer et al., 2003). In the long term, a low level of connection between populations may also have detrimental genetic effects due to fixation of deleterious mutations. There is an urgent need to understand and quantify in detail the demographic and genetic processes that take place under fragmented habitat scenarios in order to generate appropriate conservation measures.

In a habitat fragmented territory a network of microreserves can help solve some of the needs. Microreserves are small-scale protected areas, usually less than one or two hectares, with a high concentration of endemic, rare or threatened species. Their small area and simplicity in legal and management terms allow them to be established in great number and to complement the larger, more conventional protected areas (Laguna et al., 1998).

Most reserves have not been located in places that contribute systematically to the sampling or representation of the biodiversity of the region. Only in recent years, has attention been focused on systematic conservation planning (Margules and Pressey, 2000; Groves et al., 2002), involving scientific prescriptions based on biogeographical theory, metapopulation dynamics and mapping. They also include techniques such as interactive geographic information systems (see Draper et al., 2003), decision trees, and complementarity—a measure of the extent to which an area, or set of areas, contributes unrepresented features (such as species) to an existing area or set of areas. Conservation planning also involves social, economic and political factors that may well modify priorities based on scientific theory and application. The need for such a consistent and structured approach to conservation planning is all the more urgent because of the enormous pressure on land and resources caused by global change, leading to complex patterns of fragmentation of landscapes and ecosystems where many of the options for conservation have already been foreclosed. A current initiative in this direction included in the IUCN Species Survival Commission's Plant Conservation Programme is the promotion of the conservation of important plant areas by refining the criteria for identification of Centres of Plant Diversity (Davis et al., 1994–1997) and other priority plant areas, for example, those that are natural or seminatural sites exhibiting exceptional botanical richness and/or supporting an outstanding assemblage of rare, threatened and/or endemic plant species and/or vegetation of high botanic value.

Conservation planning is also essential for effective conservation of plant genetic resources hotspots. Thus, Maxted (2003), indicates ways for efficient, active conservation of plant genetic resources in European protected areas and for identifying gaps in the in situ conservation of key resources for Europe.

Decision making is another element, inherent in all stages and areas of conservation, that is directly related to the cost-efficiency of the process. The decision to be made varies: sometimes it is a question of whether or not a species should be moved to the endangered list; in other cases, it may be how best allocate resources; or it can be deciding whether or not a remedial action is necessary after landscape degradation. Regardless of the specific nature of the decision problem, the common thread underlying these scenarios is the need to formally account for uncertainty in the decision-making process (Wolfson et al., 1996). Bayesian statistical inference provides an alternative way to analyse data that remedies many of the problems inherent in standard hypothesis testing and, most important, allows the incorporation of uncertainty. Therefore, it is increasingly being used in the treatment of ecological and environmental problems. Marin et al. (2003) introduces some of the specific tools that are needed for the Bayesian approach and illustrate the use of these tools through case studies.

3.4. Species-oriented conservation

3.4.1. Red lists of threatened species

A concern for species-oriented conservation stems from the evidence that in recent times species extinction rates have significantly increased due to human activities. The question of assessing extinction rates of species is an area where there has been considerable polemic but little progress. Although we have made considerable advances in studying the actual processes of extinction in individual species, our only factual knowledge of species losses derives from country assessments and global evaluations and syntheses of these, notably by the IUCN Red List Programme.

The *2000 IUCN Red List of Threatened Species* (IUCN, 2000a) which aims to present a snapshot of the state of the world's plant diversity at the end of the second millennium lists 7022 species that are threatened to some degree with eventual extinction The fact is that we do not even know with a reasonable degree of certainty just how many species there are in all groups of organisms and it is estimated that we have only described scientifically about 1.75 million (ca. 13%) of the 13–14 million species that are estimated to exist today (Heywood, 1995). Even the number of plant species has been subjected to substantial revision in recent years, with current estimates being around 400,000 species as opposed to the previously commonly cited figure of 250,000. Most projections of future extinction rates rely essentially on species–area relations combined with estimates of habitat loss ('area') due to deforestation or other processes (May et al., 1995), derived from MacArthur and Wilson's theory of island biogeography, although a growing number of biologists do not consider this approach to be appropriate or relevant.

Given the difficulties of approaching species conservation at a global scale most efforts are being made at a national or regional level. The existence of large numbers of species presently believed to be under some degree of threat make necessary the use of a system of classification that help categorise species according to the risk of extinction they are facing, and thereby, prioritise efforts and resources in the most endangered species. The IUCN Red List categories have been widely used throughout much of the world, with notable exceptions such as the USA and New Zealand. The original IUCN categories, which were in place for 30 years, were revised in 1994 in line with the progress of conservation biology, having evolved from mostly subjective and qualitative criteria towards a system based largely on objective quantitative parameters. Therefore, as both knowledge about the existing global species and threat classification systems have progressed, some countries have developed improved versions of national Red Lists and Red Books, highlighting the priorities for species-oriented conservation. An example of this sequential improvement can be seen in Moreno et al. (2003). On the other hand, the introduction of the revised IUCN criteria has been a mixed blessing, and practical difficulties in applying them to plants has led to some major Red Lists continuing to use the old criteria. The consequence has been to hinder progress in assembling global sets of data.

3.4.2. *In situ conservation of target species*

Considerable confusion surrounds the notion of in situ conservation of individual or target species. Simple presence of a species in a Protected Area is no guarantee of its conservation: we have sufficient evidence to suggest that the dynamics of ecological change may lead to considerable changes in the plant and animal composition over short periods of time, so that unless there is active intervention in the ecosystem or specific management of individual populations, the continued survival of the target species may not be ensured. At a minimum, monitoring of the populations is needed to follow what is happening to them.

Surprisingly, the Global Strategy for Plant Conservation, in its explanation of its Target 7: '60 per cent of the world's threatened species conserved *in situ*', appears to endorse the 'benign neglect' approach when it states that 'Conserved *in situ* is here understood to mean that populations of the species are effectively maintained in at least one protected area or through other *in situ* management measures'. Unless 'effective maintenance' means some form of management or intervention, successful conservation in situ is unlikely to be achieved in many cases, but since it then goes on to say that 'In some countries this figure has already been met, but it would require additional efforts in many countries', it seems clear that active intervention is not what is meant. The fact is that, with the exception of a number of species recovery plans for endangered species in some countries, very few serious attempts have been made to establish and maintain in situ conservation areas for target species such as crop relatives, fodder species, medicinal plants. For this reason, a GEF project on 'Design, Testing and Evaluation of Best Practices for in situ Conservation of Economically Important Wild Species' was proposed and following recent approval is about to begin (2003).

3.4.3. *Biological information and diagnosis of the factors threatening populations*

Once the problem is reduced to the in situ conservation of a particular set of species, the basic objective is maintaining the viability of their populations. Initial efforts towards this task have been plagued with significant failures and mistakes due to incorrect diagnosis of the factors threatening the populations, frequently based in fragmentary and subjective perceptions of reality. As recently as 1994, Pavlik (1994) was writing that 'No plant taxa have, in fact, been recovered and consequently regarded as conserved in situ, despite numerous attempts to protect, create or enhance populations'. Recovery projects have failed to achieve their aims for a variety of reasons, the lack of detailed demographic data being the 'greatest and most common deficiency in species recovery projects' (Pavlik, 1994).

Thus, when approaching the task of conservation of populations of a threatened plant species, a series of basic questions need to be addressed. We need to know: is the population under study actually declining? Which are the factors that determine the viability of the population? Which life stage is most critical for the viability of the population? Is legal protection of the habitat alone a sufficient measure to maintain population viability or is a more active intervention needed? Which management strategy offers the greatest chances for facilitating the survival of the population? What may be the consequences on the population of particular human-induced environmental changes on the habitat?

To overcome these initial failures and provide adequate answers to these questions, a systematic collection of baseline data on the natural history of the species is needed. This will enable the assessment of the biological status of the species, the identification of the life history stages most critical for population growth, and the determination of the main biological causes of demographic variation at these stages (Schemske et al., 1994).

Censuses and demographic monitoring of the populations concerned over a series of years provide basic information about population demographic trends. These trends are determined by the vital parameters (survival, growth and reproduction) that in turn are conditioned by genetic and environmental factors (see Iriondo et al., 2003).

Genetic processes can lead to changes in the number or frequency of alleles in populations and in levels of heterozygosity. Both variables have shown to be related to vital rates in some plant species. In theory, reduced heterozygosity can result in decreased population growth due to inbreeding depression (Charlesworth and Charlesworth, 1987). On the other hand, allele richness may contribute to population growth through its effect on evolutionary potential, or the ability of a species to respond to changes in its selective environment. This explains the abundant existing scientific literature aimed at the study of the genetic diversity and structure of threatened plant species.

Genetic studies have sometimes been criticised for being too expensive and providing few practical results for conservation and management. An additional concern is the fact that most studies are based on allozyme or DNA markers (RAPD, SSR, AFLP), which are considered to be neutral and not affected by selective pressures. The correlation between these traits and the adaptive traits of a species, which are subjected to selective pressures and directly contribute to the fitness of individuals and the viability of the populations, is not straightforward. Anyhow, in addition to information about population genetic diversity, molecular markers provide relevant information for identifying units of conservation and about the genetic processes that take place in the populations such as patterns of genetic flux, generation of genetic neighbourhoods and incidence of genetic drift.

The environmental factors shown to influence vital rates are diverse (e.g. climate variability, biotic interactions, intraspecific density) although their direct effects on the persistence of the populations have received comparatively little attention. Among these factors the spatial dimension is one that has been historically neglected and that is now prompting the attention of conservation biologists. For instance, the presence of spatial structure in the genetic diversity of a population indicates the existence of restricted gene flow. This situation may affect reproductive success and fitness of the individuals, and has direct implications on the way the population should be managed and conserved. Escudero et al. (2003) show the wide array of spatial analysis techniques that are now applicable to genetic studies.

Overall, there is an increasing need to better integrate genetic and ecological studies with the study of the processes that condition the viability of populations. Thus, the relative roles of population genetic and ecological characteristics in causing extinction need to be addressed, as well as the effect of population size on the relative contributions of genetic and ecological attributes to population processes. Novel approaches linking genetics and demography are being taken and Oostermeijer et al. (2003) provide some good examples in this direction.

Raw data from censuses of the number of individuals of a population through time provide a simple estimation of the demographic trend of a population. However, there is only one approach to evaluating biological status that provides both an assessment of population growth and identification of the life history stages that most affect population growth. This involves the use of stage-structured population models and demographic monitoring that follows the fates of individual plants in a population through time. As demographic parameters in plants are determined more by the life history stage or size of an individual than by its age (Caswell, 2001), the Lefkovitch stage-based model (Lefkovitch, 1965) is the most appropriate one for studying plant populations (see Marrero et al., 2003; Moreno et al., 2003).

Population viability analysis (PVA) is a procedure that allows the simulation through models of the extinction processes that act upon a population. In essence, it predicts the probability of persistence of a population over a specified amount of time (e.g. Shaffer, 1981; Menges, 1990; Ginzburg et al., 1990). PVA is a form of risk assessment that can account for factors such as density dependence, effective population size, loss of heterozygosity, and various stochastic events, such as likelihood of fixation of mutant alleles, environmental and demographic effects or catastrophes. Sensitivity is a measure of how population growth rate responds to small changes in a demographic parameter. The use of sensitivity and elasticity (a measure of proportional sensitivity) analyses on demographic models provides useful information for the determination of the biological factors that are most likely to be critically affecting the trends of a population. Moreover, PVA allows the study of the relevance of changes in natural conditions and to estimate extinction probability under different circumstances. The latter as well as the estimation of the decline of the populations over a period of time can be directly applied in determining the degree of threat of a taxon under the most recent IUCN Red List criteria (IUCN, 2000a).

PVA techniques are still under development and undergoing constant changes. The use of spatially explicit models, the integration of GIS environmental data, and the application of metapopulation approaches incorporating migration between populations are some of the recent improvements in these techniques. The consideration of metapopulation dynamics may be important in assessing the biological status of the species and in establishing the number of populations required for species persistence (Schemske et al., 1994). In this sense, the 'minimum available suitable habitat' (Hanski et al., 1996) is a relatively new concept with great potential in restoration. It has been successfully used by Quintana-Ascencio and Menges (1996) in Florida or by Valverde and Silvertown (1997) with *Primula vulgaris* in UK. Further challenges lie in the appropriate modelling of individual variation due to genetic or microenvironmental diversity and in finding ways to circumvent the difficulties associated with applying these techniques to certain plant species, such as those showing cryptic clonal growth (e.g. bulbils), long-distance seed dispersal, and extreme longevity in some size or stage classes (Pavlik, 1994).

Another promising tool for assessing and contrasting alternative hypotheses and diagnosing the relative importance of the factors that may be affecting population viability is the application of structural equation modelling. This technique, widely used in economy and psychology, is recently becoming applied in ecological studies and holds a considerable potential in conservation (see Iriondo et al., 2003).

3.4.4. Recovery actions

As previously stated, the lack of basic biological data has been responsible for most failures in species recovery projects. Thus, biological information must first be gathered, the most declining populations in an endangered species identified, and the factors that most affect this decline assessed. A set of actions can then be established to minimise these factors, to reverse the declining trends and to fulfil the objectives of the recovery plan. These recovery objectives should be established according to the population dynamics of the species.

The most appropriate actions for recovering declining populations can be determined by experiments that test the effect of different management regimes derived from competing hypotheses about critical factors that limit population growth. However, quite often the situation of the natural populations will be so critical that experimentation may not be feasible due to legal and ethical constraints. In these cases, the comparison of demographic characteristics between viable and non-viable populations may provide relevant information as well as well as the use of PVAs simulating alternative management regimes. Although PVA techniques are still in the developmental phase their utility in formulating and evaluating possible restoration strategies for many species cannot be questioned. Their application is already providing relevant information for species conservation and management both at local (Marrero et al., 2003) and national scales (Moreno et al., 2003).

Some of the limits to population growth will be readily overcome by restoring natural processes or key species to the habitat. Others may require that population size be increased artificially, through transplanting or intensive care of natural seedlings and juvenile plants (Pavlik, 1994), or that new populations be established in order to facilitate metapopulation dynamics and to decrease the risk of extinction of the species as a whole. Thus, reinforcement, introduction and reintroduction actions may be required to fulfil the objectives of the recovery plan.

The design and management of new populations is perhaps one of the most challenging tasks associated with conservation of endangered plant species. In contrast to monitoring natural populations, creating new populations begins with experiments to determine the most important factors limiting the growth of the founding population. This is followed by the prescription of appropriate management, and trend analysis of the newly created population in subsequent years (Pavlik, 1994). Created populations not only reduce the risk of extinction, they also amplify our understanding of the target species and increase our ability to successfully manipulate its natural populations.

There are, of course, space and time dimensions in the recovery process which require that natural populations be within appropriate, protected habitats and able to maintain themselves over long periods. Successfully restored populations or communities will likely be limited to legally protected preserves, national and state parks and forests, or natural areas owned by private conservation organisations (e.g. Marrero et al., 2003). Meanwhile, remaining unprotected large blocks of natural vegetation are constantly shrinking because of pressure from ever expanding human populations, agriculture, industrialisation, and other forms of development. Hence, restoration ecology and the disciplines it embraces are likely to assume increased importance, not only as regards restored populations, but also for maintaining extant populations living in an increasingly fragmented world.

Reintroductions often involve translocating genotypes across geographic ranges. Some have criticised this practice, arguing that organisms tend to be highly locally adapted, and that such movements introduce 'incorrect' genotypes where they do not belong. The situation is even more critical when 'foreign' genotypes are used to reinforce natural populations because of the possible appearance of outbreeding depression. This is an intriguing and controversial problem in which the need to maximise genetic diversity is balanced against maintenance of coadapted gene complexes (Avise, 1992; Ellstrand, 1992), and will probably vary tremendously from species to species.

In spite of all the earlier-mentioned biological considerations that need to be taken into account when restoring endangered species, Bowles and Whelan (1994) remind us, that, however, 'biology, is often the least of the problems', since restorations necessarily occur within a societal context where economic and social issues may be more significant factors in determining progress and eventual success.

3.4.5. Ex situ conservation

Ex situ conservation techniques are critical components of an integrated global conservation programme (Ashton, 1987; Conway, 1988). They are especially well suited for plant conservation because particular stages of a plant life cycle (seeds, spores, pollen) are naturally adapted to maintain its viability over long periods of time. They complement in situ conservation by medium- or long-term storage of selected samples of populations' genetic diversity, allowing a better knowledge of the anatomical, physiological and biochemical features of the stored material, and providing material for its use in education, crop breeding programmes and reintroduction plans.

Ex situ conservation methods imply collecting selected or representative samples of the genetic diversity of each species and storing them outside the natural environmental conditions in which the species have evolved. The storage of germplasm of endangered plants is carried out by botanic gardens and dedicated germplasm facilities (seedbanks, field gene banks, tissue and cell culture, cryopreservation; Laliberté, 1997). The storage of seeds is one of the most valid and widespread methods of ex situ conservation at present due to its simplicity and economy in terms of technology, infrastructure, manpower and operating costs. Thus, it is feasible to keep a great number of seeds of many different plant species over long periods of time in a reduced space and with a minimum risk of genetic damage (Iriondo and Pérez, 1999). Protocols for ecogeographical surveying and sampling strategies and the technology of seed storage are now quite well established and were primarily developed for crop genetic resources (Hawkes et al., 2000; but see Gómez-Campo, 2000). They have yet to be adapted and applied to ex situ collections in many botanic gardens (Heywood, 2002). In fact, a distinction has to be made between the ex situ conservation of samples of landraces or cultivars of crops in agricultural genebanks and the ex situ conservation of wild species. The main focus in the former case is on intensive sampling of infraspecific diversity, in particular of alleles carrying useful traits such as disease resistance, aridity tolerance, etc. The latter aims at a broad coverage of species rather than intensity of sampling, and focuses especially on rare or endangered species (Heywood, 1999). As Debouck (2000) points out, the enormous amount of variation and ecologically highly specific requirements of wild species (including crop relatives) often makes their ex situ conservation difficult.

Much greater attention needs to be paid to issues of sampling—what to sample, how much, for what purpose, etc.—and to the efficiency and quality of seed collecting and storage so as to ensure that *ex situ* conservation of wild species makes a significant contribution to conserving the genetic diversity occurring in wild species. As restoration projects increasingly demand the availability of genetic resources of particular populations of species, GIS methodologies have a great potential in helping to decide which populations should be sampled and stored (see Draper et al., 2003).

3.5. The ecosystem approach

It is axiomatic of the conservation movement today that biodiversity is best conserved generally by preserving habitats. Certainly this is an attractive approach in that a single listing is sufficient per habitat no matter how many species it contains, the assumption being that all the species will be preserved (although as discussed earlier, this is not necessarily so). Because of the enormity (and, some would say futility) of the task of species conservation, and the limited funds available for conservation, many conservationists have criticised a species-based approach. They regard this as essentially a crisis management approach, and favour an ecosystem-based or landscape approach. Even if we were to focus only on those that are threatened, the numbers are still overwhelming. We have neither the financial nor technical resources to allow us to address the conservation of all these species, or even of the globally or nationally threatened species.

This anti-species backlash has manifested itself recently in the Convention on Biological Diversity through its advocacy of 'the ecosystem approach'—a strategy for integrated management of land, water and living resources that promotes conservation and sustainable use of these resources in an equitable way—although this embraces both species and ecosystems as they are mutually interdependent. As Soulé and Mills (1992), observe, a pure ecosystem approach is as absurd as a pure species approach.

Developing sound and practical strategies to restore degraded ecosystems is not a straightforward task. Many issues and processes must be individually and collectively understood for effective action to take place. It can be difficult to determine which areas to restore, what species and/or vegetation communities to target in restoration programmes, and what threatening processes need to be mitigated. According to De la Cruz-Rot (2001), focusing on the community level can help fill the gap between species and ecosystem approaches to plant conservation. Plant communities are in fact basic components of the landscape and their extent and arrangement has consequences both for species survival and for ecosystem processes. The assessment of the structure of plant landscape (e.g. the number, size and location of plant communities in space) can link some ecological processes, such as primary production, succession and matter flow to essential species features like carrying capacity or metapopulation structure and suitable management options can be derived.

3.6. Focal species approach in landscape restoration

Lambeck (1997, 1999) proposed the focal-species approach in an effort to provide a more scientific basis for landscape restoration. Under Lambeck's (1997) approach, species are grouped according to the processes likely to threaten their persistence, and the species perceived to be most sensitive to each threat are selected as a suite of focal species. The idea is to manage a landscape for this suite of focal species, each of which is thought to be sensitive to a particular threatening process. The focal-species approach is now being applied in Australia and in other parts of the world (Noss, 1999; Foreman et al., 2000). Nevertheless, despite the merits of this approach and the enthusiasm for its implementation, some authors believe that its theoretical and practical underpinnings are not well established. The fundamental assumption of this approach is that if resource management or restoration efforts are targeted at a group of species, the needs of the other taxa will also be met (Lambeck, 1999). However, some authors have raised concerns about the conceptual, theoretical, and practical basis of taxon-based surrogate schemes (e.g. Andelman and Fagan, 2000; Lindenmayer et al., 2000, 2002). From this debate it is clear that a pressing challenge for present and future plant conservation is to find sound scientific approaches to develop strategies for the conservation and management of plant communities and ecosystems. The problems associated with the implementation of any approach caused by limited or inadequate data are real, and society should be made aware of the importance of allocating resources to gathering baseline information on species and ecosystems. In the absence of appropriate data, the best alternative currently available for management is the use of general principles of landscape ecology, although these confer no ability to specify the requirements for preventing further loss of species (Lambeck, 1997).

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