ECOSYSTEM MANAGEMENT

QUESTIONS FOR SCIENCE AND SOCIETY

Edited by
Edward Maltby • Martin Holdgate • Mike Acreman • Antony Weir

Ecosystem Management:

Questions for science and society

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Synthesis of the first Sibthorp Seminar: *'Advances in ecological science as a basis for conservation and ecosystem management in the third millennium'*, held at the Royal Holloway Institute for Environmental Research (RHIER), Royal Holloway University of London, 21-22 June 1996

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CONTENTS

List of Boxes

List of Figures

List of Tables

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FOREWORD

Miss Mary Sibthorp, who died in 1991, left money to IUCN - the World Conservation Union, on the understanding that IUCN would establish a charitable trust, with the overall aim of promoting for the public benefit the conservation of nature and natural resources. Through Miss Sibthorp's will and subsequent discussion by the appointed Trustees, the Trust's main areas of activity have been identified. The Trust will:

- promote the objectives of IUCN, especially in the United Kingdom
- promote the implementation of the World Conservation Strategy (IUCN 1980) and Caring for the Earth (IUCN/UNEP/WWF 1991)
- promote the study of key environmental issues through the commissioning of workshops, study groups and seminars and the production of papers.

Miss Sibthorp worked for many years for the David Davies Memorial Institute, and in this capacity played an active part in convening expert study groups and publishing their findings. In her later years, she became increasingly interested in environmental issues. She therefore hoped that the Trust established by IUCN would particularly support critical analyses of the interrelationships between humankind and the natural environment, and contribute to the wise and sustainable use of the environment in different regions of the world.

Miss Sibthorp had a keen and critical mind and believed in the importance of questioning "established wisdom". The Trustees have from the outset emphasised their wish for a challenging approach to the issues they examined.

The first Sibthorp Seminar was designed to look critically at the findings of recent ecological research and to consider how far it made a radical re-design of conventional conservation practices necessary. The First Sibthorp Paper reports the outcome of this seminar, held at the Royal Holloway Institute for Environmental Research on 21 and 22 June 1996. The meeting was organised by Professor Edward Maltby and supported by the Sibthorp Trust. The results formed the basic document for a workshop held at the First World Conservation Congress in Montreal, Canada, on 17 October 1996. They are now presented in a more complete form as a contribution to the continuing debate over the management of our environment.

As Chairman of the Trustees I am delighted that our work has begun in this way and I hope that both the Sibthorp Synthesis and Sibthorp Papers will prove a stimulus to new thinking in the world conservation community.

Sir Martin Holdgate
Chairman of the Trustees of the Sibthorp Trust,
Cambridge, July 1997

MARY SIBTHORP

(1905 - 1991)

Mary Sibthorp (1905 - 1991) was a remarkable person. Largely self-educated, her incisive intellect allowed her to hold more than her own in any discussion. In her twenties, she came to the notice of Lord Davies of Llandinam, a fighter for peace and a better world order. She worked with and for him until his death in 1944, and continued to work for his New Commonwealth Society. When the David Davies Memorial Institute of International Studies was founded in 1951, she became its Assistant Secretary and in due course its Director, a post she held until her retirement in 1980. Her energy and inspiring vision gave her great influence on the work of the Institute. Her outstanding characteristics were iconoclasm, dislike of utopianism and suspicion of received wisdom. She was greatly interested in the wise use of natural resources and in the international problems arising therefrom.

In her generous will she left instructions to set up the Sibthorp Trust which is sponsoring this seminar. The more critical our discussions are of unsound views and the more forward looking they are, the better they will fulfil her purpose.

Professor Sir Hermann Bondi FRS,
Member of the Sibthorp Trust,
Cambridge, August 1996

PREFACE

Traditional ecological concepts, which have long underpinned conservation, are increasingly being questioned. The establishment of the IUCN Commission on Ecosystem Management (CEM) in 1994 at Buenos Aires is a reflection of the perceived need for new scientific perspectives on fundamental issues of conservation and management of natural resources. It's mission was crystallised as a result of an extensive consultation process resulting in the first Strategic Plan 1996-1997.

The Commission on Ecosystem Management (CEM) will endeavour to provide expert guidance on integrated approaches to the management of natural and modified ecosystems, to further the IUCN mission. CEM will link science, society and ecosystem management through three interrelated approaches:

- Improving understanding of the management of ecosystems, by bringing together the latest thinking in ecosystem science, distilling it and communicating it in an accessible form;
- Convincing decision makers of the relevance of the ecosystem approach to conservation, by highlighting priority issues, new developments, threats and opportunities and practising advocacy;
- Facilitating the implementation of integrated ecosystem management principles, by assisting "stakeholders", through identifying crucial issues and developing solutions to ecosystem management problems.

The first Sibthorp Seminar was held to:

1. challenge traditional ecological thinking and explore the relevance and application of recent scientific advances;
2. evaluate current understanding of the ecological fundamentals underpinning ecosystem management;
3. facilitate structured debate of key issues;
4. assist in the future development by IUCN of a Handbook on Procedures for Integrated Management of Ecosystems for Conservation and Resource Utilisation;
5. provide a key input into the development of the programme of the IUCN Commission on Ecosystem Management.

The papers initiating discussion at the Sibthorp Seminar are reproduced in this volume, which also includes additional material presented at the World Conservation Congress in Montreal, where draft copies of the first chapter were disseminated widely for critical review. They represent the views of the individual authors and have been subject to minimal modifications by the editors. The salient points of each paper, and the reports of the working group sessions, have been distilled in a set of 10 principles for ecosystem management. This distillation forms the basis for the first chapter. It is not the first time such principles have been elaborated (see for example Jensen and Bourgeron 1994) nor are any of the ideas separately unique but collectively they may provide some contribution to new thinking on the need to better link science to conservation. The text is not intended to be a comprehensive academic or practical treatise on the subject, but a starting point for further debate.

The intended audience includes scientific and management professionals concerned with the application of the ecosystem approach in biodiversity and natural resource issues; decision - makers in government and non-government organisations involved in conservation strategies and policy especially those concerned with interpretation of the ecosystem approach in the context of the Convention on Biological Diversity; and students interested in some of the questions facing those trying to develop innovative approaches to deliver more effective conservation. Many of the examples cited are from the anglophone world. It would be useful now to build up additional cases from a wider world perspective and continue the very good start made by the analysis of the institutional, technical and operational profiles of twenty-four different field projects carried out by IUCN for the World Bank (Pirot and Meynell 1998).

Professor Edward Maltby
Director of the Royal Holloway Institute for Environmental Research
Chairman of the IUCN Commission on Ecosystem Management

1. PRINCIPLES OF ECOSYSTEM MANAGEMENT

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1.1 Ecology and ecosystems

Ecology was the 'new' science in the 1950's and 1960's that raised awareness of the linkages between organisms and their environment and which gave impetus to increasing environmental concerns, and provided intellectual and scientific rigour to underpin the growing conservation ethic. Ecological concepts now permeate more or less completely the approaches of conservation organisations and those concerned more broadly about the sustainable use of the earth's natural resources. However the formulation and application of these concepts have not necessarily progressed in the light of new knowledge. It has been necessary also to take a flexible and adaptive approach in the use of ecological science in conservation because of the changing needs and priorities of society.

It is some sixty years ago that Tansley formalised in the term 'ecosystem', an idea which had been around since the beginning of the nineteenth century, and probably very much earlier (see Box 1). Indeed, so called organic theories in which the Earth was conceived as a functioning organism can be traced back to Greek times. Sir James Lovelock (1979) has re-popularised such thinking in his 'Gaia' concept, in which we can think of the entire planet as one huge ecosystem.

The ecosystem model has become a fundamental element of the way in which scientists view the natural world. It formalised the recognition of functional links between species or groups of species such as trophic hierarchy or food webs, and between organisms and their environment such as habitat requirements and climatic range. Flows of energy, (usually in the form of separate meals), nutrients and materials such as water and sediment, maintain the structure, stability and biological diversity of ecosystems. Yet almost immediately after Tansley's introduction, the concept became the subject of divergent interpretations and definitions. A basic division has developed between those recognising the 'concrete' existence of ecosystems and using the term to describe discrete ecological units such as a forest, grassland or bog, from those restricting the use of the term to conceptual models of reality.

There is the implication that in the case of 'concrete' ecosystems, we have determined the complexity of interactions among the various elements of the ecosystem. In fact, even at best we only have estimates of the major flows of energy and matter, and these for limited periods of observation. Research to determine more precisely the dynamics and functioning of ecosystems is expensive and even if limited to key elements, will be restricted to a few case studies. Nevertheless the recognition of tangible units of land or water (or combinations of the two) as functioning ecosystems provides a powerful basis for interactive management. The alternative approach maintains a conceptual emphasis. It is understood that ecosystems are artificial simplifications of the real world and that their analysis depends fundamentally on how boundaries are drawn. Critics often cite the artificiality of boundaries and the practical difficulties of describing fully the energetics and dynamics as limitations to their practical designation in the field. Nevertheless, the concept has proved an invaluable aid to advance scientific thinking, research and management by virtue of at least a few key features identified by Odum (1972), and outlined below.

Box 1 What is an Ecosystem?

Humboldt in 1807 wrote on plant geography that "*In the great chain of causes and effects nothing and no activity should be regarded in isolation*".

Tansley (1935) formally defined an ecosystem as "*a unit of vegetation which ...includes not only the plants of which it is composed but the animals habitually associated with them, and also all the physical and chemical components of the immediate environment, or habitat which together form a recognisable self-contained entity.*" In 1946 he added that all parts of such an ecosystem may be regarded as interacting.

Fosberg (1963) describes an ecosystem as a "*functioning, interacting system composed of one or more living organisms and their effective environment, both physical and biological. The description of an ecosystem may include its spatial relations, inventories of its physical features, its habitats and ecological niches, its organisms and its basic reserves of energy and matter, the nature of its income of matter, and energy and the behaviour or trend of its entropy levels*".

Polunin and Worthington (1990) have proposed that the term *ecocomplex* should be used for larger and less-integrated systems, such as lakes, rivers, islands or forests, which often contain several ecosystems as defined by Tansley.

In this paper, *ecosystem management* is treated as a human activity that affects both ecosystems as traditionally defined as well as larger spatial units such as ecocomplexes.

1.1.1 Ecosystem characteristics

First, the model is monistic. It brings together the plant, animal and human worlds and environment in such a way that their interactions can be analysed within a single framework. The emphasis is on the functioning of an entire or holistic system, rather than separated components. Second, it is structured. The hierarchical arrangement of ecosystem elements facilitates understanding of complex interactions, and assists in the organisation of data collection. Third, its emphasis is 'functioning'. This follows from knowledge of how the various elements are connected, and enables us to determine the factors affecting different parts of the system. In particular it is possible to trace and predict the movement of pollutants and/or other contaminants and to forecast the likely results of changes to particular elements in the system. Fourth, it is a type of general system. This enables modelling within the rules and concept of general systems theory with the possibility of predicting the effects of change in one or more of the variables in the ecosystem.

Application of the ecosystem model has improved considerably our understanding and application of environmental management in areas as diverse as fire management and pesticide control. This has resulted primarily from investigation within relatively small and well-defined ecological systems such as a discrete forest, heathland or lake. The now classical studies at Clear Lake, California on the fate and impact of DDT confirmed the essential need of treating physical and biological components together (Hunt and Bischoff 1960, and also described in Carson 1962). The applications of DDD to eradicate a small gnat (*Chaoborus astictopus*) were quickly undetectable in the lake water but concentrations were being amplified along the food chain leading to reproductive failure in the population of Western Grebes. Knowledge of ecosystem structure and functioning would have enabled more accurate prediction of the wider consequences of pesticide treatments. It has certainly underpinned the development of biological control and systemic application methods for the pest management. The overriding value of the ecosystem model is its flexibility in application. This has been recognised fully in conservation and the term has become increasingly part of the language of important international agreements such as the Conservation of Biological Diversity. Tansley, however, would possibly find it difficult to recognise immediately the new interpretation of his original term as used in the 'ecosystem approach'. The new application of the concept has elicited considerable debate in recent years and represents a significant departure from its more restrictive scientific origins.

1.2 Developing ideas on ecosystem management

People have managed ecosystems ever since they first altered the biotic composition of their immediate environment in order to derive some form of benefit. Historical accounts of the origin of ideas on ecosystem management are well documented (Malone 1996, Grumbine 1994, Vogt *et al.*. 1997 and Czech and Krausman 1997).

Leopold (1949) was a visionary in his holistic viewpoint of ecosystems and the management of them, and stated that people should take care of the land as a 'whole organism' and try to keep 'all the cogs and wheels' in good working order. This is perhaps the first attempt to elaborate concepts of ecosystem management, as we may interpret the term today, where the thinking was to sustain diversity of ecosystems while maintaining their productivity to meet human needs. However, early attempts to combine resource management and ecological thinking at the landscape level were not successful (Grumbine 1994).

In the 1960's ecology increasingly emphasised the study of ecosystems and at the same time increased environmental awareness steered the discipline towards conservation and resource management. Walter (1960) emphasised the need for a holistic treatment of ecological factors, including the anthropogenic one, in landscape ecology and management. The inter-disciplinary study of ecosystems was strengthened further and systematised during the 1960's and 1970's through the implementation of the International Biological Programme (IBP) and the influential writings of Odum (1969), Van Dyne (1969), Watt (1968).

Despite the scientific interest in the ecosystem concept, the approach had yet to be implemented as a policy tool. Caldwell (1970) advocated using ecosystems as the basis for public land policy in the US. Attention on ecosystem management was refocused by Frank and John Craighead, with their grizzly bear (*Ursus arctos*) population study in Yellowstone National Park (Craighead 1979), and by the work of Newmark (1985) who compared the legal and biotic boundaries of some parks and reserves in Western North America. Both demonstrated that geographic and political boundaries set to protect an ecosystem were often not sufficient to protect all components of that ecosystem. The ecosystem management concept really began to gain wide acceptance in the United States in the early 1980's, with support from scientists, land managers and resource policy analysts who were looking for ways to better address declining ecological conditions.

In the late 1980's and early 1990's, ecological research began to focus on studying systems over greater spatial and temporal dimensions, and the question of ecosystem sustainability became a strong focal point (Lubchenco *et al.*. 1991). There was an explosion of interest in ecosystem management (see Czech and Krausman 1997 for review), and the number of projects based on deriving principles of ecosystem management, trying to balance and integrate ecological, economic, social and political goals has proliferated in recent years.

1.2.1 *What is an ecosystem approach?*

It is now the case that the term 'ecosystem approach' has been used to describe a particular form of environmental management which is far removed from the original meaning of the term *ecosystem* and indeed has generated much confusion over its exact interpretation. The 'ecosystem approach' has been a particular feature since 1993 of policy numerous agencies of the US federal government (See also Box 2).

Box 2. Why is an Ecosystem Approach Needed?

Hydrological, biological, chemical and physical processes, vital for the maintenance of ecological systems, are being disturbed throughout the world. The speed of change is generally accelerating, resulting in degradation of ecosystem structure and compromising the options for sustainability. One immediate consequence may be loss of biodiversity, but overall there is a progressive reduction in important environmental services.

Human societies and their governments are locked into a sectoral approach to conservation and natural resource management. Traditional sectoral approaches are an inadequate basis for the management of the ecosystems which support species, communities and resulting biodiversity, together with environmental services and natural products. This results from the lack of understanding or full appreciation of the significance of ecosystem functioning. There is a failure also to recognise the importance of different scales, both in time and space, for the management of ecosystems, for example, the need to manage the catchment as well as a site. The lack of integrated and long term planning essential to maintain the integrity of ecological systems can be redressed only by new systems and approaches to ecosystem management.

An Interagency Ecosystem Management Task Force (1995), defined the ecosystem approach as 'a method for sustaining or restoring natural systems and their functions and values. It is goal driven, and it is based on a collaboratively developed vision of devised future condition that integrates ecological, economic, and social factors. It is applied within a geographic framework defined primarily by ecological boundaries'.

The Task Force defined the goal as

> "To restore and sustain the health, productivity and biological diversity of ecosystems and the overall quality of life through a natural resource management approach that is fully integrated with social and economic goals...."

In the case of the Convention on Biological Diversity, the approach would aim to deliver the convention three key's objectives: conservation of biological diversity, sustainable use of its components, and fair and equitable sharing of benefits arising from genetic resources.

The essential ingredient of the ecosystem approach is that it is holistic. Beyond the original scientific application, it recognises explicitly the interrelationships between natural systems and economic, social, political and cultural systems. It provides a structural rationale for various societal groups, public and private managers to husband natural resources.

The approach involves *inter alia.*

- Ensuring all relevant and identifiable ecological and economic (together with social, cultural and political) consequences are considered in area management.
- Improving co-ordination among responsible agencies and people.
- Forming partnerships between governments at different levels, social groups, landowners and other stakeholders.
- Improving communications with the general public and individual stakeholders.
- Using the best science.
- Improving information and data management.
- Being able to adjust management as new information becomes available.

(Adapted from report of the Interagency Ecosystem Management Task Force 1995).

We are still far from successfully implementing such an approach except in a limited number of special cases. There are needs in particular for fundamental changes in approaches to policy instruments, such as incentives, legal systems and conflict resolution among competing sectoral interests.

1.2.2 Direct versus indirect ecosystem management

Direct ecosystem management involves the *direct* interaction with physical, chemical or biological process, species or populations. It is normally applied at a relatively small scale such as the biomanipulation of small lakes by the introduction or removal of herbivorous/carnivorous fish populations or amelioration of acidity in aquatic ecosystems by application of lime. Ecosystem management, however, is more generally effected by indirect means, especially at larger scales, such as mountain systems, large lakes, catchments and coastal zones; usually involving intervention (reduction, maintenance or enhancement) in human activities (physical, social, cultural and economic). This aspect of control might be better referred to as *ecosystem-based* management because the emphasis is not on ecosystem processes *per se* but on the human impacts on those processes and consequences for ecosystem structure and function (Box 3).

The development of effective approaches to ecosystem management involves close collaboration among many specialists including ecologists, hydrologists, economists and sociologists. Its implementation means working with numerous and diverse stakeholders, including local communities, local and regional governments, private organisations and protected area agencies, and seeking common objectives for land use planning and resource allocation.

Box 3. What is Ecosystem Management?

Ecosystem management is the manipulation of the physical, chemical and biological processes which link organisms with their abiotic environment and the regulation of human actions to produce a desired ecosystem state. Ecosystem management can include:

- adjusting the chemical conditions by controlling pollution or altering the input of nutrients and contaminants to atmosphere, waters, soil or more directly to vegetation;
- regulating physical parameters, for example by making controlled releases of water from a dam or entry of saltwater into coastal impoundments;
- altering biological interrelationships, for example by controlling grazing and predation, or preventing the colonisation of grassland or heathland by bushes and trees or intervening in vegetation development or dynamics by burning or cutting;
- controlling human use of biological productivity, for example by limiting the use of fertilisers and pesticides, or regulating fish net mesh sizes;
- intervening in cultural, social and economic processes, for example by compensating farmers for reducing the intensity of their operations in the interests of conservation.

Good ecosystem management will maintain the integrity of ecosystem functioning to avoid rapid undesirable ecological or environmental change. It will aim to maintain and, where possible, enhance biodiversity and environmental services such as water quality and food chain support. The ecosystem approach provides a wider basis for management, as the size of the management unit covered can be adjusted for different temporal and spatial scales, according to the nature of the problem and to the scales at which the ecosystem processes are operating. It also provides a more practical structure for co-operative efforts among governments, communities and private interests, to enable a management approach which is integrated, interdisciplinary, participatory and sustainable.

1.2.3 The need for ecosystem management

Calculations indicate that extinction rates over the next 100 years will be at least four orders of magnitude faster than the background rate in the fossil record (Lawton and May 1995; see also Pimm *et al.* 1995). This rate is comparable with that during previous 'extinction spasms' like that during the late Cretaceous and early Tertiary period, celebrated for the disappearance of the large dinosaurs. The current spasm of mass extinctions is the first to be caused by a single pan-dominant species rather than by some natural environmental change or extreme catastrophe. It results largely from human alteration of ecosystems and pre-emption of their biological production. Human impacts frequently result in more rapid changes than can be accommodated without significant alteration in ecosystem structure and biodiversity.

Nevertheless, we should not apply limited conservation resources to the maintenance of species alone, particularly where populations exist at the edge of their range or where environmental change means that loss may be inevitable (see Lawton this volume). Extinction of species has been a normal phenomenon of Earth history - it is the accelerated rate of loss due to human activities which is the most important trend that needs to be addressed. The ecosystem approach emphasises functional characteristics which are the result of processes controlled by the interaction of biotic and non-biotic elements of the ecosystem, rather than focusing on species alone. Biodiversity is a vital element but only a part of these interactions.

Ecosystem functioning generates goods and services which are of value to both organisms and society. Particular species may play key roles in functioning but we are largely ignorant of the extent to which there may be substitutability or redundancy among species in functional terms.

Nearly 40% of the potential net primary productivity of terrestrial ecosystems is pre-empted, used or wasted by humans, or foregone as a result of land use change (Vitousek *et al.* 1986); and between 24-35% of the net primary production of marine upwelling and shelf systems (where the majority of the world's fisheries are concentrated) now ends up in fish caught by people (Pauly and Christensen 1995). Use of resources often proceeds without knowledge of possible side effects, such as those resulting from bye-catches of non-target species in commercial fishing (Alverson *et al.* 1994). It is clearly essential that management practices are made ecologically efficient and do not needlessly destroy biological diversity.

There are implications for conservation. In the past, this focused on the protection of places with distinctive or diverse ecosystems or ecocomplexes. Today the universality of human impact, ever-mounting human needs, and pressures such as those from climate change, make this strategy insufficient. Moreover, it is recognised that human quality of life depends on living within the tolerances of the environment, and using its living resources sustainably. Conservation of individual species remains important, and is one purpose of strict nature reserves, backed by botanical gardens and zoos, but the overriding need is to manage land and sea in ways that balance long- and short-term human needs and protect the long-term integrity of ecosystems. Integrity refers to the ability of the ecosystem to accommodate and to adjust to outside pressures/stress without losses of functioning. This capacity is possible normally only with the flexibility permitted by a certain minimum level of biodiversity.

This means, in turn, that the conservation of biological diversity cannot treat species and habitats in isolation from people. Indeed, the point of intervention is almost always the interaction between people and their environment. Such intervention includes control over the practices of agriculture, forestry or fisheries; over the harvest of wild species on land and at sea; the regulation of waste disposal and emissions of potential pollutants; the protection of soil from erosion; and the management of water resources. It operates on many scales, from the local to the global (although most large-scale action is achieved by a vast number of more localised and smaller-scale activities).

The starting point is to consider what ecosystems are being managed for. There are many purposes, which *inter alia* are:

a) to support people;

b) to maintain appropriate levels of biodiversity;

c) to maintain genetic traits;

d) to maintain ecological functions and biogeochemical cycles;

e) to keep future options open;

f) to maintain aesthetic values, including those in cherished artificial ecosystems such as the chalk grasslands and lowland heaths of Europe, or the terraced rice fields in Bali.

We need to be clear what is understood by "ecosystems" (see Box 1). They are concepts, not precise physical entities: a framework used by scientists to impose conceptual order on the natural world. Yet we are trying to conserve not concepts but real entities - assemblages or communities of particular plants and animals, their diversity and productivity, and habitats in which they live and the dynamic interactions among the various elements essential for ecosystem stability/maintenance and which yield functions vital for both people as well as wildlife. Treating ecosystem conservation and species conservation as if they are alternatives is not only futile but meaningless. Ecosystem management needs to be developed and implemented with this perception in mind. The concept must be used where it can benefit conservation actions, in ways which are as practical as possible.

We must recognise that we do not manage ecosystems as complete entities. Management practices manipulate chemical, hydrological or biological factors that the manager believes will create or maintain desired environmental conditions, particular species assemblages or configurations. A parsimonious approach seeks to achieve desired results with minimal effort. (We need to learn a lot more about how to do this). However, it is important to understand that this manipulation of individual factors cannot take place in isolation. The components of an ecosystem are often complexly inter-linked, so management of one factor may affect others in ways which may not be predictable or fully understood. For this reason species management should not be undertaken without a basic understanding - a conceptual model - of the ecosystem or ecocomplex as a whole, and due consideration of how different factors interact. Furthermore, species management should contribute to achieving the objectives for the ecosystem as a whole.

One problem arises because conservation management is often the responsibility of different government departments or agencies from those responsible for forestry, agriculture or fisheries. This traditional, sectoral approach is an inadequate basis for managing ecosystems to safeguard biodiversity and ensure the sustainable supply of environmental services and natural products. Sound ecosystem management demands a recognition of the complexities of interaction between the cultural and natural environment, and the need to involve all those with an interest in a particular area or region (the stakeholders) in decisions regarding their environment.

In Agenda 21, developed at the Earth Summit in Rio de Janeiro in 1994, it is stated that integrated management of natural resources is the key to maintaining ecosystems and the essential services that they provide. The ecosystem concept is firmly embodied in International Treaties and Conventions such as the Convention on Wetlands of International Importance and the Convention on Biological Diversity. However, experience of fully integrated management and planning of natural resources is limited, especially in the developing world. There is a great deal of expertise in many countries, but most specialists have been trained in narrow disciplines, and there is limited interaction between the traditional sectors and departments.

1.3 The need for Principles of Ecosystem Management

Caring for the Earth (IUCN/UNEP/WWF 1993) set out an analysis and a plan of action to guide policy development which would enable society to put a new ethic for sustainable living into practice (Box 4). Good ecosystem management is one of the fundamental means to that end.

Box 4. Principles of a Sustainable Society (from IUCN/UNEP/WWF 1993)
Respect and care for the community of life
Improve the quality of human life
Conserve the Earth's vitality and diversity
Minimise the depletion of non-renewable resources
Keep within the Earth's carrying capacity
Change personal attitudes and practices
Enable communities to care for their own environments
Provide a national framework for integrating development and conservation
Create a global alliance

Sustainable living depends on the conservation of the Earth's vitality and diversity. But conservation is not simply concerned with protecting or preserving existing species. The objective is at least threefold:

1. Conserve life-support systems. These are the ecological processes that keep the planet fit for life. They shape climate, cleanse air and water, regulate water flow, recycle essential elements, create and regenerate soil and enable ecosystems to renew themselves.

2. Conserve biodiversity. This includes all species of plants, animals and other organisms; the range of genetic stocks within each species, habitats, ecosystems and landscapes. While these are subject to continuing change, management seeks to ensure that changes caused by people create minimal loss of biodiversity.

3. Ensure that any use of renewable resources is sustainable. These resources include marine and freshwater ecosystems, and soils, used in forests, rangelands and cultivated areas, and yielding wild and domesticated organisms and products derived from them.

Ecosystem management is essential for all three objectives. A key test might be whether management maintains the integrity of ecosystems (Box 5). Integrity means much more than the term 'health' which is often applied to describe ecosystems which appear to be in a 'good' (but rarely defined) condition. Some scientists would object to the term 'ecosystem health' because of the inappropriate anthropomorphic connotation. We, the editors, do not share this concern but stress that health is just one dimension of ecosystem integrity.

Box 5. Components of Ecosystem Integrity (modified after Westra 1994)

1. **Ecosystem Health.** The test is that ecosystems function successfully (for some designated property) in their present ("natural" or modified) state. Simplified ecosystems may be healthy even if depleted of certain species by human actions (organically cultivated farms provide one example).

2. **Ecosystem Resilience.** The ability to maintain the essential functions, diversity and structure which enables an ecosystem to recover following disturbance and to remain productive even though changing in response to external stress (see also Urbanska this volume).

3. **Ecosystem Potential.** Retention of capacity for sustainable use. This is fostered by retention of the greatest possible biodiversity, contributing genetic potential and a capacity for continuing evolutionary development.

1.4 Regional differences

Concern over the progressive and widespread scale of human impacts on the planet's ecosystems, manifest in desertification, loss of tropical forests and wetlands and reduction in major fisheries, is universal. However, human impact on land and vegetation is increasing at different rates in different regions.

In North America and Europe, where human populations are broadly stable and agricultural production on the best land has been increased greatly, considerable areas have been released from food production and have reverted to more natural systems (especially woodland). In Vermont, only 20% of the State was wooded a hundred and fifty years ago; today more than three-quarters are under forest that has regenerated by natural processes. In large areas of southern France, abandoned olive groves are reverting to maquis. In Italy, the use of natural gas means areas of maquis are no longer cut for fuel and so are reverting to evergreen forest. Forests in much of Scandinavia are steadily becoming less intensively managed. Furthermore, in the last fifty years considerable areas have been set aside as nature reserves. For the first time in several thousand years, large areas of Europe, although still subject to the impact of air pollution with nitrogen and sulphur oxides derived from fossil fuel combustion, are being used less intensively.

The "new wilderness" as it has been dubbed, (Akeroyd 1995), poses both opportunities and threats. Opportunities, as large areas of land are no longer being intensively changed by humans. Threats, because some of the new ecosystems are less species-diverse than the managed areas they replace. Indeed, the lack of traditional management - like the coppicing of woodland, the mowing of sedge fens, or the grazing and burning of heaths - is now a substantial threat to plant diversity in Europe.

In contrast, in many developing countries there is a mounting pressure for more intensive land management to sustain growing human populations. Agriculture, forestry and fisheries are also extremely important earners of foreign currency. Pressures on the land are aggravated in some regions by poverty and insecurity of tenure. As a consequence, large tracts of forest and other wild habitat are still being converted to low-quality cultivation or pasture. More sustainable land management in such countries will depend on economic growth, supported by the transfer of resources and technology and by new administrative and social measures that will encourage more productive and sustainable agriculture on the soils best suited to it. The principles of ecosystem management must, therefore, take particular account of regional differences in social circumstances, which in turn drive the uses of the environment.

1.5 The implications of recent advances in scientific understanding

Ecological science has evolved since the traditional approaches to conservation management were developed in the 1950s and 1960s, and even since the publication of the World Conservation Strategy in 1980. In particular, it has become clear that:

- plant communities do not necessarily develop along a more or less orderly succession, for example from open water to fen woodland or raised bog, or from land cleared by fire or volcanic eruption through scrub to the 'climax' forest characteristic of a region. It is clear that the processes are much more dynamic and opportunistic. This means that an apparent climax vegetation type will not necessarily be reconstituted after disruption, and even without such disruption is likely to be changing slowly;

- as a consequence, there is no 'pristine' state towards which a conservation manager should seek to 'restore' a modified ecosystem;

- major low-frequency events (major droughts, glaciations, volcanic eruptions and meteor impacts) are likely to be of special importance in causing large-scale ecological change and accelerated extinction;

- population dynamic theories and models, while able to explain some periodic fluctuations in numbers (as in lemmings and snowy owls, or some insect parasite-host and predator-prey systems), cannot yet illuminate more complex interactions: these need to be treated stochastically;

As a consequence, ecosystem managers have been forced to reject the previous assumption that ecosystems are stable, closed, internally regulated and behave in a deterministic manner. Instead they must be treated as open, in a constant state of flux, usually without long-term stability, and affected by many factors originating outside the system. Following environmental change, many individual species will reassemble into different communities, with subtle differences in their ecological processes.

One major stimulus to the current state of flux is the elimination, as a result of almost universal human travel and much deliberate or accidental transport of plants and invertebrates, of the biogeographical barriers that formerly separated major biotas and were an important reason for evolutionary divergence and a good deal of the planetary biodiversity. Today's world is one of 'recombinant biogeography', in which species are moving across such barriers and creating the familiar problem of the 'invasive alien'.

Additional insights have come as a result of understanding of the global climate system and human impact upon it, and through research on the impact of pollution. It is clear that regional changes in acid sulphate and nitrate deposition, and the fertilising effect of deposited nitrogen, are having substantial impacts. These, together with global changes in climate and UV-B penetration through the depleted stratospheric ozone layer, are modifying regional environments faster than ecosystems have had to respond to during the past 10,000 years. It is clear that the global and regional pattern of ecosystems and ecocomplexes is likely to change as a consequence of these major changes in driving variables and new interactions between species. High orders of extinction are inevitable as a consequence of these changes and recombinations, and also as people use more land directly, and appropriate more of the global energy flow.

It follows that it may be counter-productive for conservation managers to select patches of today's vegetation and associated fauna as nature reserves and expect either management or *laisser-faire* to perpetuate them. It is more likely to be effective to set aside a significant area of diverse habitat, with wide altitudinal and microclimatic range, and manage in ways that allow the dynamics to unfold. And, recognising that most of the world's ecosystems and ecocomplexes have been affected by human impact, ecosystem management should seek to incorporate the human element in the dynamic.

This means, of course, that ecosystem managers have to look beyond ecology as a source of the insights they need. Ecosystem transformations are being driven by processes which are studied primarily by disciplines outside ecology, notably economics and social science, and these are important sources of knowledge for the ecosystem manager. Moreover, the goals ecosystem managers themselves pursue have assumed an increased social dimension, in which the aesthetic and cultural importance of particular ecosystems or species is increasingly emphasised.

1.5.1 Species and protected areas.

Much conservation effort has been species focused and has adopted protected areas as the means of delivering habitat or species maintenance. This has required particular scientific approaches and tools; such as, the development of ideas of carrying capacity or optimum sustainable yield. These tools and approaches are not necessarily in conflict with an ecosystem based approach but variation and adaptation is essential if a wider vision of management is to be obtained.

There are schemes that offer at least a spatial solution to the linkage of a protected area approach to the more integrated management of an area for people as well as wildlife. The Biosphere Reserve concept, promoted by UNESCO's MAB Program provides one such approach. In the model reserve a protected 'core' is surrounded by a 'buffer zone' and then a 'transition area'. Use of the buffer is limited to activities which do not conflict with protection of the core; while more intensive human activities are permitted in the transition area. An overall ecosystem management system would go beyond this to ensure that unforeseen damaging effects were not consequential on actions occurring in the transition zone, even though spatially remote (e.g. transfers of pollutants or nutrients by water or air) and that processes in the wider reserve structure were not adversely impacting other ecosystems off-site.

Terminology

A plethora of terms have developed in the scientific and conservation management literature which refer to ecological units at different scales. In addition, there are terms and ideas which have strong parallels with those within that of ecosystem management. They include 'bioregional planning' (Box 6 below and see also Heywood this volume), 'landscape-scale planning', 'landscape planning' and 'landscape management'. There is an urgent need to reduce the growing confusion which seems to be associated with an expanding list of terms being used by different groups of the scientific, conservation and land management communities.

Box 6. Defining a Bioregion

A bioregion is a land and water territory whose limits are defined not by political boundaries, but by the geographical limits of human communities and ecological systems.

Such an area must be large enough:

- to maintain the integrity of the region's biological communities, habitats, and ecosystems;
- to support important ecological processes, such as nutrient and waste cycling, migration, and steam flow;
- to meet the habitat requirements of keystone and indicator species; and
- to include the human communities involved in management, use, and understanding biological resources.

It must be *small* enough for local residents to consider it home.

1.5.2 Scientific Gaps

Despite advances, considerable scientific gaps remain;

Links between biodiversity and ecosystem function

Fundamental questions are being posed in contemporary ecological and policy arenas (Convention on Biological Diversity; Agenda 21; UN Conference on Environment and Development) about the relationship between biological diversity and the functional integrity of ecosystems. The concept of species redundancy, which implies that an ecosystem may change little even if a particular component species is removed, because others are available to take its place, is currently the subject of controlled experimentation and field investigation.

The following are among the many key scientific issues, of direct relevance to the conservationist and ecosystem manager, that need to be resolved:

- how far can the principles developed in any one specific experimental setting be applied to the wide array of communities and ecosystems that exist in nature ?
- are the time-scales of experiments sufficient to test adequately the links between structure and function of an ecosystem ?
- are ethical questions being addressed satisfactorily? Even if a species appears to be ecologically redundant, this does not justify complacency about its extinction;
- has too much emphasis been placed on studying ecological processes under controlled or constant-environment conditions, rather than examining the role that elements of the

biota may play in maintaining ecosystem functional integrity during extreme events? (A species that appears to have little ecological importance today may be a 'keystone species in waiting' with a crucial role following such an event.)

Restoration ecology

Recent developments in restoration ecology have contributed significantly to the remediation of areas already degraded or damaged by human action but further development is still required. There are attendant dangers, however, in assuming that a deterioration in ecosystem structure and function can readily be rectified by ecological restoration techniques. Some fundamental issues relating to this are:

- should new ecosystems be created *de novo* or should existing damaged or dysfunctional systems be rehabilitated or restored? Which approach will be most cost-effective in socio-economic and in ecological terms?

- are policies such as the US 'no net loss' approach to wetlands over-confident? Is it really acceptable to destroy a natural or semi-natural area in one locality (for industrial or agricultural economic reasons) only to replace it with a synthesised wetland of similar size in another location, especially given that it is virtually impossible to create one with identical ecology and biodiversity ? Is *restoration* ecology really advanced enough to be termed *creation* ecology? Can we bring about ecosystem restoration?

- can restoration ecology help in the preparation of areas for managed retreat (for example where, as a result of global sea-level rise, estuarine systems, salt marshes and mangroves may need to shift laterally into areas currently in use for agricultural, industrial or domestic purposes)? Is this a socially acceptable approach and if so, how can it be done best?

Island biogeography

Long term experiments and studies of island biogeography (Box 7) could contribute further, evaluating optimal sizes and patchiness in protected areas. Recent developments in metapopulation ecology, concerned not with continuous populations of a given species but with groups of populations existing at the same time but in different places, are also providing new insights into population dynamics at broader geographical scales. The somewhat paradoxical importance of maintaining mosaics and corridors of habitat that meet the needs of organisms not yet there are becoming increasingly apparent as a result of such studies. The implication is that managers need to maintain habitat structures which permit organisms to move in response to changing environmental scenarios or ecosystem management regimes.

A number of important scientific questions remain:

- should conservationists and managers always seek to maximise the size of an area under management or protection?

- as a "rule of thumb" does it make sense to devise management strategies oriented principally towards conservation of organisms at the top of the food chain; will this always ensure that organisms at lower trophic levels are adequately protected?

- more research is needed on the merits and limitations of managing several smaller units of the landscape (e.g. 10 units of 4 hectares) compared with one large area of the same overall size (e.g. 40 hectares). Under which circumstances (both ecological and socio-economic) are managers and conservationists justified in opting for which of the contrasting approaches?

- given that metapopulation studies have, for inescapable practical reasons, been dominated by a consideration of individual taxa, how can the scientific methodology and debate be broadened to integrate several taxa?

- how can we ensure that we have information that is representative of the full range of taxa within an ecosystem?;

- how can metapopulation studies be used to their full potential by conservationists and ecosystem managers?

Box 7. Island Biogeography and Metapopulation Studies

Classical island biogeographical theory, as developed three decades ago by MacArthur and Wilson (1967), has been hailed by many ecologists and conservationists as a fundamental paradigm of contemporary ecology, with profound practical implications for the design and management of protected areas and the management of individual taxa. The theory suggests that:

- the size of a protected area influences the equilibrium number of species that it can hold;

- the shape may be important too. It is not clear whether it is best to minimise the ratio between the perimeter length and overall area, (because circular reserves avoid isolation and extinction due to edge effects), but ecotones, boundaries and edges are often areas of great biodiversity;

- as a precaution against climate change, it may be most effective to align nature reserves along the axis of change rather than perpendicular to it, so maintaining the potential for migration and colonisation in the future.

Databases and integrated spatial information systems (GIS)

Over the last 15 years there have been dramatic developments in fields such as GIS, remote sensing, global on-line networking and data management. Conservationists and ecosystem managers have been slow to realise the potential of these areas for designing, implementing, and monitoring the effectiveness of ecosystem management measures and the development of expert systems for ecosystem management.

The new Biodiversity Conservation Information System (BCIS) initiative of a consortium of conservation bodies is an example of an innovative approach to database assembly to produce added value for improved biodiversity management. The IUCN Commission on Ecosystem Management is developing its 'Red List Approach to Threatened Ecosystems' with BCIS partners to take full advantage of information already existing in dispersed forms.

Other areas

At the same time as the improvements have occurred in technology, there has been a decline in taxonomic expertise, especially in developing countries, where there are few taxonomists and collections established by expatriates are deteriorating. Even where effort has been made, knowledge is still insufficient. For example, according to recent estimates there are about 1200 different fish species in the Mekong, but only about 500 have been registered in the field identification guide *Fishes of the Cambodian Mekong* (Rainboth 1996).

There is often a lack of connection between empirical research on an ecosystem and its management.. Scientific results must be transferred into ecosystem management techniques more effectively (and cost-effectively). One task for a "think-tank" like the IUCN Commission on Ecosystem Management might be to review the results of research on ecosystems and use them to define goals and options for ecosystem management.

Other topics include the extent of redundancy in ecosystems, the reality of keystone species, and the nature and extent of resilience (see Urbanska this volume).

1.6 Ten Principles for Ecosystem Management

The Sibthorp Seminar distilled its key conclusions as a series of Ten Principles for Ecosystem Management, set out in Box 8 and elaborated in the following sections.

Box 8. Ten Principles for Ecosystem Management	
Guiding Principles	1. Management objectives are a matter of social choice 2. Ecosystems must be managed in a human context 3. Ecosystems must be managed within natural limits 4. Management must recognise that change is inevitable 5. Ecosystem management must be undertaken at the appropriate scale, and conservation must use the full range of protected areas
Operational Principles	6. Ecosystem management needs to think globally but act locally 7. Ecosystem management must seek to maintain or enhance ecosystem structure and functioning 8. Decision-makers should use appropriate tools derived from science. 9. Managers must act with caution 10. A multi-disciplinary approach is needed

1.7 Guiding Principles

1.7.1 Management objectives are a matter of social choice

(a) Rationale

Ecosystems change over time as a result of natural processes as well as human pressure. There is no single reference point to decide exactly what should make up an ecosystem, as a basis for management targets. Science cannot, therefore, provide an objective definition of what a particular ecosystem should be like. Furthermore, a great many ecosystems have been altered massively by human intervention; in many cases it is difficult to distinguish between natural drift and historical human action.

Social choice will determine which ecosystems are conserved, how they are managed and what they are used for. The choices will be based on judgements of need, value and benefit. Furthermore, objectives and approaches to conservation differ according to the specific needs of particular communities. Ecosystem management will therefore need to balance the sometimes conflicting demands of different interest groups and the alternative means of achieving an appropriate relationship between people and their biotic and abiotic environment. Science can advise on the implications of various ecosystem management objectives and whether they are realistic, but cannot make decisions regarding the choice of objectives.

Choices are greatly influenced by social conditions, such as security of land tenure, and by social imperatives such as the relief of poverty. Where access to resources is open and people are driven by immediate needs, then long term sustainability is readily discarded. Ecological sustainability has social sustainability as a prerequisite. Ecosystem management can only be fully effective if the welfare of human communities is secure. Yet ecosystem management can contribute significantly to the maintenance and enhancement of human welfare.

(b) State of scientific understanding

Social choices are driven by perceptions of value and opportunity. Techniques have been developed to quantify the value that people place on goods and services provided by natural systems such as wetlands (Barbier *et al.* 1997), or woodlands (Willis and Garrod 1992). There is a substantial literature on the valuation of non-timber forest products, such as medicinal plants, and of biodiversity, and considerable discussion of how to build intangibles such as cultural value into social decision processes. But social choices will inevitably vary between societies and be influenced by their state of economic development, and the implication is that different approaches to ecosystem management may be chosen even in situations that are ecologically similar.

(c) Examples of relevance/application of the principle

Ten thousand years ago, Chobham Common (in Southern England), was a barren tundra. Then, between roughly five and ten thousand years ago it was progressively birch, pine and oak-hazel woodland, and finally under the combined human impacts of felling, fire, primitive agriculture and grazing it turned into heathland (ericaceous shrub dominated lowland with thin acid soils) . Now, if left unmanaged, it reverts back to scrubby birch-pine woodland. But 'society' has decided that heathland is more valuable from a nature conservation and amenity point of view than birch-pine woodland, and Chobham is one of a number of heaths actively managed to restore and maintain this increasingly rare ecosystem. The decision has no rational basis in science, and indeed works to arrest natural processes of ecological change: it also reduces potential biological productivity, diminishes the value of the area as a 'sink' for atmospheric carbon dioxide, and may well reduce its biodiversity.

It is increasingly recognised that the goals of ecosystem management in a region should be decided through dialogue involving all interested parties, especially the area's inhabitants. The role of external scientists and experts is to listen, help to define possible choices, and suggest approaches, but not to press particular solutions (Chambers 1993). Science can tell us about the likely biological outcome of a decision or action, but not which, of all outcomes, we should value highest. The appropriate solution should be decided by the community. The process is well illustrated by the community-based schemes for conservation and sustainable resource use that have been established in many locations, for example forest villages in India (Poffenberger and McGean 1996). However it is generally much more difficult to resolve at larger scales such as regional, national or supranational populations.

(d) Needs for the Future

It is important to build awareness among local people and governments of the importance of natural systems and their biodiversity, not least in providing economic benefits, and to help them understand the choices for sustainable resource management, so that a community makes ecologically sound choices and allows ecosystems to be managed in a sustainable way.

For this to be achieved, those making the choices need to know the capacity of the systems to deliver various benefits and the value of the goods and services they can provide, and to have the means of managing them and making use of what they offer. Links between conservation and development need to be strengthened, so that improvements in welfare are sustainable ecologically and integrated with sound ecosystem management.

It needs to be recognised generally that externally dictated scientific approaches are unlikely to succeed, and that there must be dialogue involving all interested parties, and especially local communities, in decisions about management objectives.

Traditional use of natural resources was based on the belief that use should not be restricted unless someone objects and can show that their welfare is affected adversely. This was reasonable when resource use was small in comparison to availability and users were part of the local community. As resource use grows with standard of living and expectation, users are increasingly not part of that community (a situation well demonstrated by foreign fishing fleets or international logging companies) and inequities can be aggravated. New kinds of dialogue are needed to resolve conflicts and build consensus. This may mean changing the concept of a *right* to use resources to one which sees capacity to use them as a *privilege*. Even private owners of resources must recognise the potential effects of their actions far from their own location, or at some distant future date, and be held accountable for adverse impacts (Mangel *et al.* 1996).

Social choice in a democracy expresses itself, theoretically, through government policy, which provides guidance to decision makers. Governments are also the custodians of the rights of the community as a whole, and of future generations. Agenda 21 makes it clear that Governments must take responsibility for sustainable development as a national policy, and facilitate it through actions at all levels of society.

1.7.2 Ecosystems must be managed in a human context

(a) Rationale

We have no choice but to manage ecosystems in a human context. Humans have been exerting large-scale impacts on their environments ever since people began to use fire, and especially since agriculture began, thousands of years ago. The human species is now pan-dominant and there is no place on earth unaffected by its influences. It is a principal end-user of biological productivity and as human numbers and consumption (especially of energy) continue to increase, humanity's influence will become even greater. But biological capacity is not unlimited. Humanity must change the way it interacts with the ecological systems which directly or indirectly support it, or there will be risk that environmental and socio-economic systems will collapse and the quality of life in those societies that fail to make this essential adjustment will decline.

The challenge is to manage the relationship between people and nature in all its various aspects. Even in protected areas, management must take account of the human dimension, for people can rarely be excluded, and policy will always be driven by wider political goals. Ultimately, ecosystem management will always be a question of distribution of costs and benefits amongst different interest groups in society. Science is applied in a social context and social assumptions and imperatives determine what ecosystems are managed for.

(b) State of scientific understanding

The role of human influence in determining the structure and function of ecosystems has long been established, especially in temperate regions. The development of moorland and heath from prehistoric forest in oceanic and continental Europe and the recovery of forests through abandonment of farm practices in the north-east United States are familiar examples. The recognition that even tropical rain forests have been profoundly affected by people is more recent (Denevan 1992). But the converse is also true: ecosystem functioning has influenced the development of the communities which rely upon the natural resources supplied by the ecosystem. The life styles of the Arctic Inuit and of many fishing communities illustrate this.

We increasingly understand the nature and scale of human effects on ecosystem health and integrity (see Box 5) but there is still difficulty in defining levels of tolerance, resilience to change and carrying capacity. Some scientists stand by the need to define such concepts better by means of fundamental and applied research. Yet others feel that these concepts are too vague to be useful and do not merit investment of research time and money. Probably the greatest deficiency is in understanding the relations between human actions and ecosystem function, diversity and resilience. The concept of carrying capacity also begs many questions - what is being carried, by what kinds of ecosystem process, with what form of sustainability, over what period?

(c) Examples of relevance/application of principle

Traditional practices have in some cases created distinctive systems. For example, traditional Alpine grazing maintained herb-rich upland meadows, and the frequency, timing and method of grass cutting on the wetlands of the Somerset Levels in South-west England led to a unique assemblage of flora and fauna. The Keoladeo National Park in India is another good illustration. It is a small wetland located near Bharatpur on the Gangetic plain, designated by local kings in 1750 as a man-made wetland to attract migratory birds for hunting, for which it became famous. The present day rich biodiversity, particularly as a bird habitat with over 354 recorded species, including wintering Siberian cranes, has been maintained by water management and the centuries-old tradition of grazing buffalo, which control the growth of water weed, *Paspalum distichum*.

But these traditional methods may no longer be socially appropriate and their abandonment means ecological change. In the Alps, as grazing has declined, scrub and forest have taken over from meadow grasses and flowers. In European 'Environmentally Sensitive Areas' farmers have been paid subsides to continue traditional management methods, as one means of maintaining biodiversity and landscape quality. In the Keoladeo Park, problems arose in 1991 when an amendment of the Wildlife Protection Act prohibited the grazing of livestock in a National Park. The removal of buffaloes caused changes in the ecosystem of the Park due to the build-up of plant biomass. The Siberian cranes were less able to find the tubers on which they feed, and their numbers declined. The prohibition of grazing also led to a breakdown in relations between the Park and the people living in the surrounding area.

Keoladeo illustrates how the uncritical application of an imported concept - that the consumptive use of natural resources should not be permitted in National Parks - led to the erosion of the values for which the Park was established. In contrast, the CAMPFIRE (Communal Areas Management Programme For Indigenous REsources see Box 9) programme in Zimbabwe has had some success in dealing with the problem of ecosystem management in open access areas. It allows local communities to derive benefit from the sustainable exploitation of wildlife resources on communal lands for meat, trophy hunting, and products sold to the tourist industry. It has given the communities concerned not only a cash return but also an incentive to conserve indigenous species, grazing native vegetation, rather than converting these lands to cattle ranching.

Box 9. An Example of Historical and Social Factors in Ecosystem Management

The interplay of ecosystem management and socio-historical factors is well illustrated in parts of Africa. Prior to 1850, much of the land was treated as a communal resource, managed by traditional cropping and grazing systems which were sustainable where people lived at low densities. Traditional cultures were disrupted in many areas by the slave trade and then by European colonisation. The latter imposed cultivation and management practices ill-suited to landscapes, soils and climates that the colonists understood only poorly. (The East African groundnut scheme in the late 1940's was a catastrophic example of this). A large amount of the least productive land remained open access but with higher population density. Therefore, the degradation was higher but there was no incentive for sustainable management. Radical changes in approach clearly became necessary and have been sought progressively in different countries following independence. The Zimbabwe CAMPFIRE programme is one example of such an innovative approach.

(d) Needs for the Future

It is clear that management must be developed with people as part of the equation - defining what the ecosystem or unit is managed for (including aesthetics), and recognising that local concerns and needs often dominate but that the wider context must also be recognised. The distinction between 'protective' and 'productive' ecosystem management should be treated with caution. Risk assessments and cost benefit analyses are useful tools in judging between options, but intangible and unquantifiable benefits must also be considered. The ecosystem management strategies ultimately adopted must reflect human needs and capacities, and the methodology for achieving this needs further improvement.

Consideration must be given to who is to manage a particular area and its component ecosystems, and how management decisions fit the wider political process. Examples of 'comprehensive land utilisation' strategies combining forest or other semi-natural habitats, agriculture and aquaculture should be investigated more fully to assess the possible benefits to both biodiversity and to people by this approach.

1.7.3 Ecosystems must be managed within natural limits

(a) Rationale

There are practical limits to the conservation of ecosystems. In theory, by manipulating key factors, they can be conserved in situations where their persistence would otherwise be most unlikely. However, such extreme action is rarely appropriate in natural habitats (some botanical gardens and enclosed artificial habitats do maintain such systems, for conservation, academic, educational and commercial benefits; pineapples are grown in greenhouses in Scandinavia and 'orangeries' for citrus trees were a feature of many European great houses). Management efforts will be far less, and social options greatest, where they address ecosystems that occur naturally in the region in question. Each system needs to be evaluated in a wider context, so that conservation effort is not wasted on areas where it cannot be successful: for example, at the extreme limits of the system's potential existence.

(b) State of Scientific Understanding

Science can define where ecosystems are near the limits of the environmental conditions which maintain their character (e.g. isolated forest patches critically dependent upon hilltop mists or Arctic-alpine relicts dependent upon mineral-rich springs and cold micro-climates), and where they depend upon artificial conditions created by human action (e.g. grasslands below mean sea level, dependent upon artificial sea walls or irrigated croplands and palm groves in an arid desert).

If the environmental tolerances and requirements of species can be defined, it becomes possible to determine how near they are to their limits in a particular situation. At present, however, this has been done for only a handful of species and ecosystems. Even then, there is no agreed rationale to establish the conservation value of maintaining ecosystems at the edge of their range where this demands major and costly management effort. Research on climate trends is, however, promising to yield information on the probability of changes that will make the perpetuation of a particular ecosystem impossible.

In contrast climate change may make other marginal ecosystems more stable. Many species only exist in very extreme environments and survive through extreme events: for example, the seeds of many desert plants lie dormant until a rare rainfall occurs. It is very difficult to assess whether a change in the frequency of such rare events, caused by climate change, will improve the survival of such species.

(c) Examples of relevance/application of the principle

It is not always practical or feasible to conserve certain ecosystems. For example, it may make better sense to re-establish coastal marshes in place of artificially maintained pastureland defended by expensive flood protection structures. Salt marsh would be the natural ecosystem to occupy these areas. The idea of 'managed retreat' in reaction to rising sea levels is being applied already but analysis of inevitable change in environmental conditions , as a result of human land and water use or climate alteration, should be taken to the point at which the viability of present ecosystems and the possible future patterns of species recombinations can be assessed.

If the likely changes in climate or other factors can be defined, it should become possible to establish the viability of a system, or the management effort required to perpetuate it. It may become clear that certain small nature reserves, established to protect a species or ecosystem today, cannot possibly serve that function in the face of inevitable change, and further major management expenditure that flies in the face of nature may then be judged pointless.

It may also become evident that some ecosystems have been damaged beyond their capacity to maintain functioning and character (see Box 5), even with extensive management. For example, in one study, extensive ski run construction involved removal and disposal of topsoil. This left mineral soils with truncated horizons and no plant cover. In most cases natural re-colonisation was extremely slow, but the colonising species were consistently a mixture of various successional stages, among them even, so-called climax species: colonisation did not follow the classical successional principle. In such cases ecosystem management may never be able to restore previous vegetation and fauna. Inappropriate action may be as bad as, or worse than, taking no action, as it may waste effort and scarce resources.

(d) Needs for the future

The central problem is the feasibility of maintaining biodiversity within the land and water use patterns that respond to human demands and inevitable environmental change. For nature conservation, the issue is how far it will be feasible to maintain unconnected and isolated reserves surrounded by hostile or unsupportive areas. There are many biogeographical similarities between true islands and habitat islands, but there are also substantial differences. The application of island biogeography to ecosystem conservation needs to be related to the type of landscape being managed.

More studies are needed on:

- the effectiveness of conserving fragments of original ecosystems when climate and other changes impose mounting stress;
- the relative rates of change in ecosystems, species and driving variables and the implications for the feasibility of conserving current patterns of biological diversity in particular areas as natural limits change;
- the importance of past events in constraining management options for the future;
- the boundary conditions for species survival, including minimal and optimal patch size and proximity, and the capacity of species to disperse across the matrix.

Restoration ecology needs to be guided by these considerations and metapopulation dynamics has a great role to play here. It will generally be counter-productive to make great efforts in order to hang on to species at the retreating margins of their distribution.

Environmental change will lead to changes in the ability of ecosystems and species to sustain their present geographical positions. Ecosystem management must address whether the future environmental variables will be suitable for the ecosystem that is currently being managed. If not, it will need to consider whether the effort should be abandoned, and what new options should be considered.

1.7.4 *Management must recognise that change is inevitable*

(a) Rationale

There is no stable "pristine state" to which managers can seek to restore particular systems, or a climax that persists indefinitely. All ecosystems are in a constant state of change due to the internal dynamics of populations, natural processes of evolution and dispersion, changing external inputs such as climatic oscillations (see Box 10), and the redistribution of species by natural dispersion and human action. Since different species respond in different ways to environmental change new assemblages of species will develop, producing new ecosystems.

Ecosystem management must be planned in recognition of the lessons of new thinking in ecology and of observed changes in the biosphere. It should accept the inevitability of continuing changes, including the redistribution of species now that biogeographical barriers have been eroded by human actions. The treatment of all "alien" and invasive species as unacceptable intruders may have to be replaced by a more pragmatic judgement based on recognition that organisms have always redistributed themselves, and on the acceptability of their new ecological role and the feasibility of control.

Box 10. Climatic Change

Some 10,000-20,000 years ago, during glacial phases in high latitudes, rainfall over the current Sahara desert and Middle East was much higher and percolation of water to underlying rocks led to the build up of substantial groundwater resources (Goudie 1977). However, the recent drier climate in these regions means that recharge is much reduced and groundwater exploited is not being replaced at the same rate. Superimposed upon natural climate cycles are human-induced global changes. The consensus is that during the next century global temperatures will rise by about 0.2 C per decade (IPCC 1996), with some areas exceeding this rate and some areas cooling. However, it is uncertain how this will affect water resources. Evaporation is likely to rise, but changes in rainfall patterns are less easy to predict. However, it is feared that many areas will become drier and that floods and droughts may become more frequent and more extreme.

Management should take account of likely major changes in habitat conditions (notably climate and hydrology) and seek to maintain habitats for those present species that will probably survive and those likely to invade through natural processes. It should discriminate between changes in species composition that are socially acceptable and ecologically tolerable and others that are not. It may need to consider the introduction or reintroduction of appropriate species as part of a wider programme of biodiversity conservation and so as to maintain ecosystem integrity.

(b) State of scientific understanding

The classical belief in a stable climax system has been questioned and replaced increasingly by a 'nature-in-flow' paradigm. The exact species make-up of an ecosystem is certainly liable to change over time, and examples of the same basic ecosystem differ in their precise species composition.

Mapping species distributions and correlating such distributions with gradients in climatic factors can reveal potential habitats for the future, given certain climate change scenarios. Many palaeoecological studies have provided strong evidence for a considerable turnover of species. European and North American trees shifted their range by 2000-3000 km in the time between glacial and interglacial maxima. Some models indicate that the range of European beech (*Fagus sylvatica*) may shift thousands of kilometres in response to future climate change. Too many conservation management plans implicitly assume stability in the climate and other driving variables; they must be adjusted to take full account of the probability of the changes now indicated by science. But precise prediction is not feasible: ecosystem management must recognise the stochastic nature of ecological and population models. Moreover some evolutionary changes may be too slow to respond to sudden shifts in climate and rapid habitat changes.

It is now widely recognised that non-linearity and threshold effects are pervasive in biological systems. A pathogen may suddenly become a plague once it passes a threshold density: reproduction may not occur until population densities pass a threshold high enough for individuals to find each other; and populations of a given species may only be viable above some critical threshold of patch (habitat) size, below which a refuge is ineffective (Mangel *et al.* 1996). Such effects are all non-linear, and imply, for example, that a given change in population may have a different effect depending on the actual population density, and that changes can occur suddenly and unexpectedly (May and Oster 1976).

(c) Examples of relevance/application of the principle

Ecosystem managers must reject assumptions of steady state systems and look at models and analyses that indicate the most likely dimensions of environmental change and ecological response. Some of these may be very large: for example climate change models suggest not only modifications in global and regional temperature and precipitation, but also that a cut-off in the formation of cold deep water in the North Atlantic could cause sudden changes in ocean circulation (including the Gulf Stream). Ecosystem managers need to consider what these alterations could mean for biodiversity, productivity and conservation.

The safest course, in the face of inevitable change, is to ensure that species and ecosystems have the opportunity to redistribute themselves. An obvious consequence is the need to make management units or protected areas as large and diverse, topographically and climatically as possible. Thus the creation of large national parks like the Serengeti is a good application, and the principle that nature reserves should be no larger than required to safeguard the species that they contain under today's conditions is a denial of the principle. Another technique is to plan for reserves to be linked by corridors, so that migration is possible:

Species-area relationships can predict how many species are likely to become extinct as a result of reduction of ecosystem size. However, this prediction does not show when this may occur.

Species may be able to persist in affected ecosystems for some considerable time before being lost. For example, despite the dramatic alterations of the landscape in Europe over the past 400 years plant species extinctions have actually not been common. This may be both good and bad news. It may mean that if ecosystems can be brought under prudent management then the loss of these species may be delayed or even prevented. However, species may remain but be incapable of regeneration, disappearing from the ecosystem when the last surviving individuals die: such species may be described as 'functionally extinct'. Other species may have enhanced vulnerability to disturbance. These characteristics must be defined further as an essential basis for management.

(d) Needs for the future

A distinction must be drawn between managing *for* change on the basis of the best scientific knowledge and management *of* change (for example by controlling greenhouse gas emissions). Both are important, but ecosystem management is likely to be concerned particularly with the former.

Managers must expect, accommodate and manage for change rather than obstruct it (see Box 11). They must evaluate probabilities and risks, and incorporate flexibility (taking account of the precautionary principle). Restoration ecology must be viewed as the art of the possible. Management should facilitate the dynamic response of species, using techniques such as natural regeneration and deliberately assisted re-distribution. Landscape management must provide heterogeneity and connectivity at all ranges of spatial scales.

Box 11. Processes of Colonisation and Dispersal.

Recent developments in palaeoecology, together with the study of contemporary patterns of dispersal and migration, have indicated that a dynamic biosphere, with a rapidly changing climate, will result in the formation of ecosystems with unfamiliar configurations of species. Ecosystems for which there are no contemporary analogues can be found in the Quaternary palaeoecological record, and new configurations should also be expected in the future. These contrasts arise because ecosystems do not migrate en bloc in response to environmental change, and species have different rates of dispersal. The implications are:

- we should expect and plan for new and unfamiliar species configurations in changing ecosystems;
- we need to understand the impacts of human activities (e.g. land-use change, transport of soil, plants and other goods, international human travel) on rates of dispersal;
- we should be prepared to intervene, if necessary, to assist the dispersal of individual species where this is important ecologically or socially, and will help maintain options for the future.

Leaving re-distribution to nature will mean many extinctions and major ecosystem transformations. There is a need to identify which species will survive where, and whether it is worth helping them to survive. Biodiversity conservation must take a wider view, abandoning fine distinctions between 'natural' and 'alien' species in particular areas. Species most at risk from climate change will be those with little overlap between the current and future ranges. These species and their (future) ecosystems should be targeted for actions such as pro-active reintroduction programmes. The principle implies a need for analysis of whether conservation effort in these instances has any value.

Research is needed to provide greater understanding of the ways in which environmental factors determine species distribution, and of how species respond to change, especially by altering distribution. There is a need for greater understanding of the nature of these factors and the speed of their operation.

1.7.5 Ecosystem management must be undertaken at the appropriate scale, and conservation must utilise all categories of protected area

(a) Rationale

The concept of integrated land management, emphasised in Agenda 21, implies the optimisation of resource use. It demands an evaluation of a region, with its catchments, coastal seas and diverse land areas, a consideration of the different purposes for which they are used, and an assessment of how the components of the pattern interact and how far the use of each is optimal and sustainable.

Ecocomplexes range in scale from small desert oases a few square metres in extent to large continental forests such as those of Amazonia. For ecosystem management to be successful, it must be undertaken at the appropriate scale. Because small sites are affected by processes and practices beyond, there is an increasing appreciation of the need to manage defined landscape units, such as coastal zones or river basins. Ecosystem management must examine how processes and management practices in one part (for example, upstream in a river basin) affect other parts of the area and how, in turn, management of the area affects adjacent sites (Acreman and Lahmann 1995).

The appropriate management scale depends upon the structure of the system, the aims of integrated land use, the scale of natural disturbances (e.g., fires, landslides, floods), pertinent biological processes (e.g.. disease, foraging, reproduction) and dispersal characteristics and capabilities of the component populations. The fundamental unit for water-related management issues is normally the drainage basin, as this demarcates a hydrological system, in which ecosystem components and processes are linked by water movement. It is important to remember however that surface water and groundwater catchment areas may not coincide. For issues where air quality is influential, such as acid rain, the "airshed" (as opposed to the watershed) will be more appropriate, implying the integrated management of source areas (for example industries in UK or Germany) and receiving areas (such as affected river basins in Scandinavia) (Acreman 1997).

The same principle of management at the appropriate scale applies to protected areas. The traditional view has been that these are islands of natural or semi-natural environment in an otherwise human-modified world. National Parks and nature reserves conforming to IUCN Protected Area Management Categories I to IV have been set up in many countries. They are indeed an essential means of ensuring the survival of ecosystems and the biodiversity within them, but they need to be planned and managed within their broader eco-regions.

IUCN also recognises two other categories of protected area;

V Protected Landscape/Seascape and

VI Managed Resource Protected Area.

Such areas are places where the sustainable use of natural resources and the conservation of species and habitats need to be blended. Areas suitable for designation under category V include the long-managed landscapes of parts of Europe which are rich in both cultural and natural values, or the rice terrace landscapes of South East Asia where the soil, water and vegetation cover protection have been practised for centuries.

Category VI areas might be forests in tropical areas that have maintained their largely natural qualities but have also been managed by local communities for a sustainable supply of products (e.g. rattan, honey, meat from wild species, medicinal products, nuts and fruit), or in-shore or lake fisheries which have been managed sustainably by local fishermen.

Category V and VI areas are important, both for the direct contribution that they make to human welfare and for the part they can play as 'models' of sustainability to be replicated elsewhere. Hence the growing interest in including such areas in national and local plans for integrated and sustainable land management. It is inevitable that only a small proportion of the planet will be designated in the most strongly protected area categories and so sustainable natural resource use will focus increasingly on land outside the more formal protected area network. These are likely to be areas such as large-scale biophysical systems such as river basins, coastlines, mountains, rangelands and large marine areas. Scientists, agencies involved in sectoral land management and owners will need to work together to develop practical solutions for implementing sustainable ecosystem management.

(b) State of scientific understanding

Although the concept of an integrated approach is promoted by a broad cross-section of scientists as well as politicians, what this actually constitutes is not clearly defined. However, the science of land evaluation is well advanced, although it often fails to take sufficient account of likely changes (see 4.6.4). But detailed understanding of the sizes of the unit that are necessary for effective ecosystem management is much less well developed, and methodology for assessing the benefits of alternatives (for example of managing a forest for timber, by rotational harvest or for multi-purpose extraction of non-timber forest products, or for ecotourism, or - as an extreme case - by replacing the forest by cultivation or pasture) is less perfected.

Very similarly, there is a great volume of knowledge of the techniques appropriate for the management of the different IUCN categories of protected area, but less appreciation of how non-ecological sciences (especially social sciences) should be applied when overall plans for Category V protected areas (Protected Landscapes/Seascapes) are prepared. There is a considerable need for methodological development in this whole area, before this principle can be applied effectively.

(c) Examples of relevance/application of the principle

The current interest in national and local Agenda 21s, National Conservation Strategies and sustainable development strategies, and in other plans which set out to optimise land use in river basins or administrative units are an illustration of how the principle of managing ecosystems or ecocomplexes at the appropriate scale is gaining ground. National Conservation Strategies in many countries seek to review the state of the national environment, review resource use impacts and linkages, and define operating principles and mechanisms for achieving wide goals of conservation and sustainable development (PNCS 1992, Box 12).

So far as protected areas are concerned, the issue is essentially whether the right units have been chosen. Major national reviews such as the UK Nature Conservation Review (Ratcliffe 1977) and Biodiversity Action Plan (1994) address this: if the units are appropriate, ecosystem management plans can succeed, but if they are not, areas may not be viable and efforts may be wasted. Area in this connection relates closely to the species to be managed: a viable population of large predators, such as wolves, will demand a much greater area than if such keystone species have been eliminated and human management is used to control large herbivores such as deer.

There are examples of successful management of protected landscapes. In Luzon in the Philippines, for example, rice terraces cover some 20 000 km^2, and date back 2000 years. Much is on very steep slopes, and yet in a wet and unreliable climate (annual rainfall of more than 3000 mm) and in an earthquake zone, the soil, water and forest resources of the area have been effectively conserved and the long-term productivity of the area maintained. In contrast, in many other areas of rugged terrain in tropical countries soil erosion has occurred. The local Ifugoa people have created this landscape and are bound together by strong cultural traditions, centred around an annual life cycle which is geared to rice production. The Philippines Government has drawn up a master plan for the protection and restoration of the terraces, which involves working closely with local communities and developing the area's economy in a sustainable way, including development of ecotourism based upon the area's scenic and cultural interests.

Box 12. Fourteen Priority Programme Areas in Pakistan (PNCS 1992)

1. Maintaining soils in croplands
2. Increasing irrigation efficiency
3. Protecting watersheds
4. Supporting forestry and plantations
5. Restoring rangelands and improving livestock
6. Protecting water bodies and sustaining fisheries
7. Conserving biodiversity
8. Increasing energy efficiency
9. Developing and deploying renewables
10. Preventing/abating pollution
11. Managing urban wastes
12. Supporting institutions for common resources
13. Integrating population and environment programmes
14. Preserving the cultural heritage

The Boabeng-Fiema Monkey Research in Ghana illustrates appropriate ecosystem management of a category VI protected area. It is a remnant forest in the lowland zone. It retains most of its natural qualities, being protected by the local community which reveres the resident troupes of black and white colobus and mona monkeys, and as a result other wildlife also benefits. However, the development values of the area are equally impressive; the reserve provides a regular supply of clean water, forest palms produce palm wine which is consumed locally and sold to visitors to the reserve, bee hives are maintained by the community in the forest for their highly prized honey, materials taken sustainably from the forests are used to produce baskets and other handicrafts for local need and also for sale to tourists. In addition, some limited parts of the forest are farmed periodically under a traditional rotation system.

(d) Needs for the future

The most important requirement is for countries to develop integrated resource management, planning at a bioregional level rather than relying on administrative boundaries. Ecosystems should be managed within the context of large-scale biophysical systems such as river basins, coastlines, mountains, rangelands and large marine areas. National Conservation Strategies and National Sustainable Development Strategies should be prepared, to define the optimal mosaic of land and water uses, and the units for ecosystem management. It is important to develop ways of implementing the programme priorities such as those of the Pakistan NCS within the frame of the guiding principles of ecosystem management.

Within their wider strategies and policies, countries should develop systems of protected areas, using the full range of IUCN categories, rather than, as so often in the past, only networks of highly protected parks and reserves. Such systems would enable a range of environments to be protected, from the near natural areas to those where a long history of a sustainable relationship between humans and nature is evident. The importance of including the human modified areas within the protected areas network is that they not only contain important natural values in their own right (many modified habitats are rich in wildlife, some species indeed now being dependent upon human modification to survive), but also because the management of such places has important messages for other areas which are not formally protected; they can be 'greenprints' for rural areas generally.

However, the protection of human-modified areas requires a range of novel skills, not always well developed among protected area managers trained in the conventional way. They may need to offer agricultural extension advice and fishing assistance - and generally find ways of encouraging the survival and reinforcement of traditional ways of resource use. Clearly this is a difficult area, with its emphasis on a multi-disciplinary approach (see 1.8.5 later) to ecosystem management. There is another benefit associated with a subtle shift in emphasis towards human-modified areas. It will further a process of change which will engage more of the sectoral interests which must be integrated to achieve more widespread application of integrated ecosystem management.

1.8 Operational Principles

1.8.1 Ecosystem management needs to think globally but act locally

(a) Rationale

The ultimate scale of ecosystem management is global, since all components of the earth's system are interrelated. Policies for ecosystem management may be developed by intergovernmental panels at a global scale, such as international treaties to reduce CO_2 emissions, or at the landscape or large marine/coastal unit scale, by national organisations. However, these policies are only effective if implemented at the local level by local authorities, companies or individuals, and they are more likely to be concerned about issues such as jobs, food and health.

Probably the best example of thinking globally but acting locally is public reaction to stratospheric ozone depletion. This is truly a global issue, but it captured the imagination at the local level. Influenced by some good publicity, people demanded CFC free spray cans and safe ways of disposing of old refrigerators. This fed back to pressure on manufacturers to provide substitutes and on politicians to eliminate ozone-depleting chemicals.

Ecosystems also have to be managed at the local scale, often by people living on the ground. Success therefore depends on local social conditions, including security of land tenure and the nature of social imperatives, such as poverty. Ecological sustainability has social sustainability as a prerequisite. Problems in areas of human pressure need to be addressed first locally, then regionally, and globally last of all.

Global biodiversity conservation therefore depends on the success of these innumerable local actions for appropriate ecosystem management around the world. Local thinking and action need to be informed by awareness of their wider importance, and this demands good communications and management approaches that recognise local social circumstances and imperatives.

Similarly, the thought behind Agenda 21 is that if all the 'stakeholders' in a community come together and debate what sustainable development means in their own context, and how ecosystem management can meet their needs, wider national and global goals of sustainability will be met - whereas without such local action little will happen except talk. Even in affluent societies, global problems seem remote and unrealistic and local problems dominate.

(b) State of scientific understanding.

Most traditional management practices are small-scale and patchy (for example manipulating nutrient input, grazing regimes, water flows and woodland coppice cycles), with the result that much is known about the effects of such actions on the small-scale and short-term dynamics of ecosystems. It is much more difficult to predict and manage for long-term trends, or to accommodate large-scale perturbations like major floods, droughts or volcanicity. It is also difficult to predict how local management actions will aggregate to a regional or global level. Indeed, chaos theory implies that it may not be possible to predict the large-scale consequences of innumerable small-scale local actions.

There are, however, obvious exceptions. Many large-scale pollution problems have been recognised and addressed through actions that 'cascade down' from the level of a global political agreement to specific action to ban a chemical or clean-up emissions. It is less clear that the scientific knowledge exists to read across into the area of biodiversity conservation or natural resource management and ensure that global and regional goals are implemented by well-designed and efficient measures.

(c) Examples of relevance/application of the principle

There are scarcely any places on Earth that have not been influenced by people, and many apparent wildernesses have been modified a great deal by fire, by human predation or by the deliberate selection of favoured trees within a forest. These modifications all result from actions that began on the small and local scale.

Today, community participation in selecting the goals of ecosystem management is recognised as essential. In the CAMPFIRE project in Zimbabwe, communities have made a deliberate choice to manage and crop local wild species on communal lands. This demonstrates how the larger issues of regional biodiversity and ecosystem pattern can be addressed by modifying how local people interact with their environments, monitoring the consequences and fine-tuning the process.

There are many illustrations of how a global framework has been set, in the form of an international Convention or other instrument, and how national and local actions have been linked to them. In Europe, long-range trans-boundary air pollution was the subject of an international Convention that led, ultimately, to the setting of targets to reduce national sulphur dioxide emissions and the adoption of technical controls on industry. Under the Montreal Protocol, ozone-depleting substances have been phased out, and action taken by many governments and industries to develop and introduce substitutes. Action against pollution is almost always a continuum from local controls binding on products, industries or users to national laws or strategies that set goals and impose standards, and regional or international agreements on the issues to be addressed. The Convention on Biological Diversity is another framework for global action to integrate and inspire local actions. The test is how far the science of ecosystem management is good enough to guide them and make them effective.

(d) Needs for the future

The inter-linkages between local and global processes need to be understood far better. Current efforts to decentralise global climate models are one step in a necessary direction. On-the-ground community action will be almost impossible unless local consequences of global processes can be predicted. Similarly, the implications for large-scale ecosystems of local-scale modifications need more research. Work on population viability analysis and critical areas for survival, especially of large species with extensive home ranges and low population density, also needs to be applied in local-level survival strategies.

If ecosystem management is to be optimised at a local level, there needs to be much more capacity building and empowerment of local communities, so that they are able to care for their own resources without distorting the pressures of poverty and expediency. These communities must be given access to the technology and additional information requirements, to aid them in their actions.

Global conventions such as Ramsar, the Convention on Biological Diversity and the Convention on Migratory Species also need to be implemented locally . They provide ready frameworks for putting local actions into a global perspective - for example when making wise use of a local wetland at a Ramsar site, the contribution to global conservation can be realised. These global-local linkages need to be developed and backed by efficient techniques for ecosystem management.

1.8.2 Ecosystem management must seek to maintain or enhance ecosystem character and functioning at an appropriate level

(a) Rationale

Biological, chemical and physical processes, vital for the maintenance of ecological systems, are being disturbed throughout the world. The speed of change is generally accelerating, resulting in degradation of the structure and functioning of many ecosystems and compromising the options for sustainability. One immediate consequence may be the loss of biodiversity, but there is also a risk of progressive reduction in important environmental services, like groundwater maintenance, water quality, nutrient cycling and food chain support.

The full importance of ecosystem functioning is generally only realised when it is impaired, with results such as decline in water quality and fisheries. Indicators of the status or health of ecosystems are required to assess the need for and to measure the success or failure of management practices. Some believe that the presence or absence of large top-of- the-food chain species provides an indicator of ecosystem health. Others suggest that ecosystem management should look at the entire landscape including both biotic and abiotic components from a functional analysis point of view, recalling that functions operate at different scales.

The relationship between biodiversity and ecosystem functioning is complex. Healthy ecosystems function successfully in their present state. Ecosystem health is not synonymous with ecosystem integrity, which implies not only functional health but an ability to cope with change.

In determining 'ecosystem health', it should be remembered that it is the function of species, not least micro-organisms, together with the wide range of processes occurring within the ecosystem, that needs to be considered. Functional analysis measures different parameters from biodiversity but provides an important link to the wider socio-economic value of ecosystems (Maltby *et al.* 1996).

Ecosystems may be able to lose some characteristic species while still maintaining functions and processes. However, a significant range of species in all major trophic categories is essential for ecosystems to function satisfactorily. Ecosystem structure and use may be matters of social choice, but management should ensure sustainability, and that there is no threat to the integrity of adjacent ecosystems.

(b) State of scientific understanding

There is still much debate about the meaning and significance of concepts such as ecosystem 'health' and integrity (see Box 5). There has been a major swing in ecological research towards emphasis on the functional aspects of ecosystems. However, ecosystem functional relationships are normally complex, and our understanding of them remains relatively simplistic.

The populations of many species have ranges that include source and sink regions. In population sinks, mortality exceeds recruitment but populations are maintained by immigration from sources, where recruitment exceeds mortality. When environmental change affects a population composed in this way, the effects may be far-reaching, and impossible to predict unless the source-sink population structure has been elucidated.

Much current practice of ecosystem management is based on indicator species. For example, setting of "ecologically acceptable minimum flows" and assessment of the success (or failure) of riverine restoration schemes in the USA and UK (and soon also in Norway, New Zealand and France) is achieved by considering the physical habitat requirements of a target species, such as the brown trout (*Salmo trutta*) (Johnson, *et al.* 1996). This species is chosen partly because of its sensitivity to physical conditions, but also partly because the management objective is often a healthy trout fishery.

(c) Examples of relevance/application of the principle

Introductions such as the Nile Perch (*Lapes niloticus*) in Lake Victoria have had a major impact on the biodiversity of other species, particularly endemic Cichlids, but we need to know more about the interactions and effects in functional terms in order to produce sound ecosystem management solutions. Alteration of the population size or balance of fish species, for example by increasing herbivorous species, may be a means of reducing lake eutrophication and improving water quality. Such bio-manipulation has been used in the Norfolk Broads, UK to reduce fish predation on the zooplankton enabling greater phytoplankton consumption and increasing water clarity. This has allowed successful re-establishment by benthic plants (Moss *et al.* 1985) and provides an example of direct ecosystem management. .

Microbial processes such as denitrification may play a vitally important role in providing a nitrate sink function in waterlogged soils at some distance from streams and rivers, but still important in maintaining their water quality. Diversion of flows around such areas removes the functional link between the wetland and the river. It may not be possible to re-establish water quality simply by means of buffer strips paralleling the river (Maltby *et al.* 1997).

(d) Needs for the future

Whilst a growing body of empirical work points increasingly strongly to the crucial role of key species or particular environmental conditions in ecosystem processes, there is a long list of unresolved details in this rapidly expanding field, and rather few guiding theoretical principles.

The functioning of species, especially micro-organisms, and of the factors controlling key processes need to be considered in more detail. Practical indicators and criteria for assessment of ecosystem functioning, health and integrity are required. Decision-makers need guidelines, especially to recognise critical conditions before they become irreversible, so that these can be addressed in ecosystem management while there is time. The goal of ecosystem management then becomes avoiding those impacts that are irreversible.

There is a particular requirement for guidelines on restoring degraded ecosystems, which should examine how best to recover different functions, or combinations of them.

1.8.3 Decision-makers should be guided by appropriate tools derived from science

(a) Rationale

To be effective, ecosystem management must be based upon the laws of nature; fundamental physical, chemical and biological processes will ultimately constrain human desires and actions, not the reverse. The challenge for natural science is to understand the laws which govern the natural environment and the biological dynamics which result. This is an essential basis for guidance to ecosystem managers which will in turn ensure that human activities are sustainable.

There is still a wide gap between those who possess the necessary fundamental scientific knowledge and decision-makers whose actions influence ecosystem management at different scales. Effective tools which incorporate the knowledge of experts are essential for sound ecosystem management. Monitoring of the ecosystems and the effects of management must feed back into future practices and policy decisions. Science should assume a more active role in informing policy-makers of the consequences of various ecosystem management options (including that of no action).

Previous scientific research and theoretical developments have not necessarily produced the most appropriate tools for the manager, and not all the areas of ecological theory sometimes debated in the context of ecosystem conservation are relevant to the actions required. For example, discussions of resilience have a useful function in the theoretical scientific literature, as measures of properties of mathematical models of ecosystems, but it is debatable whether there is currently any way of even starting to use them to define practical conservation objectives. Similarly, while scientific techniques for measuring species extinction rates in ecosystems provide interesting data, the manager needs to know how to conserve the effective functions of ecosystems and their components. Management of habitat fragments may be hindered by uncertainty over the optimal shape and size of these fragments, or even of their long-term viability. In many cases, these fragments are a legacy of past land-use practices and the overall pattern is therefore beyond the control of today's ecosystem managers.

Whilst scientific understanding of ecosystem functioning in itself will not achieve conservation, science is essential as a foundation for sound management decisions. The findings of research must be synthesised, interpreted and made applicable, and they also need to be explained to decision makers and the community at large. Effective scientific argument will influence policy and decisions and reinforce the conservation case, for example by demonstrating the role of protected areas in flood control and water supply. Management itself must feed back into future practices: the regular monitoring of ecosystem conditions is vital, in order to elucidate the impact of management and allow the procedures to be adjusted.

(b) State of scientific understanding.

There is a vast body of ecological and other scientific literature and many published case studies of ecosystem management. The issue is whether these address the key parameters and interactions emerging from modern theoretical ecology. Examples include population viability analysis, ecosystem dynamics, patch dynamics, resilience, the relationship between diversity and ecosystem function, unpredictability of change and chaos theory.

The important question is how far theoretical science is, or can be made to be, relevant to the practical manager of ecosystems on the ground. We are not dealing with a perfect or exact science. The key issues are whether scientific understanding is good enough for decision makers, and whether they will accept the risks that result from gaps in models, procedures or understanding. In the past, this has always been the case, generally because the risks have been unknown or assumed to be minimal. Today, there is less willingness to accept risk: the precautionary principle is an expression of social preference to avoid actions that could cause environmental damage. Scientific research is one means of reducing uncertainty and unpredictability, and narrowing the margins of precaution, and this in turn will lead to more effective and economical management and a more efficient use of natural resources.

(c) Examples of relevance/application of the principle

The concept of 'critical loads' (Battarbee 1995) - the maximum input of a pollutant that an ecosystem can tolerate without undergoing a significant degenerative change - is one practical expression related to tolerance and resilience, and has been taken up and applied in legislation to limit pollution emissions. Critical loads are not an abstract theoretical concept: they are a tangible way of defining resilience and making ecological theory applicable by practical ecosystem managers.

Ecological science has established many aspects of the environmental impacts of acid rain, including interactions with dry deposition on trees and with the soil, the nitrogen cycle, cation-anion balance, and soil buffer zones. Using the results of these data the impacts of ecosystem management practices on biogeochemical cycles can be predicted. For example, the effects of forest felling in acid rain-prone areas can lead to a reduction of pollutant capture, increased mineralisation of N leading to nitrification/acidification, and associated cation production (see Ineson this volume). This knowledge has been developed into practical guidelines for forest managers.

Functional assessment procedures for wetlands have focused on interpreting naturally occurring ecosystem processes in terms of their role in maintaining vital functions and developing indicators that can be used by decision-makers to assess (i) overall functioning of the ecosystem, (ii) a particular function of interest, and (iii) the effect of specific impacts on the system or individual features of it. Generalised techniques are well developed in the United States. In Europe, empirical research across a range of river marginal ecosystems has been based on the functioning of distinct landscape areas termed hydrogeomorphic units (Maltby *et al.* 1996, and Maltby this volume). Functional assessment procedures set out a disciplined basis for decision and the most useful will adopt an expert system type approach, drawing on the best available scientific knowledge. The concept is not specific to wetlands and can be extended to other ecosystems, but it is essential that development is in dialogue with potential users.

Evidence from one reserve established 100 years ago in North America (Drayton and Primack 1995) established that 155 of 422 plant species had been lost due to human actions. As a result management practices were set in motion to reduce and reverse this loss. Reintroduction featured strongly in this programme. Furthermore, science and management interacted in this process by attempting to re-establish species using several different techniques in controlled conditions. This led to an objective comparison of ecosystem management techniques which could feed back into future decisions (see also Primack and Drayton this volume).

(d) Needs for the future

There is a need for good scientific research to be transferred more effectively to ecosystem management, so as to define goals and options better, and enhance cost effectiveness. A 'think tank', perhaps within the IUCN Commission on Ecosystem Management, could usefully bring together suitable representatives from all parties interested in ecosystem management. This could then consider how to optimise the management process to achieve social goals, more precisely define units of management and enhance protection strategies. An exchange of representatives and information from all relevant groups is needed. For example, environmental and conservation scientists should attend industry-orientated meetings, and members of the industrial or business sectors should be present at scientific workshops.

Managers and decision makers need to be trained to a sufficient level to enable them to make informed decisions. In the area of ecosystem functioning, the use of simple equations rather than complex ecosystem models offers the possibility of translating to practical managers measurable indicators and models of processes. Basic research in biogeochemistry and ecology can and should provide the foundation for ecosystem management techniques, but these will only be taken up by managers if they are presented in a suitable form (other than just the scientific literature).

The cost-effectiveness of various approaches to ecosystem management needs to be considered. Rigorous quantitative work should be undertaken to both assess areas of focus and to appraise different management techniques. The development of decision-support systems using 'expert system' approaches to provide managers with the best available translation of scientific knowledge need to be encouraged.

Education must play a key role in integrating conservation science with other thinking. An increased environmental awareness amongst students not specialising in this area might be provided by giving a basic training in ecological sciences to future policy-makers, economists, developers and industrialists. But this training should be positive and practical, emphasising the capacity for ecosystem management rather than uncertainty and dire portents of environmental catastrophe.

1.8.4 Ecosystem managers must act with caution

(a) Rationale

The natural world is highly complex and human understanding is inevitably limited. We lack (and perhaps always will lack) a comprehensive, predictive understanding of the ecological consequences of human activities. We do not understand the full implications of loss of biodiversity or ecosystem functions. Although such information may improve as time passes, it is not available now, and yet important decisions about the exploitation or conservation of ecosystem resources must be made.

Scientists must characterise risk and uncertainty in terms that lay people can understand, differentiating carefully between fact and judgement. Policy makers should not ask for firm conclusions where the facts do not support them, or distort scientific results to suit preferred policy objectives.

Demonstrating that resource use will not be damaging is the responsibility of those who want to undertake that use. The burden of proof needs to be moved from those who want to conserve to those who want to develop resources. Those who use natural resources must recognise the true costs of their actions. In economic jargon, the costs which are currently external and ignored must be internalised. The "polluter pays" principle should be adopted and generalised to cover all resource use. Behaving in a risk-averse manner may avoid losses or unacceptable risks, achieve equity among user groups and between generations, and make use sustainable.

One particular approach consistent with the precautionary principle is the 'safe minimum standard' of conservation. The term originally referred to a conservation strategy applicable to wild species with a critical threshold population size below which it could not recover (minimum viable population). Its aim was to ensure that at least this minimum population size was maintained as long as the cost of doing so was not intolerably high.

(b) State of scientific understanding

The precautionary principle is needed because our knowledge will always be deficient. At present, caution is made more necessary because we have only a limited number of long records from which to determine what impacts will cause irreversible changes. The track record of ecosystem management is not entirely encouraging: unexpected events like the susceptibility of bird populations to DDT and PCBs caught scientists by surprise. It is one objective of science to reduce uncertainty by well-focused and high-quality research: the results should be more cost effective environmental management because the safety margin can be narrowed.

Science is rather more precise when it comes to the consequences of habitat fragmentation. Island biogeographers have demonstrated both theoretically and empirically that island species, especially endemics, are especially likely to be susceptible to the effects of such fragmentation: one third of known, threatened plant species are island endemics.

(c) Examples of relevance/application of principle

Use of the precautionary principle is evident in such international agreements as the Montreal Protocol on substances likely to damage the ozone layer or the Declaration of the Third Ministerial Conference on the North Sea with respect to the dumping of potentially toxic materials (O'Riordan and Cameron 1994). It is also evident in national pollution control measures that demand a substantial safety margin between the maximum allowable concentration of a substance in the environment and the lowest concentration at which effects have been demonstrated.

Impacts on ecosystems are difficult to predict, and adverse effects may not be detected until the functioning of the ecosystem is compromised (see Urbanska, this volume, for a discussion of the ability of ecosystems to undergo stress). Conservation strategies address this situation by demonstrating awareness of the possible consequences of inappropriate management actions or inadequate precautions, and by well-designed monitoring.

To conserve large species at the top of the food chain, small species on which they feed also need to be conserved. If, therefore, these species are the focus for ecosystem management and a Minimum Viable Population of each is defined and safeguarded, other components should also be conserved. This approach avoids lengthy research on all the features of the system. But monitoring is essential, to check that the assumptions are valid.

(d) Needs for the future

There is a need for management to conserve options rather than necessarily preserve ecosystems in their current state. The precautionary approach should be developed in relation to the maintenance of ecosystem functioning. Areas of high diversity and high functional significance (e.g. water resources, fisheries production, erosion control) should be targeted for conservation resources.

When available information is insufficient to make sound decisions, activities must only be authorised if a survey and data collection plan is developed and implemented. Resource use should not increase faster than knowledge of the size and productivity of the resource and its functional role in the ecosystem.

We must embrace adaptive management and not be too prescriptive. Ecosystem managers must be willing and able to amend management policies and practices as often and as quickly as necessary, this must include the willingness to abandon concepts and to admit mistakes. To facilitate this, the management process must be fully accountable to all stakeholders and should continually undergo biological, social and economic appraisal.

1.8.5 A Multi-inter-disciplinary approach is needed

(a) Rationale

Ecosystems have a variety of physical, biological, chemical and human components. Wetland ecosystems, for example, are influenced by a variety of land uses, communities, laws and traditions. Thus, implementation of a truly integrated approach means managing the entire system by integration of ecological, economic and social factors to control the biological, physical and human systems (Wood 1994). It is clear that biological diversity will only be maintained if a holistic approach is taken in land management (the "integrated approach" called for in Agenda 21). This entails the establishment of inter-disciplinary teams including hydrologists, water engineers, biologists, agriculturalists, foresters, physicists, pedologists, planners, human and animal health experts, ecologists, sociologists, demographers and legal experts. Other specialisms should not be excluded where a relevant contribution can be made.

These teams need to address a wide range of sectoral topics including population dynamics, water quality modelling, irrigation, health problems, water plant, fish, herding, legislation, training, and participatory rural appraisal. In addition there will be many cross-sectoral issues, such as development of a geographical information system to overlay various spatial data sets, equitable allocation of resources, development of community participation in resource management, establishment and running of authorities to co-ordinate planning and management.

Conventionally, different disciplines tend to be grouped in separate sectors. For example, hydrologists and foresters belong to different ministries between which there is little formal contact. Each sector often has its own agencies and authorities responsible for development, many of which relate to water issues. Given the inter-connectedness of the ecocomplex, it is critical that inter-sector, interagency collaboration is established to develop the multidisciplinary team. Indeed ecosystem management must accept that no individual or agency can cover all the different aspects involved. The various agencies should collaborate on all aspects of planning and implementation of projects, including problem analysis, project design, data collection, analysis and modelling, policy development, management and enforcement, monitoring and evaluation.

The integrated approach does not stop with the multidisciplinary team. The whole of society needs to be involved. Information, education and dialogue are essential. Economic models need to give full weight to environmental values. The scientific community needs to explain the imperative for conservation and sustainable use. Social changes, including changes in demand, must be foreseen alongside the expected changes in physical and ecological systems.

(b) State of scientific understanding

There are numerous examples of multidisciplinary approaches to *environmental* (but not *ecosystem*) management. These processes have, however, revealed a number of imperfections in scientific methodology. Although many projects and scientific studies have involved partners from various disciplines, the objectives have frequently been uni-sectoral. Economic valuation of natural systems, for example, requires quantification of natural functions, such as the wildlife support and nutrient retention function of a wetland. This is fairly straightforward, but cost-benefit analysis in its classical sense does not really address intangibles such as natural beauty or the sacredness of some forest groves. There are relatively few trans-discipline methods which allow rigorous comparisons of options.

(c) Examples of relevance/application of principle

The formulation of national conservation strategies, national sustainable development strategies and other cross-sectoral plans, described in preceding sections, illustrate this process. Essentially, the approach has been to assemble multidisciplinary teams, working to steering committees involving all the principal sectoral interest in government, and to expose the plans to debate at local and regional community level. Environmental impact assessment also involves inputs from various disciplines.

An economic evaluation of the Hadejia-Nguru wetlands in Northern Nigeria (Barbier *et al.* 1997) involved collaboration of ecologists, hydrologists and economists. It demonstrated that the value of the traditional agriculture and fisheries in the wetlands exceeded that of an intensive rice irrigation scheme upstream to which water had been diverted that had led to reducing flooding in the wetlands.

Development of a management plan for the restoration of the Waza-Logone floodplain in Northern Cameroon, which includes the Waza National Park protected area, has involved many specialists including sociologists, ecologists, hydrologists working with local communities, local and regional governments, private organisations and Park authorities. They sought common objectives for land use planning and resource allocation (particularly water allocation from Lake Maga, formed by a barrage which spans the floodplain) (Wesselink *et al.*. 1996).

(d) Needs for the future

The primary need is to develop effective mechanisms which enables different disciplines to interact and for different scientific approaches to be integrated. It is probably not necessary to alter any sectoral discipline so much as to enhance ways of interrelating them in the social decision-making process. Some techniques for community decision-making, conflict resolution and the like offer prospects here. It is clearly important to enhance cost effectiveness, using economics and commerce as well as science.

1.9 Applying the principles

If these ten principles are to be applied, a number of conditions must be met. Would-be ecosystem managers are advised to:

1. place the management in a social context - defining what the ecosystem or unit is managed for (including aesthetics), recognising that local concerns and need often dominate but that global context must be recognised;
2. incorporate risk assessment and cost benefit analysis in choosing between options.
3. consider who is to do the management and how management decisions fit in the political process;
4. place the management unit in a spatial context - addressing issues on the appropriate scale, within the local landscape and the wider regional or global context;
5. also consider the appropriate time frame for management;
6. define the key parameters for functional assessment and management and also for monitoring, including key environmental parameters, key species and socio-economic factors;
7. model interactions and driving variables. Relate these to species ecology and distributional units and define probable patterns of species composing the system under various scenarios;
8. adopt adaptive management, expecting, accommodating and managing for change rather than obstructing it. Evaluate probabilities and risks, and incorporate flexibility (precautionary principles). Management decisions should include a safety factor to allow for the fact that knowledge is limited and institutions are imperfect; and
9. see restoration ecology as the art of the possible and elaborate the techniques required to bring about effective ecosystem restoration.

1.9.1 Some Over-arching Questions: Summary of the 1996 World Conservation Congress workshop discussion of the Sibthorp Seminar

Martin Holdgate

The Workshop addressed some fundamental questions. Perhaps the biggest is whether the ecosystem approach is really valid for conservation management? A second linked question is whether the same principles can be applied to management on land and in the sea? The third big question is what we really mean by management? The fact is that we manage human actions that in turn have an impact on ecosystems. We use the status of key, prominent or valued species as indicators of the state of an ecosystem and we judge the 'success' or 'failure' of our efforts by their responses. We do not monitor ecosystems as a whole.

Our thinking advances through intellectual stages which are valuable as much for the challenges they provide as for the solutions they generate. Early mathematical models of ecosystems posed methodological challenges which led to more useful second-generation models. Ecological economics has developed in response to another major challenge - how to 'value' ecosystems and their components and reflect them in economic models and judgements. Ecosystem management will move forward by a similar process of challenge and response. If social choices are the paramount determinants of action, maybe we will have to converge with the social sciences. The concept and practice of Primary Environmental Care is one example of a blending of social and natural sciences as a basis for policy.

It is clear that we face many professional challenges. One is over the accuracy of data and adequacy of sampling. Sound social judgements are difficult, and even dangerous, if they are based on extrapolation from shaky data. Another major challenge is over the significance of functional duplication in and between ecosystems, and the extent to which substitution of species and systems is acceptable. We need fundamental research to fill gaps in knowledge, as a basis for application.

The Ten 'Principles'

Five conclusions emerged from the discussion of the ten 'principles',

First, and as a general point, it was agreed that they do constitute a useful starting point, but they clearly need adaptation to more specific national and local conditions, especially because management at the local scale is critical. They may need subdivision, because not all are equally important in all situations, while at some local scales, sub-sets of a principle may be highly relevant.

Second, there was a broad consensus that ecosystems must be managed in a social context, by a process of adjustment of human impacts. There must be management plans to articulate social choices. But there must be good communication to secure social consensus for proposed actions. Moreover, we must allow for the fact that the social context is changing continually, and opportunistic responses are necessary. And the social context must not be too narrowly defined. Local societies are subject to many major external influences.

Third, we must remember that ecosystem integrity is different from ecosystem health and sustained productivity. We need to maintain diversity and resilience as a basis for alternative futures. The workshop saw this demonstration in relation both to steppes and wetlands.

Fourth, at the scientific as well as the social level, management is the art of the possible. Science can provide guidelines - and also highlight areas of uncertainty, and possible mistakes to avoid. But the spatial scale of management needs to be chosen with sensitivity to the dynamics of the system itself, and to the scales of movement of component species, including migrants. On land, protected areas, safeguarded by buffer zones and habitats, and linked by corridors, can be defined relatively exactly and their management will need to have very different spatial dimensions, and both boundaries and management prescriptions will be less capable of relationship to specific geographical features. Managers should always proceed with caution.

Fifth, the inevitability of change is accepted, but we need to define alternative scenarios with care, recognising that a single change may allow several social choices. Options must be kept open - but some clear pathways must be signposted NO ENTRY.

Pointers for the Commission on Ecosystem Management

Four particular ideas emerged for consideration by the Commission as it develops its programme of work.

First, it should examine the scope for restoration ecology and note the inevitability of species invasions, as biogeographical barriers are broken down. Efforts to prevent invasions need to be judged against a sharp test of practicability - and the implications of unavoidable invasions for ecosystem management need to be assessed.

Second, it should recognise that interdisciplinary action can confer 'hybrid vigour' on the resulting management practices.

Third, it should seek the further development of the topics and principles presented to, and discussed in the workshop and the present volume. But there needs to be cross-linkages with other Commissions, to which these concepts are also highly relevant. Like the former Commission on Ecology, CEM should be a basic scientific powerhouse for IUCN.

Fourth, it should provide IUCN with guidance on the Union's inputs to external bodies like the Convention on Biological Diversity, the Framework Convention on Climate Change, the Convention to Combat Desertification and Drought, and the Intergovernmental Panel on Forests.

1.10 Case Study 1. Ecosystem-Based Management: A Marine Perspective

Tundi Agardy
WWF - US

Marine conservationists have an invaluable perspective to bring to the table in discussions of what ecosystem-based management means, and what it practically entails. This perspective has largely been ignored in past meetings of minds, in part because marine conservation is itself a marginalised activity. If, however, we will ever succeed in operationalising ecosystem management in any context or biome, the marine point of view must be brought to bear on the problem. The following four rather provocative statements capture the essence of the debate as far as marine managers and conservationists are concerned, and point to the significant differences facing those who seek to conserve the seas and those who would do so on land.

1. Both the theory and the practical application of *marine* ecosystem-based management concepts are significantly different from terrestrial ecosystem based management. Whether this is a difference in kind or scale is debatable, but such debate is largely irrelevant. We need merely to acknowledge the differences and adapt our thinking and methods to each system as appropriate.

The following factors differentiate marine ecosystems;

- The dynamic, fluid nature of marine ecosystems differs from the relatively fixed terrestrial environment. Many components of the marine system are in constant motion, and in some ecosystems - pelagic ones being the prime example - the entire system is dynamic in the three dimensional plane. Coral reef and benthic ecosystems are the exception, and are essentially terrestrial analogues but for the inevitably greater connectivity that being in a fluid rather than atmospheric medium provides. This movement makes it difficult to detect boundaries; as a consequence ecosystem managers will have to guess at boundary conditions and sometimes employ artificial boundaries to initiate ecosystem-scale management.

- There is a far greater degree of non-linearity in marine ecosystems and the food webs that these ecosystems support are (relative to terrestrial) largely unstructured. Therefore consequences of management actions are not always immediately apparent and making predictions about such consequences of management action is a difficult business with notable risks.

- The scales at which ecosystems on land and in the sea operate are vastly different. In marine systems, spatial scales are quite large while temporal scales are fine. This difference is significant for management, because the consequences of a management activity or environmental change may be detected immediately but are unlikely to be confined to a small space. As a result, ecosystem-based management requires co-ordinated action over wide spatial scales, even more than is necessary in similar attempts to practice management on appropriate scales requires on land.

- Compared to terrestrial ecosystems very little is known of marine ecosystem ecology in general and its functions and processes in particular. A quote from a former poet laureate of the United States captures this uncertainty :"The oceans are in everything we do - yet we remain on the shores of what we know." (R. Wilbur). Because this is the case and because we cannot afford to wait for more complete knowledge, it behoves planners and managers to try and incorporate traditional knowledge into our more western-based analyses and assessments. This will require that we openly legitimise traditional knowledge and think of ways to make it more "rigorous" - or at least use traditional knowledge to help orient scientific endeavour so that it fills gaps in our scientific knowledge.

2. Contrary to what is being implied in the title of this symposium and many others like it in the past, we do not manage ecosystems, we manage, or attempt to manage, human beings and their impacts on those ecosystems. Ecosystem-based management is by regulation of humans and their impact not directly of the ecosystems themselves. This is particularly true in the marine realm, where we cannot fence off ecosystems and keep them isolated from exogenous impacts.

- Marine ecosystems with their geographically immense connectivity cannot be thought of as 'closed' ; therefore ecosystem managers cannot hope to fully control all inputs and outputs in what are essentially open systems.

- 'Land'-use must be considered when practising true ecosystem-based marine management. The large spatial scales, dynamic nature, and the ecological connections that make marine systems "open" requires us to incorporate watershed management and land use planning in the coastal plains. This in turn leads us to consider virtually all the earth's surface as part of the marine realm - creating what I call (and fully admit to sharing) "the supreme arrogance of the marine conservationist".

3. Marine ecosystem planners and managers have much to teach their terrestrial counterparts. We are more likely to be innovative in our approach to ecosystem-based management, in part because we are not confined by rigid and established paradigms.

- Opportunities for practising management on true ecosystem scales are thus probably greater in marine areas than on land due to the lack of confining preconceived notions and history. In the developing world especially, this freedom to be innovative and unconventional means that new models are emerging and are being tested all the time.

4. Despite this optimistic and ambitious stance, marine ecosystem management in its purest sense is not yet practised anywhere. While some advances have been made in the delimitation of large scale biophysical units called Large Marine Ecosystems (LMEs) world-wide, these ecoregions do not explicitly include the landward portion of coastal ecosystems.. Such LMEs are largely based on neritic or pelagic fisheries resources and need to be expanded and refined before they can be considered viable management units for ecosystem-based conservation.Some progress has also been made in integrating coastal management activities through Integrated Coastal Management (ICM). However, the boundaries of such integration are usually political - and here the emphasis is typically on the land side of the coastal zone, in contrast to the above situation with LMEs.Marine and coastal protected areas (MPAs) do, in some cases, provide examples of how to integrate management, bring the natural and social sciences together to help frame objectives and design solutions (together with the stakeholders themselves). Areas can then be zoned for use according to their vulnerability and ecological importance. However scaling-up to the scales truly appropriate to ecosystems still remains a problem.

Although some bits and pieces of coastal ecosystems are being effectively conserved to meet socially-derived objectives, the sorts of initiatives mentioned above (LME, ICM and MPAs) are themselves not being brought together to constitute *true* ecosystem based management.The United Nations Environment Programme's Regional Seas Programme provides one potential framework to achieve true ecosystem based management by incorporating MPA, ICM and LME management into single, ecosystem-based initiatives, but for various reasons the progress with this program has been fitful.

Perhaps because of our "supreme arrogance", we see these shortfalls not as a disappointments but as potential opportunities for ecosystem-based management. We fully expect to capitalise on these opportunities, and will hope to provide the conservation world leadership by demonstration in the years to come.

1.11 Case Study 2. Establishment and Management of Three Protected Areas in Central America

Gerardo Budowski

The 10 'Sibthorp Principles' for ecosystem management were examined for three Central American protected areas, two of them in Costa Rica with over 30 years of existence and one in Guatemala yet to be created. The areas were chosen mainly because I have been following their development closely.

Guanacaste Conservation Area, Costa Rica

This is a conglomerate of various areas in Northwest Costa Rica, the Santa Rosa National Park covering 37 217 terrestrial ha and 78 000 maritime ha, adjoining the Guanacaste National Park of 32 512 ha (see figure 1.1a). It began 30 years ago as a small park around an historical house called "La Casona", today an historical museum, where the Costa Rican battle against the filibuster William Walker was fought successfully over 140 years ago and gradually increased in size. One of its outstanding natural features such as the arrival of tens of thousands of sea turtles on one of the beaches, was only discovered later. The park was created in the late sixties and its first management plan was drafted in the Department I headed then.

The Monteverde Cloud Forest Reserve, Costa Rica

This private reserve was established about 50 years ago by a Quaker colony devoted to milk and cheese production covering a small area Northwest of San José at 1000 to 1700 m elevation on the continental divide (see figure 1.1a). Its original purpose was to protect an important watershed but it has gradually expanded and developed into an important ecotourism centre. Also around the village of Santa Elena it was partly populated by expatriate artists, biologists, writers and other foreigners looking for a place to live close to nature. The core area is administrated by a private non profit organisation, the well known Tropical Science Centre in San José and the whole protected area has considerably expanded as a result of world interest and funding to purchase nearby non disturbed forest areas.

Both the Guanacaste conservation area and the private Monteverde Cloud Forest Reserve and surrounding protected areas, have considerably increased in size and many lessons have been learned during that period.

The Laj Chimel proposed "Ecological Reserve for Peace" in Central Guatemala.

This proposed cloud forest covering about 800 ha but linked as part of a corridor to other mountain forests, is the brainchild of Rigoberta Menchú, the 1991 Nobel Prize laureate for Peace, a Mayan Indian of the Quiché group, about 20 Km North of Uspantán, where Rigoberta was born (see figure 1.1b). The proposed reserve belongs to an Indian community of which Rigoberta is a prominent member but plans are being drawn to convert this magnificent forest into a centre for visitation and meditation, with activities promoting communion with nature and peaceful relationship. Presently small scale extraction of timber is taking place to supply local needs. The major threat is shifting cultivation which encroaches on the forest, although on a small scale. The area was particularly ravaged in the past by warfare between Indians and the Guatemala army.

I completed the first short survey of its flora and fauna and steps are being undertaken to manage it for conservation and careful ecotourism while benefiting the local Indian communities living close to the proposed reserve.

A field test of the 'Sibthorp Principles' was made on the basis of the experience in these three areas. Nine of the 10 principles were found to be useful but very broad. Hence additional subdivisions for most of them are proposed.

How do the 10 'Sibthorp Principles' fare in the light of the three case studies?

1. Management objectives are a matter of social choice

This is a wide embracing principle for which four subdivisions apply.

a) Planning and management are highly influenced by the urgency to act. In the case of Monteverde and Guanacaste their creation as protected areas was triggered by a number of threats, the main ones being deforestation and changes of land use after clearing. Hence measures leading to their declaration were immediately implemented. However, it is much easier to have a decree towards declaration approved rather than setting up to proper administrative structures. These came only several years later.

b) Opportunism is paramount. It helped considerably to get action when very important persons are behind the drive towards adequate management. This was certainly the case in Guanacaste and Laj Chimel. In Guanacaste the historic importance of the place made it an attractive political issue for the Government. It also helped that shortly before the time of the creation of the Santa Rosa National Park, a good part of the land and the "casona" were purchased by the very disliked military dictator of neighbouring Nicaragua and this outraged the Costa Rican Government and the people in general. In the case of Laj Chimel in Guatemala, the fact that its main mover was the 1991 Nobel laureate Rigoberta Menchú brought immediate responses and support from a Swiss Foundation as well as from UNESCO.

c) Management plans (even imperfect ones) are indispensable. They should underline a few highly visible and impacting objectives. A management plan (or work plan), and hopefully an operational plan, possibly on an annual basis, fixes goals, objectives and strategies, that cannot easily be ignored. This proved to be extremely important for Guanacaste since a part of it was expropriated from a large land-owner. With the subsequent change of government and possibly to return favours, the incoming President intended to give back this piece of land to its former owner. A public debate resulted; the Vice Minister, a well known conservationist, resigned in disgust. The ensuing uproar and the discussions centred very much on the goals and objectives of the management plan, many of which had also been incorporated in the formal decree creating the park. The management plan was prepared by the protected areas division (headed by Dr Kenton Miller), in the Department of Natural Renewable Resources of the Inter-American Institute of Agricultural Sciences, Turrialba, Costa Rica, which I headed at that time. Dr Miller later became Director General of IUCN.This was a battle won by conservationists and the fact that there was a management plan made the difference. In the case of the mountainous Monteverde the first management plan clearly indicated the importance of watershed protection, with the benefits for a number of settlements downstream including of course the Quaker settlements.

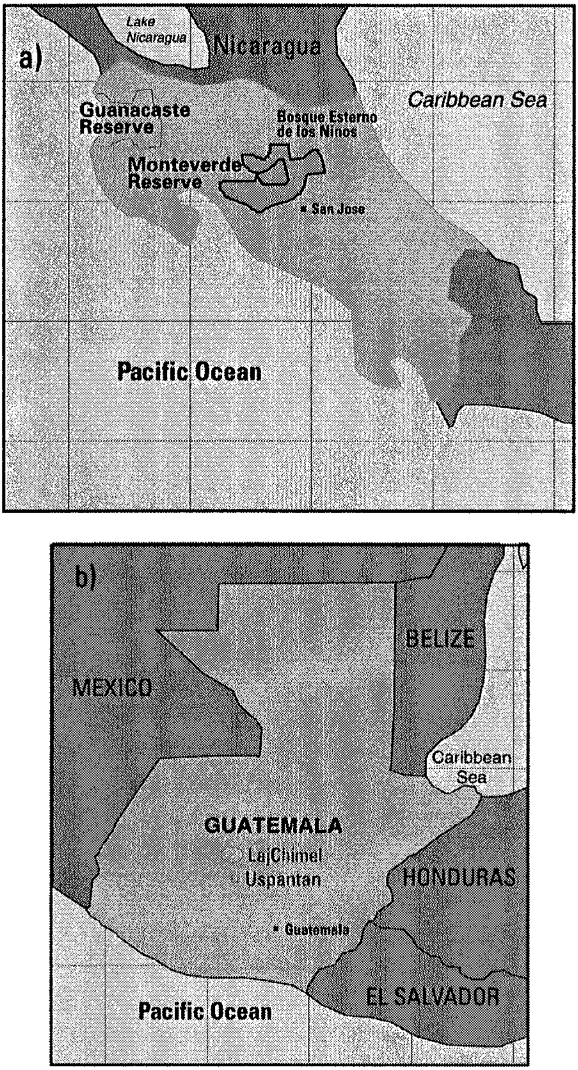

Figure 1.1. Location of Guanacaste, Monteverde (a) and Laj Chimel (b) Reserves.

d) Funding from "outside" greatly helps to get management objectives accepted and implemented. In all 3 cases external funding was available. In the case of Monteverde, support came from WWF. When Monteverde later expanded, many additional sponsors intervened favourably. The best known case is an additional surface of about 8 000 hectares called "Bosque eterno de los niños" (eternal forest of the children) with money collected mainly from thousands of school children in Sweden. (A delegation of these children later visited Costa Rica and "their" forest. I had the privilege of showing them around at the University for Peace). For the Guanacaste Conservation Area, funds were also made available initially by WWF. Later different countries and institutions co-operated notably because this area became the working ground of Dr. Daniel Janzen, a well known tropical biologist, who was able to rally much support from donors throughout the world, notably from Sweden. Outside support be it for funds, interventions or other actions supporting local conservation efforts is very important in Costa Rica. They enhance the sense of pride of the country and are well advertised in the local media.

2. Ecosystems must be managed in a human context

Creating protected areas often creates conflicts since traditional rights of settlers located nearby or even within the new protected area are often drastically curtailed because of the new protected status. The following additional subdivisions were found useful.

a) Conflicts with nearby population groups demand early attention and often very different approaches and compromises, if their resolution is to be successful. A new conservation area usually implies losses and benefits for local populations but every effort should be made to reduce the former to a minimum and greatly increase the latter on the basis of adequate compensations. This was achieved in Guanacaste and Monteverde and is expected to be the case in Laj Chimel. Ecotourism and scientific as well as educational activities, involving job opportunities are at the core and they certainly have proved their value in Costa Rica as many examples show. In the case of Laj Chimel, economic benefits for the nearby Mayan communities are foreseen in the planning stage, compensating for shifting cultivation and extraction of timber.

b) Buffer zone management is important, with tangible benefits for local communities. This implies incentives for rural land use schemes activities that are as compatible as possible with the conservation objectives of the protected areas. Education activities targeted to the needs of the local populations have been successfully carried out in Guanacaste and Monteverde and have proven to be invaluable. They are planned for Laj Chimel.

3. Ecosystems must be managed within natural limits

This rather obvious principle is easier said than applied but it is certainly valid as a long term objective. The problem is of course the difficulty to achieve natural limits by incorporating areas which have legal property rights. This implies a number of financial and administrative constraints. The basic ecological principle however is clear and the sooner natural limits are achieved, the lesser it will cost in a long run and the easier -and scientifically much more viable it will be to manage. However for protected areas the concept of buffer zones and corridors must also be incorporated hence the following additional points:

a) Achieving natural limits often becomes the "art of the possible". It depends very much on a number of factors such as political will, external and local funds and other successful interventions. Often a compromise must be sought.

b) Animal migrations, when better known, often imply increasing the limits. This proved to be the case in both Guanacaste and Monteverde because of formerly unknown seasonal migrations of birds and butterflies (both altitudinally as well as from dry to wet and vice-versa, usually linked with food supply). This required changes of boundaries which fortunately could be achieved.

c) Buffer zones and the concept of corridors influence the limits. Buffer zones, particularly when forested, even rows of trees or certain agroforestry combinations, can be useful for corridors. They help achieving the objectives of procuring natural limits and improving the movement of animals.

4. Management must recognise that change is inevitable

For managers of protected areas, that often means that success is linked with the correct insight of changes likely to take place, more than simply "reacting" to change. A clear case is the strong surge of ecotourism and its biologic, economic and social consequences in Costa Rica, likely to be followed in Guatemala. This was foreseen in all 3 cases but reacting to new threats proved unavoidable, hence a certain capacity to adjust to new conditions is indispensable.

a) Science can help by displaying and analysing different scenarios in time with high, medium and low projections. This allows changes in priorities and helps decision making. The use of certain tools (controlled fires, limited grazing were experimented in Guanacaste) help to design new management tools.

b) Monitoring changes in human perceptions is indispensable. Both Monteverde and Guanacaste were clear cases of initial conflicts with some groups of neighbours. Fortunately their attitudes gradually turned into approval and co-operation over the last decades. Poaching, use of destructive fires, fence breaking to allow illegal grazing, have all been threats that had to be gradually overcome. The ever increasing number of ecotourists is presently a formidable challenge which needs to be considered in future planning, not only concerning carrying capacity but also in relation to cultural implications.

5. Ecosystem management must not be restricted to protected areas.

This point is also illustrated in the former principles and their subdivisions, particularly in relation to buffer zones, corridors and natural limits, as well as educational programmes within and close to protected area. It may be noted that managers of protected areas are often poorly trained to face this challenge, particularly the relation to the growing industry of ecotourism, including for example the construction of hotels or lodges in the vicinity of protected areas. This was clearly the case in some of the two areas of Costa Rica but is being handled with increasing success.

6. Think globally, act locally

It may be noted here that the great interest generated inside and outside Costa Rica towards these protected areas is likely to affect relationships with populations adjacent to the two protected areas. Costa Rica is fortunate in receiving a wide and generally very favourable publicity through the foreign media for its outstanding system of protected areas. It has certainly favoured local managers and their attitude concerning greater global concern. Every Costa Rican nature guide or park manager is clearly aware of the relatively high percentage of the worlds biodiversity that is protected through its system of protected areas. This applies equally to the Laj Chimel reserve in Guatemala, which is likely to have global repercussions.

7. Ecosystem management must seek to maintain or enhance ecosystem character and function at an appropriate level for social choice.

The following subdivision may be added:

a) It is very desirable to make decisions where management options are kept open, that is, allowing flexibility for future decision making.

b) Such concepts as resilience, carrying capacity and critical thresholds must be clearly identified, understood and carefully monitored.

c) Science can help by describing and evaluating the beneficial functions of natural ecosystems, as well as the services provided to mankind both at present as well as in a near and long term future.

8. Decision makers should be guided by appropriate tools.

There are of course many different tools depending on a variety of conditions. In the three cases, two complementary easily understood subdivisions can be added, as to the role of scientists in this endeavour.

a) Scientists should help devising guidelines for decision makers. Such guidelines should not only indicate what needs to be done and why, but also, and possibly even more importantly, what are the mistakes that must be avoided and why.

b) Scientists should play a more active role in providing scientific data to relevant NGO's. It is important that the increasing number of responsible NGO's involved in one way or another with ecosystem management, be supported in their task by sound scientific data that provide them with the necessary credibility -and sort them out from highly emotional groups.

9. Managers must act with caution.

This principle was found to be the least useful, perhaps because it is too obvious. It is implicit in most of the other principles, particularly in subdivision a) of principle 7.

10. A multi-interdisciplinary approach is needed.

Again this is an obvious principle. It may be added that at this moment on the basis of the three case studies examined, the addition of social sciences is possibly the weakest of the three disciplines that shape sustainability – namely environmental (including biological) economic and social aspects.

On the economic side the most urgent need is the just appraisal and valuation of the externalities provided by natural ecosystems -the relatively new science of "ecological economics".

Finally such aspects as public relations, fund raising, good communication, providing awards for or publicising success stories as well as different education and training activities, have proven to be extremely successful in the Costa Rican case -and are likely to be so in Laj Chimel.

Conclusion

The 'Sibthorp Principles' have proven to be useful in the three cases examined. However, because of their broad nature, it is more appropriate to formulate a number of subdivisions when they are to be applied to specific situations.

2. CHALLENGES TO TRADITIONAL ECOLOGICAL THEORY AND NEW THINKING TO UNDERPIN ECOSYSTEM MANAGEMENT

2.1 What to Conserve - Species or Ecosystems?

John H. Lawton

Introduction

My brief on being asked to open the Sibthorp Seminar was 'to provide a critical and provocative overview of past paradigms and their application in traditional conservation', which I take to mean species-orientated conservation, and to ask how the new paradigm, based on ecosystem theory, 'can be translated into practical field conservation?' I will argue that the apparent distinction made in this brief between species conservation and ecosystem conservation is artificial and unhelpful, because it misinterprets and confuses circumstances, time scales and the inevitable interplay that exists between species and ecosystems (Lawton 1994; Jones and Lawton 1995). It is trite but true that you cannot have species without ecosystems, nor ecosystems without species. I will illustrate the interplay between species and ecosystems as the essay progresses.

A second major aim of this introduction to the Seminar is to dispose of the myth that there is some pristine, Garden-of-Eden-like state for all ecosystems, from which they have been disturbed by human actions. Yes, most of the world's ecosystems have suffered from human impacts, some to the point of total destruction. But no, there never was a Garden of Eden that we should strive to recreate. Ecosystems change continuously at all time-scales, and the further back in time we go, the more different they become. Deciding what we want to conserve is not, therefore, a strictly scientific question, because there is no bench-mark virgin state that we can refer back to. I will justify this position as the essay develops.

My third and final aim is to argue forcefully that there are areas of ecological theory that are at best irrelevant and at worst a wasteful distraction for the pressing conservation tasks that confront us. (This is not a diatribe against all theory. Some, as I hope to show, is very useful. But quite a lot is neither useful, nor informative in the present context). My qualifications for arguing this point are, on the one hand as a theoretical ecological scientist, and on the other as Chairman of Council of the Royal Society for the Protection of Birds (RSPB), the largest, voluntary wildlife conservation body in Europe, and one with an impressive, practical record in delivering conservation at the sharp end.

Take one example. My brief asked me to discuss 'resilience' as something that conservation biologists ought to be concerned with. I cannot imagine anything less useful for conservation to waste time and money on. There are at least two quite different definitions of resilience in the theoretical ecological literature. One is the rate at which model systems return to equilibrium after a disturbance (Pimm 1984), measured for example, as the reciprocal of the real part of the dominant eigenvalue of the community matrix (e.g. Pimm and Lawton 1977). The second definition seeks to specify how far a system can be disturbed before it collapses and is no longer capable of recovery (Holling 1973). Both notions have a useful life in the theoretical scientific literature; each measures a quite different property of mathematical models of ecosystems (so that one cannot talk glibly of 'resilience' - one has to define it precisely); and I currently know no way of even starting to use them to define practical conservation objectives. The gap between theory and practice here is vast, and despite exhortations to bridge it (Arrow *et al.* 1995), I believe that the time and money available to conservation biologist would be better spent elsewhere.

It is, of course, possible to redefine resilience as something more tangible, and to study that. The second definition overlaps with the concept of 'Critical Loads' for example (Battarbee 1995), with important implications for ecosystem management. But critical loads are not an abstract theoretical concept, and they are not the same as resilience. The issues confronting conservation science are too pressing, and too important, to become bogged down in a game of words.

Finally, to clear the remaining decks, I am going to focus on the conservation of species within ecosystems, but not the conservation of genetic resources within species, nor (except in passing) the maintenance of 'ecosystem function' as a conservation goal. I am also going to concentrate on terrestrial ecosystems, for two reasons. Space precludes a proper discussion of both terrestrial and marine systems, and my own experience and expertise is decidedly dry. This is not to say that marine systems are not important. In many ways, the science that deals with the conservation of renewable marine resources (fish, whales, etc.) is considerably more advanced than terrestrial conservation in its thinking. The 'Large Marine Ecosystem' concept applied to the conservation of fish stocks (World Resources Institute 1994, pp.193-4) is a model blend of species and ecosystem conservation.

The scale of the problem

Conservation biology is not a cosy, academic game. What we do matters, and the scale of the problem is daunting (Morris 1995). Although they will be familiar to most readers, it is worth rehearsing some basic facts:

i) Very crudely, natural 'background' extinction rates in the fossil record are of the order of one species a year (Lawton and May 1995). Over the last few hundred years, human beings have steadily increased this rate (e.g. Steadman 1995), by perhaps 2 or 3 orders of magnitude. Looking ahead, and by a variety of calculations, it is difficult to avoid the conclusion that extinction rates over the next 100 years will be at least 4 orders of magnitude faster than the background rate in the fossil record (Lawton and May 1995; see also Pimm *et al.* 1995). This rate of extinction is unique in the history of life on Earth.

ii) It is not difficult to see why. About half of the ice-free terrestrial surface of the Earth has now been transformed, is managed, or is used by humans (Vitousek 1992, 1994); nearly 40% of the potential net primary productivity of terrestrial ecosystems is used or dominated by humans, or foregone as a result of land use change (Vitousek *et al.* 1986); and between 24-35% of the net primary production of marine upwelling and shelf systems (where the majority of the world's fisheries are concentrated) now ends up in fish caught by people (Pauly and Christensen 1995). No single species has ever dominated the world as Homo sapiens now does. The odds stacked against many of the 10 million or so species with which we share the planet are overwhelming.

It is for these stark reasons that ecosystem conservation, rather than species conservation is seen as the only viable option. Fifty years ago, species conservation alone made sense, because there were still natural and semi-natural ecosystems in abundance. As human pressures mount, purely species-orientated conservation is no longer viable, except in zoos and botanic gardens; there can be no wild pandas without bamboo forests. But equally, ecosystem conservation alone is also not enough; species conservation still matters, as I will now show.

The problem of surviving fragments

Ecosystem conservation may deliver unlogged forests, clean water and whatever other 'ecosystem services' (Ehrlich and Wilson 1991) we regard as valuable, but still fail to conserve pandas, or myriads of other threatened species, if the surviving fragments of natural and semi-natural habitats are too small, too isolated, or both, however well they are protected. Two related bodies of ecological theory (island biogeography and metapopulation dynamics) predict loss of species from small and/or isolated habitat patches (Diamond and May 1981; Hanski *et al.* 1995). (These are both, incidentally, useful theories because they can be made operational, and their relevant parameters measured in the real world; 'resilience' cannot be made operational in the same way, and is therefore too abstract to be useful.) Data do indeed confirm that species tend to disappear on small, and/or isolated habitat patches (Thomas *et al.* 1992; Thomas and Jones 1993; Åberg *et al.* 1995; Hanski *et al.* 1995); the ecosystems survive, but not a proportion of their characteristic species. Indeed the message from population biology is even more gloomy. Extinction of populations on surviving habitat patches takes time (e.g. Tilman *et al.* 1994; Hanski *et al.* 1996), so that current levels of diversity on many habitat remnants may be unsustainable in the long run. Many species, in Dan Janzen's graphic phrase are 'living dead'. We must expect to see even greater simplification of many ecosystems in the future, even in protected areas.

At least two important conservation messages flow from these simple observations. The first we can refer to as the 'nature reserves for missing species' problem; the second is concerned with the role of species in ecosystem processes. I will say something, briefly, about both.

Nature reserves for missing species

From a species-orientated perspective, the vacant habitat patches (ecosystem remnants), lacking one or more characteristic species are not irrelevant for conservation. Currently vacant patches may be recolonised at some time in the future, and be essential for the maintenance of a metapopulation in the larger landscape (Hanski *et al.* 1995, 1996 and references therein). Moreover, if serious efforts are made to restore and recreate ecosystems (as RSPB and other conservation bodies are currently doing in the UK with heathlands and wetlands, and which is also going on in many other parts of the world), then currently vacant patches may form vital stepping stones for recolonisation of recreated ecosystems. It takes a mighty leap of conservation faith to spend time and money conserving ecosystem fragments from which important species have disappeared, and to spend even more time and money recreating ecosystems for species that are not there yet. But all the evidence suggests that maintaining and recreating a network of appropriate habitats (ecosystem conservation) will benefit species conservation in the longer term.

Species and ecosystem processes

The perversity of trying to drive a wedge between species conservation and ecosystem conservation is made all the more clear by viewing the missing species problem from an ecosystem perspective. Ecosystems may be able to lose some characteristic species and still 'function' satisfactorily. But the local extinction of other species may have major consequences for ecosystem processes on surviving patches. A growing body of empirical work points increasingly strongly to the fact that species matter for ecosystem processes - however, which species, and how many remains open to debate (Lawton and Brown 1993; Lawton 1994; Naeem *et al.* 1994, 1995; Tilman and Downing 1994; Jones and Lawton 1995; Tilman 1996; Tilman *et al.* 1996). There is a long list of unresolved details in this rapidly expanding field, and rather few (simple!) guiding theoretical principles. But in the present context the message is reasonably clear. Ecosystem conservation and species conservation are inextricably intertwined. You cannot have one without the other.

Minimum viable populations and 1/f noise

The fact that small, isolated populations are highly vulnerable to extinction, and that even large, isolated populations are at risk in the long term has lead to the development of population models designed to predict 'minimum viable populations' or MVPs (e.g. Soulé 1987). There is now a very large literature on the subject, which I do not propose to review here. The reason that any isolated population (or indeed metapopulation - Hanski *et al.* 1996) however large, is vulnerable to extinction is because all populations fluctuate in response to density independent perturbations - so called 'environmental noise'. Almost without exception, models designed to predict MVPs assume that this noise is random 'white noise' (by analogy with white light) in which all frequencies have equal power; static on the radio is white noise. It seems increasingly likely that real environmental noise is nothing like this. Rather it is 'coloured noise', often 'red noise' in which, as in red light, longer wavelengths predominate (Steele 1985; Pimm and Redfern 1988; Halley 1996). More formally, real environmental noise is 1/f noise, in which the magnitude or the power of the noise is inversely proportional to its frequency, f. Little perturbations happen often, disasters very rarely, and minor catastrophes somewhere in between. How often have we heard conservation managers complain that they were well on the way to saving endangered species x when they had an 'abnormal' or 'unexpected' event. There is no such thing; there is only 1/f noise.

This insight has two consequences. First, most predicted MVPs are too optimistic, with profound and depressing consequences for future extinctions. Second, as I pointed out in the Introduction, there is no fixed, virgin state for any ecosystem. 1/f noise means that the longer we observe populations or ecosystems, the more they will change. There is no such thing as 'normal variation' (with a characteristic mean and variance); 1/f noise implies that all ecological systems fluctuate within wider and wider limits, over longer and longer time periods. This has major implications for how we decide what to conserve, to which I return below.

If we are going to do MVPs, do them on large-bodied species and species high in the food chain. One more point is worth making. MVPs, done properly, can be valuable conservation tools. But there is no hope of doing them for anything other than a tiny handful of species currently queuing up in Red Lists on the road to extinction; they require too much data, and there are too many species. For birds alone (in species terms a tiny and insignificant fraction of the world's biota, Lawton 1996) there are already over 1000 threatened species (Collar *et al.* 1994), and habitat destruction guarantees many more in the next few decades (e.g. Balmford and Long 1994).

The optimum strategy would therefore seem to be to use limited conservation resources to do MVP analysis sparingly, but on large-bodied species, and species that occur high in the food chain. (This is an unreconstructed zoologist's perspective! Similar sensible criteria for rationing effort may exist for plant populations, but I am unaware of what they might be). Large bodied animal species, and species feeding high in the food chain are often, but not always the same thing, and they tend to be the kinds of charismatic organisms that attract public attention and support. For birds at least, larger bodied species are also those most at risk of extinction (Gaston and Blackburn 1995) and, of course, large bodied species and top predators need very large areas of habitat to survive. Designing an effective conservation strategy for such species, either in designated reserves or parks, or in the wider natural or semi-natural countryside, therefore more or less guarantees that many smaller, lowlier, associated species will also survive by default. An ecosystem large enough, varied enough and rich enough to support a viable cheetah population has an awful lot of space and habitat for slugs, snakes and pseudoscorpions. It is not a very sophisticated objective, but it is likely to be effective, none-the-less.

Ecosystem management for conservation and the role of science

The world is now dotted with protected areas, in the form of nature reserves, national parks and wilderness areas - the names vary but the objectives are broadly the same - to carry at least a proportion of the earth's biota through into the next century and beyond. What many people fail to recognise, and which is therefore a source of endless confusion, is that the very establishment of these protected areas (where they are, their size, and the degree of protection afforded to them) is not in itself a scientific process. Science may help to inform the process of establishment, but the decisions are ultimately political, ethical, aesthetic, even religious, and embrace much more than just scientific information.

Moreover, even when a protected area is established, the dilemmas for science and society do not stop; if anything they become more intense. Except for some of the largest protected areas on earth, all these systems require some form of management, and in general the smaller they are, the more intensively managed they need to be. Protected areas are rarely intact hydrological units; their inevitable, growing isolation from surrounding landscapes disrupts a wide variety of processes, from natural fire regimes to the migration of species in and out of the area; invading, exotic weeds do not recognise park boundaries, and so on and so forth. In smaller protected areas, edge effects become significant and ecosystem process may be drastically altered (Didham *et al.* 1996). And virtually all protected areas, whatever their size, need management policies for people, be they indigenous people with traditional land-use rights that may not be sustainable as their population grows, or visitors from outside who wish to walk, bungie-jump, photograph and shoot things.

How are the management policies to be decided? The answer is with great difficulty, because the decisions about what to manage for are again not, ultimately, scientific decisions. They too are political, economic, ethical, aesthetic and religious decisions, but they are not scientific ones. Let me try and explain why.

First, science is clearly involved in delivering effective management once management goals have been defined. Wetland recreation, changes in the fire regime, culling ungulate populations or any other management practices require an underpinning of ecological science, both to carry them out effectively and to predict their consequences. Ecological science can also inform managers, politicians, or citizens of the consequences of continuing with some particular course of action, or of changing or stopping it, and hence can help to set management objectives - to reduce or increase fishing quotas, or timber harvesting, or to change the water abstraction regime and so on.

The problem is that none of these many and varied activities, scientifically sophisticated and difficult as they may be, tell society what the ultimate management goals ought to be. What do we want to manage for? Nature conservation - why? Endangered species - which ones? Ecosystems - in what state? Sustainable fisheries - I would rather put the money in the bank, because money grows faster than fish. Resilience - meaningless. The list of difficult questions is endless. Nor are the answers obvious, because one person's burning objectives are another person's obstacles to a better life. In other words, at its heart, setting management objectives for conservation is not a scientific activity. There are even those in any nation for whom conservation in any guise is not on the agenda. Somehow, that ill-defined body known as 'society' has to decide what its environmental and conservation objectives are, and then act accordingly. To repeat myself, science can help inform that choice, but it cannot make it.

This problem rears its ugly head in the vexing, albeit apparently simple question of ecosystem management for nature conservation. In recreating, restoring or managing existing ecosystems, what is the baseline state we wish to hold to? 'Pristine' is not an answer, because as we now know, there is no such thing. Unaffected by human beings, ecosystems change continuously. 1/f environmental noise guarantees that this will be so. Ten thousand years ago, the major heathland National Nature Reserve known as Chobham Common, just a few mile away from where the Sibthorp seminar was held, was a barren tundra (ice ages occur infrequently, but with great power). Then, somewhere between roughly 5 and 10 thousand years ago it was progressively birch, pine and oak-hazel woodland, and finally under the combined human impacts of felling, fire, primitive agriculture and grazing it turned into heathland. Now, left unmanaged, it reverts back to scrubby birch-pine woodland. But 'society' has decided that heathland is more valuable from a nature conservation point of view than birch-pine woodland, and Chobham is one of a number of heaths actively managed to maintain and to restore this threatened ecosystem. But the decision has no rational basis in science. We might just as easily have decided that trees look nicer than heather, and changed the management regime accordingly.

In sum, any decision about what state to manage an ecosystem in for conservation is arbitrary. Whatever we choose, the system was probably not like that 500 years ago, and certainly not like it 5000 years ago. The best we can do is to try and minimise modern human impacts that impinge upon the system from without, and to keep Nature's options open. This is easiest in very large areas, and becomes more and more difficult as the size of a protected area declines. Paradoxically, ecosystem management in small reserves (from a few 10's to a few 1000 ha) is often dominated by the need to maintain habitats for one or a handful of endangered species, emphasising once more the inevitable interplay between species and ecosystem conservation.

Species are more constant entities than ecosystems

The arbitrary distinction between ecosystem conservation and species conservation becomes even more obvious over long time scales. There is an implicit tendency in much of the thinking in this area to assume that species are the vulnerable entities, and that ecosystems are somehow more permanent. Most readers will have been comfortable with the earlier section in which I discussed the existence of surviving ecosystem remnants lacking characteristic species. This was, of course, a journalistic trick. In reality, it is species that are the constant elements and ecosystems that are transitory. Successional ecosystems, by definition, are often very transitory, and survive for very short periods of time. Some successional ecosystems, *Phragmites* reed beds, for instance, which survive as tiny remnants in the highly modified landscapes of north-west Europe, require quite exceptional land-management efforts by conservationists to maintain them. But on longer time scales, all ecosystems are transitory. The drama of Chobham Common has been played out with different actors, on different stages, over vast regions of the northern hemisphere time and time again.

The critical point, however, is not only that ecosystems change, but also that fossil and subfossil remains of flora and fauna show that whole sets of species do not respond as tightly integrated units to natural variation in the earth's climate. Communities do not move together. Rather individual species respond in a highly idiosyncratic manner (Coope 1978, 1995; Davis 1981, 1986; Foster *et al.* 1990; Elias 1991; Graham *et al.* 1996). Many extant species combinations and ecosystems have no historical equivalents; and species combinations and ecosystems existed in the glacials and interglacials that have no modern equivalents. The same pool of species can apparently create a variety of ecosystems, depending upon the vagaries of migration and geographic isolation, and the particular climatic, geological and other environmental features that are unique to each time and place on earth (Bell *et al.* 1993). In the long term, species conservation will determine what kind of ecosystems might exist, because ecosystems are more ephemeral than species. To use an analogy, the massive, human-induced species extinctions that confront us over the next century will strip Nature's tool-kit of thousands of useful parts, with who-knows-what consequences for her ability to build the new ecosystems of the future.

Global environmental change

If this statement is true in the absence of people (human beings did not create 1/f environmental noise), it is even more true in a world threatened by human-induced global environmental change. Human beings do not create environmental change where none existed; they speed up, magnify and alter the nature of the change. We are part of 1/f noise with a vengeance. Without wishing to be dramatic, none of the earth's apparently protected ecosystems are likely to survive human-induced environmental change unscathed. Massive inputs of anthropogenic nitrogen from the air in temperate latitudes of the northern hemisphere are already a reality (Bell 1994; Vitousek 1994); so are rising global atmospheric concentrations of CO_2, with potentially profound consequences for ecosystem function and species composition (Beerling *et al.* 1993; Mooney and Koch 1994; Phillips and Gentry 1994; Cebrian and Duarte 1995; Korner 1996). In the longer term, if it has not already started, global climate change driven by rising CO_2 and other greenhouse gases, and associated sea-level rise appear inevitable. Global environmental change basically means that all our precious, protected ecosystems will be in the wrong place. Wrong because they will be destroyed by sea level rise; and wrong because the ecosystems they were set up to conserve will not be viable as the earth's climatic zones shift. We simply have no idea whether (or rather, which) species will be able to migrate fast enough across highly modified, human-dominated landscapes to survive in a rapidly changing world. And we have no global conservation strategies to cope with this potentially devastating scenario.

Concluding remarks

Although this is a grim scenario, it is not hopeless. At least we know the magnitude of the task ahead. Several things follow if we agree that our objectives are to minimise an inevitable crisis of extinction in the 21st century.

First, we should stop wasting valuable time and effort arguing about abstract, and ill-defined concepts like resilience, and marshal our limited resources to more practical and urgent ends.

Second, the world needs to bring as much of surviving wild lands and semi-natural ecosystems as possible into active protection for nature conservation. This does not mean the exclusion of people from such areas; it does imply that how people use such areas will have to be managed. Outside protected areas, we also need to ensure that land management is as sympathetic as possible to the myriads of other species that share the landscape with people, crops and livestock. Maintaining biodiversity outside strictly protected areas is a critical test of sustainable development.

Having large protected areas will not, of course, stop global climate change, but they hold the pools of species from which future ecosystems can form, and stepping stones for species' migration. The world conservation movement also needs to urgently decide whether there are management strategies to minimise the damage that will be wrought by rapid climate change. Should we actively move species, for example? Current protocols for species introductions and re-introductions are ill-equipped to deal with the need for wholesale movement of organisms outside their current 'natural' ranges. Let us then stop worrying about whether we need to conserve species or ecosystems; the answer is both, with the focus of our efforts as much determined by the size of the protected area as anything else.

Finally, we should recognise that there are almost overwhelming forces gathering on the horizon in the form of human induced global environmental change. It is no longer tenable for conservation biologists simply to focus on protecting species and ecosystems. We have to enter the political debate about energy use, human population growth, transport, and global change. Failure to do so means that we will win individual skirmishes and battles over the next half century (an endangered species saved here, a national park created there), only to loose the war in tides of immense environmental change that have no precedent in the history of life on earth.

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3. FOCUS ON KEY ISSUES

3.1 Is the Conservation of Vegetation Fragments and their Biodiversity Worth the Effort?

Vernon H. Heywood

Introduction

Fragmentation of natural ecosystems is generally seen to be one of the most serious threats to biodiversity worldwide, as anyone who has travelled extensively will have observed. It is customary to refer to fragmentation as occurring when human activities such as agricultural development, forestry or urbanisation remove large proportions of the natural ecosystem and replace them with a greatly modified matrix, within which small remnants of the native ecosystem remain (Hobbs and Lleras 1995). The result of this process is to reduce vastly the areas of natural ecosystems and to subdivide them into small, relatively isolated fragments or habitat 'islands' of different size. The consequences of habitat fragmentation on populations of the species that grew in the original continuous ecosystem is likewise to change them into subdivided, disjunct, isolated patches of habitat that are vulnerable then to extinction through demographic stochasticity, environmental stochasticity and catastrophes, loss of genetic heterozygosity and rare alleles, edge effects, invasory species, and human disturbance.

Vegetation fragments have been considered terrestrial islands and to behave like true islands, subject to the rules of island biogeography, especially as regards species' extinctions, although there are significant differences between them.

In understanding the threats faced by species in fragmented habitats, it is customary to use the framework of metapopulation theory. One of the consequences of fragmentation, therefore, may be to convert a previously more continuous population structure to a metapopulation structure, with all or most local populations becoming so small that they have a substantial risk of extinction (Hobbs and Lleras 1995). Long-term persistence at the level of local populations becomes less likely but metapopulation persistence still remains a possibility: many species have survived even if there are no large enough populations which could be expected to persist for a long time, and species do survive in continually changing landscapes. A species may be able to survive regionally if it is able to establish new local populations elsewhere where the right kind of habitat appears (Hanski *et al.* 1995).

Where there is no balance between extinction and colonisation, the whole metapopulation may decline to extinction. In many instances and certainly in the case of most tropical species, we simply do not know enough about the original size and distribution of populations nor the subsequent situation after fragmentation to allow us to make generalisations about the persistence of species. Nor do we know much about the habitat requirements of most of them. On the other hand, for many species, especially of vertebrates, and for rare and endangered species in various other groups of organisms, a large amount of empirical knowledge exists, much of it as personal knowledge of field biologists and not formally recorded or published; Hanski *et al.* (1995) suggest that this is one of the most relevant sources of information for conservation biologists and the challenge is to collect, classify and maintain this information for ready access by potential users.

As we shall see, fragmentation or patchiness of vegetation is an almost universal phenomenon. Not surprisingly, habitat fragmentation has been described as a central concern in conservation biology (Harrison 1994) and there are good grounds for considering it of similar importance in conservation management. As Margules (1987) observes, "Given the existing patchiness of much habitat throughout the world, research should be concentrated on the management problems of small patches, ... rather than with an unhealthy academic debate on the size and shape arrangements of reserve networks. Although the debate is entrenched in the ecological literature, so rarely is there an opportunity to plan reserves prior to habitat fragmentation, it is not really a major conservation issue." This is echoed by Saunders *et al.* (1991) who observe that there has been much focus on the design of nature reserves, but we are usually too late to do anything but maintain the remnants left following fragmentation.

Management has to concern itself with both the fragmented ecosystems and the populations of the species that are found in them, although practice in conservation has tended to treat these as separate fields of activity, reflecting the observation by Lawton and Jones (1993) that for almost three decades ecosystem and population ecology have ploughed their own independent furrows and developed their own paradigms, approaches and questions.

The genetic diversity of the species' populations that occur in habitat fragments is not only of interest on theoretical grounds but is a matter of serious concern in respect to the maintenance of genetic resources of species that have an agricultural or other economic values. Such 'genetic erosion' tends to be overlooked in discussions of fragmentation and conservation (Turner 1996), perhaps because many fragmented species populations are more likely to become extinct through stochastic factors rather than loss of genetic variability; if, however, conservation of genetic variation is a focus of concern, as for example, in crop relatives then such arguments lose their force.

In addition to vegetation fragments that are caused by human action, very many habitats are naturally patchy and their survival and that of the species contained in them is subject to many of the factors that affect the former. There are, however, clear differences between natural and human-caused fragments and we may learn lessons from a study of natural patches of vegetation when it comes to management and conservation.

How widespread is fragmentation?

Today most 'natural' habitats in the temperate zone are in fact small remnants of what were once much larger ecosystems (Ehrlich and Murphy 1987): islands of second growth woodland in northern Europe or the eastern United States; tiny patches of disturbed prairies in vast crop lands of the central plains of North America; patches of coastal communities in the West Mediterranean and so on. Likewise in the tropics, although large, unbroken stands of lowland humid forest remain in the main blocks of such vegetation in the Amazon basin, the Congo basin and in parts of south-east Asia, much of the remainder this biome has been converted during the past few decades into scattered and isolated fragments, often less than 100 km^2 and heavily disturbed. Dry tropical forests, for example in Central America, have suffered similar fragmentation, although less publicized.

There is nothing new in fragmentation – it is a process that has been going on since the beginnings of agriculture. What does mark out today's situation is the global scale of such fragmentation, especially in the past few decades, and the underlying socio-economic factors that encourage it, with the result that there are few parts of the world where substantial tracts of intact 'undisturbed' vegetation of any kind can be found and the few large blocks of more or less intact vegetation that do remain seem likely to follow the same path as a matter of time.

In temperate areas, fragments of 'natural' or semi-natural vegetation forming islands in a matrix of agricultural ecosystems, uncultivated land, urban development, plantations, secondary vegetation, are such a normal aspect of today's landscapes that there is a tendency to overlook their importance. In Europe, for example, the landscapes have been subjected to human modification for thousands of years and it is often difficult to find substantial enough areas of 'natural' ecosystems to set aside as protected areas. The amount of old growth forest in Western European and Scandinavia ranges between 1% and 3% (Dudley 1992) and in the Iberian peninsula, the characteristic evergreen oak forests (*Quercus rotundifolia*) that so characterise the Mediterranean bioclimatic zone are reduced to fragments and many of the remnants that do occur are secondary and heavily altered and invaded by species of pine. In fact it is sometimes difficult to distinguish 'natural' from 'semi-natural' to clearly artificial habitats.

Curiously enough, the losses of plant and animal species as a result of this long-lasting human-induced disturbance is surprisingly small – only about 30 higher plants have been recorded as extinct in Europe since the year 1600 although over 2000 are threatened with extinction to some degree. This raises a whole series of issues regarding prediction of future extinctions, time required to reach equilibrium, the role of past disturbances as extinction filters (Balmford 1996), management and policy, that are discussed below.

In the tropics, it has been estimated that 65% of the original ecosystem south of the Sahara has been subject to major ecological disturbance and 68% of natural habitats in the Indomalaysian region. Globally Hannah *et al.* (1994) have estimated that 73% of the world's land surface, other than rock, ice and barren land, is human-dominated or partially disturbed while 17% is 'undisturbed' ('undisturbed' being defined in terms of roadless areas in chunks of 100 000 ha or more) (Hannah *et al.* 1994; McNeely 1995). Indeed so many of the world's habitats have been modified to some degree by human action that there is little practical value in using such terms as 'native ecosystem', 'undisturbed' or 'virgin forest'.

It is clear, therefore, that the conservation of biodiversity has become largely a challenge of maintaining that found in fragmented ecosystems, in a matrix of other kinds of landscape use. This poses acute problems where such fragments occur in areas of high species' endemism since the populations of the species are also likely to be similarly fragmented and threatened with eventual extinction. If the species are widespread, the extinction will be local; if they are limited to the area concerned, the extinction will be global.

Fragments on islands

If we consider that a large proportion of the world's endemic plants and animals (and also, one must assume, micro-organisms) occur in island ecosystems, then the problem is of alarming proportions since these are amongst the most threatened in the world, usually as the result of devastation caused by human action. The island of Rodrigues, for example, in the Indian Ocean was once covered with a rich and luxuriant evergreen forest but as a result of three centuries of human habitation all the original plant communities have gone and the island is today mainly barren hillsides, dotted with trees or covered with a usually monotypic shrub or thicket of introduced species; only a few areas of degraded native forest exist (Strahm 1989). At least 18 endemic plant species have become extinct (Strahm 1996) and at least 11 species of endemic birds, two species of giant *Phelsuma* lizards, two species of giant land tortoises and an unknown number of insects and other organisms. According to the *Plant Red Data Book for Rodrigues* (Strahm 1989), of the surviving 36-38 endemic flowering plants 19-21 are Endangered, 7 Vulnerable and 8 Rare, with 9 of these endangered species reduced to fewer than 10 individuals and 3 known from only a single wild individual. If the combined floras of Rodrigues and the neighbouring island of Mauritius are considered, 120 taxa are known from either less than 20 individuals or just one or two populations, and 28 species are known from less than ten individuals in the wild (Strahm 1996). Despite this apparently hopeless situation, the work of Strahm and others during the last 10–15 years has, through a programme of careful management, fenced-in areas, artificial propagation of both plants and animals, replantation, weeding and promotion of conservation awareness plus the designation of several areas as nature reserves, has enabled many of these species to be rescued from total extinction.

Again, on the Island of St Helena, no unmodified natural vegetation communities survive, more than 95% of the island's area is a human-made landscape resulting from centuries of agriculture and resource exploitation, the surviving populations of endemics are both small and scattered and nine of the endemic taxa are not viable in the near to medium term (Maunder *personal communication*). As in the case of Rodrigues, all surviving areas of habitat will require varying degrees of conservation management and action to ensure the survival of the threatened plants and animals, and a programme for the conservation of the endemics has been prepared and awaiting approval (1996).

The same kind of situation can be repeated for many other islands and Table 3.1 summarises just some of the most extreme examples in small oceanic islands. The following are other classic examples of islands that have suffered extreme degradation and fragmentation of much of their vegetation and biota and that house large numbers of endemics: Madagascar where at least 80% of the original vegetation has been destroyed and over 80% of the plants species are endemic (and there is a high level of animal endemism); the Southern Domain of the island, where very few areas remain undisturbed, has 95% plant endemism and most of the legally protected areas house narrow endemic or near endemic species. The Hawaiian islands where 110 plant taxa survive in populations of 20 or fewer individuals. The Caribbean Islands which have been under European influence for about 500 years and much of the native vegetation

Table 3.1 Small oceanic islands with devastated floras (from Strahm 1989)

	(sq km)	Ex	E	V	R	I	?	nt	Total	Threatened
Ascension Island	94	1	5	–	4	–	1	–	11	10 (91%)
Norfolk Island	39	3	9	26	–	5	2	1	48	45 (94%)
Rodrigues	109	6	22	7	9	–	–	3	49	46 (94%)
St Helena	121	7	23	–	17	–	2	–	49	47 (96%)

destroyed or fragmented by human exploitation; the islands house 6550 single-island endemic vascular plant species (50% of the total flora) and over 3000 are rare or threatened.

Globally some 40 000 species of higher plants are endemic to oceanic islands and one in three of all known threatened plants are island endemics and 23% of island birds are threatened compared with a figure of 11% world-wide (WCMC 1992).

'Natural' fragments

Patchiness may be natural rather than anthropogenic. We must, therefore, recognise that many 'natural' fragments of ecosystems occur in biomes around the world: for example small patches of vegetation on mountain tops in the Mediterranean (where 50% of the region's endemic plants and some endemic animal taxa occur), in topographically fragmented coastal areas as in the Cabo de Gata region of Spain that have a special type of vegetation and a number of endemic plant taxa, and on specialised soil types such as serpentine areas in Cuba and Jamaica.

Of course the nature and structure of the vegetation fragments and the species' populations they contain are different from what is found in 'artificially' fragmented ecosystems and populations; so are their survival probabilities. Nonetheless the study of the amount of variation and genetic structure of their populations of such examples may give us an insight into how such factors affect their survival and the planning of survival strategies for small populations of endangered species. Moran and Hopper (1987) found, for example, that populations of localised species of mallee eucalypts are often similar in size to those isolated by land clearance and now restricted to natural vegetation fragments. Moreover, many of these localised species have considerable genetic diversity and they suggest that the frequently recommended minimum population sizes of 500 for survival seem inappropriate. They speculate that in the ancient landscapes of Australia it may be that eucalypts have evolved to cope with small population sizes and can persist for over 1000 years and maintain a high percentage of their genetic variation.

True islands and terrestrial islands

As noted in the introduction, in many discussions of fragments an analogy is made between fragments as terrestrial 'islands' and true islands and subject to the theory of island biogeography. For example, much research has been devoted to attempting to explain the patterns of species loss that follow fragmentation in terms of island biogeography and species-area relationships. Thus it is argued that the decrease in the size of patches of vegetation will be accompanied by a loss of species and the eventual attainment of a new equilibrium and the speed and extent to which relaxation in species numbers occurs is important in determining the long-term viability of the communities. Although there are clearly lessons to be learned, the differences between true islands and terrestrial islands are non-trivial and aspects of island biogeography theory have been accepted and applied uncritically to vegetation patches. For example, terrestrial fragments occur within a varied landscape matrix and the fate of the species populations in the fragments and the genetic variability in these populations will depend on a whole series of factors such as population size, gene flow between fragments, demography, distance between fragments, their shape, size and structure, breeding system, pollinators, within-fragment species density, some of which are dependent on features of the matrix which in island ecosystems do not exist.

Similarly the value of metapopulation theory in conservation biology has been considered as analogous with that of island biogeography theory (Harrison 1994) in that it suggests hypotheses to test and important data to gather but can produce neither powerful generalisations nor ready-to-use formulas. Perhaps we can agree with her conclusion that "conservation biology is essentially an empirical science in which theory provides guidance, but in which each case requires its own appropriate hypotheses; and that in the practical arena, we may do better to explain than hide the complexity and uncertainties involved".

How many species are involved?

We do not, of course, know just how many species occur in fragments but we can obtain some idea of the scale of the problem from various sources of evidence. We have already seen from the large numbers of species that occur in fragmentation-prone areas such as islands and mountain ecosystems, that most of the species that are likely to become extinct in the coming years are likely to be found in vegetation fragments. It would seem too that because of the extent and pattern of vegetation fragmentation that more species will be *threatened* with extinction than even the highest estimates for future extinctions based on species-area models (see for example Reid 1992). Moreover, because of fragmentation it is virtually certain that the loss of genetically distinct populations is occurring at a much higher rate than that of the loss of entire species, as Ehrlich *et al.* (1977) pointed out over 20 years ago.

Published estimates of species that will become extinct or committed to extinction in tropical forests due to projected forest loss over the next 20–30 years range from 2% to 25% according to the groups examined (variously: plants, birds, birds and plants, all species). This would be equivalent to 1000 to 10 000 times the expected 'average background rate'. In a few cases, in what Pimm (1996) has recently termed 'kilodeath black spots', documented extinctions are of this order but in most cases they are predictions of potential extinctions, based on species-area techniques, and would not be immediate. Indeed it could take decades or even centuries to reach the new equilibrium number of species.

The focus on the scale of probable *future* extinctions of unspecified species has, possibly, done a disservice to conservation in that it has drawn attention away from the urgent need to take conservation action that will reduce the risk of total extinction of the tens of thousands of severely depleted species populations that are threatened *now* and which are increasing by the year (Heywood and Stuart 1992).

Extinction is the culmination of a process, and it is important from a conservation management perspective to distinguish between potential extinctions and actual total extinctions. A threatened species may still leave time for remedial action to be taken whereas extinctions are irreversible. Since the matter of extinction rates is a sensitive issue and had led to a great deal of sterile debate, it is important to stress that to concentrate on threatened species and their probabilities of survival is in no way to minimise the importance of extinctions: on the contrary it has the advantage of dealing with the actual situations of known species, rather than theoretical (however plausible) predictions about the fate of unspecified species, and is positive in that it allows us to devise programmes for their conservation. And what is more, in view of what we have seen, the almost universal occurrence of fragmentation of vegetation will inevitably lead to putting many of the species that occur in these fragments, whether human-created or 'natural' into one of the threatened categories.

A more direct estimate of the numbers of species involved can be obtained from examining the global figures of threatened species produced by the World Conservation Monitoring Centre (WCMC) according to the IUCN categories of threat, although they do not specifically recognise which species occur in vegetation fragments. As a recent World Bank note comments (World Bank 1997), IUCN's Red Lists have long been recognised as the most comprehensive and authoritative global surveys of threatened species. Globally, WCMC currently record about 32 700 species as 'threatened' of which about 5000 are Endangered and over 8200 Vulnerable according to the old IUCN categories of threat. As will be seen from Table 3.2 most of these are plants. For higher plants, the WCMC database (as at 20 June 1996) lists 93 647 species, including 66 175 for which insufficient information is available to determine their global threatened status, and 12 222 that are not threatened. The 1996 Red List of Threatened Animals (IUCN SSC 1996), based on the new IUCN categories of threat, includes 5025 species, including all known species of mammals and birds (with 25 and 11 percent respectively threatened with extinction), but only a token number of invertebrates.

Table 3.2 Number of species recorded as 'Threatened' by the World Conservation Monitoring Centre (1996)

Threatened	Endangered	Vulnerable	Rare	Intermediate	Total
Mammals	177	199	89	68	533
Birds	188	241	257	176	862
Reptiles	47	88	79	43	257
Amphibians	32	32	55	14	133
Fishes	158	226	246	304	934
Invertebrates	582	702	422	941	2647
Plants	4301	6844	14457	5226	30918

One thing is quite clear: these figures seriously under-record the true number of threatened species since no assessment has been made of most of the world's total 1.75 million described species, especially (but by no means only) those that occur in the tropics. If we consider that considerably fewer than 10% of documented species in all groups have been evaluated, the total number actually threatened must be correspondingly higher. If we were to multiply the c. 36 300 recorded threatened species (Table 3.2) by ten to allow for this, this would give 363 000 and if we were to apply a worst case scenario and assume all will become extinct within 25-30 years, this figure still falls far short of many of the projected numbers extinctions so widely quoted in the literature (cf. Reid 1992). Part of the reason for this is that the latter are projections that not only apply to known species but also take into account the c. 87% of species that have not yet been described (Heywood 1995). Thus the discrepancy between the recorded figures for evaluated species and the probable total of threatened species is vast.

These figures only serve to highlight just how inadequate is our database of knowledge regarding current threats to species, not just those that occur in fragments, and suggest that much more effort needs to be invested in this kind of assessment, both nationally and globally, since it is on these that conservation action has to be planned.

Survival of species in fragments: how long do we have?

From a conservation management point of view we need to pay more attention to the length of time that currently threatened species and populations are likely to survive in vegetation remnants (Frankel's 'timescale of concern') since species may hang on for anything up to hundreds of years and thus allow us the possibility of taking remedial action. Hanski (1994) notes that the timescale of concern in the case of metapopulation dynamics is relatively long and it is often doubtful if a stochastic steady state will ever be reached. Indeed it is probable that a large number of threatened species occur in landscapes that are already inadequate for their long term survival: the species still exist because they have not yet had time to go extinct (Hanksi *et al.* 1995) – the so-called functionally dead species.

Clearly a critical question that has to be studied is under what conditions will a species that has evolved in a more continuous habitat survive, or evolve to survive in the more fragmented environment (Hobbs and Lleras 1995). Also, it not just a question of studying the factors that affect the survival of single populations but of the various components of the metapopulations.

We are now beginning to obtain evidence that gives some indication of the numbers and length of time that species will survive in vegetation fragments and the results are mixed as one might expect. It is instructive to look at some detailed authentications of the situation in some areas where extensive loss of habitat caused by human action together with fragmentation has occurred and dates back for decades.

In Singapore the loss of primary forest cover has been reduced from nearly 100% in 1819 to 0.2% today. In a study of plant species extinction in the state, Turner *et al.* (1994) made a detailed survey of the native vascular flora of some 2277 species and found that 594 species have become locally extinct, 117 Endangered, 391 Vulnerable, 957 Rare and 218 common. The extinctions therefore represented 26% of the flora within the Republic. It appears that at least 90% of Singapore's forests were cleared before serious inventory began in the 1880s so it is possible that some species became extinct before they were collected although they do not think that the number would have been large since the total inventory is as long if not longer than that of areas of comparable size in other tropical regions. A similar extinction of birds (28%) has also been recorded. They then estimated the effects of deforestation on species diversity, applying the species-area relationship to the forest component of the flora and found that it predicted a 76% loss as opposed to the 29% that he had estimated on the basis of inventory. They concluded that plant communities of the tropical rainforest take a long time to reach equilibrium since most of the forest clearance took place last century. The encouraging conclusion drawn from this study was that if small remnants can be safeguarded they will probably provide refuge for many species for a considerable period without requiring complex and expensive management (although this latter point may be a matter of debate). Also time will be available to use both *in situ* and *ex situ* techniques to ensure the long-term survival of these species.

In Hong Kong, despite almost total deforestation and defaunation that took place over 300 years ago, a large number of tropical forest plants species remain (Corlett and Turner 1997). This type of situation leads Turner and Corlett (1996) to suggest that there may be grounds for believing that a substantial element of the rainforest community may be relatively resistant to fragmentation, and that dire predictions of plant extinction, through loss of pollinating and seed dispersing vertebrates, may be overly pessimistic. Clearly we need more detailed information of the detailed processes of diversity loss in both tropical forest and other types of community if we are to plan our management of remnants in the most effective manner possible.

A similar and in some ways more extreme situation obtains in one of the most cited 'megadiversity' areas of the world – the Mata Atlântica of Brazil. These coastal forests originally extended from the north-eastern tip of Brazil to the country's southernmost state and covered an area of c. 1.2 million km^2. of which over 1 million km^2 had been deforested by 1990, and are now reduced to 2–12% of this original area. Much of the conversion of the forest was completed by the end of the 19th century. The Atlantic forests occur in the agricultural and industrial heart of Brazil with c. 43% of the country's population and are highly fragmented. They house some 13 000 plant species (Gentry 1982) which represents c. 13% of the whole neotropical flora (and is equivalent to the complete vascular flora of Europe) although the inventory is still incomplete and new species and even genera are still being found. Levels of both regional (73%) and local endemism are remarkably high. Animal diversity is also high although less well documented.

Despite the fact that much of the forest was cleared a century or more ago, only a handful of species are known to have become extinct (Brown and Brown 1992), although many are reduced to small populations in the fragments and are threatened with imminent extinction unless remedial action is taken. Although it has been argued by Budiansky (1994) that the species-area relationship does not appear to apply here, what appears to have happened is that for a number of not fully understood reasons, a new equilibrium has not been reached and many species seem to be able to survive so far in the remaining patches of vegetation and some are even recorded to have expanded in area in recent years. Brooks and Balmford (1996) argue that the apparent mismatch between theory and the real world is illusory since an analysis of Atlantic forest birds in relation to deforestation patterns shows that the species-area relationship predicts quite accurately the number and distribution of endemic species of bird already in serious danger of extinction through habitat loss. However, this simply reinforces the fact that despite the way in which it is commonly quoted, the species-area relationship calculates how many species are 'committed to extinction' (Reid 1992; Heywood *et al.* 1994) as a result of habitat loss as opposed to becoming extinct at a particular time and as Simberloff (1992) has noted, there is no accepted theory for the rate at which species will be lost through time and the new equilibrium reached.

On the other hand the results so far of the Biological Dynamics of Forest Fragments Project (BDFFP) – a binational project between INPA in Brazil and the Smithsonian Institution – indicate that after twelve years of observations "the forest reserve fragments are highly dynamic ecological entities and that it is an oversimplification to expect to be able to predict 'species carrying capacity' from the size of a reserve alone. Species/area relationships are insufficient to understand all of the processes that determine how many and which species will be present in a given reserve. Data have confirmed that extinction proceeds faster in smaller fragments for some taxonomic groups, whereas groups behave in ways not predicted by the species/area relationship" (Bierregaard 1996). Clearly we need to study what are the factors that make some species extinction-prone, and what are the factors that permit others (often closely related) to survive.

In these and many other cases, there is evidence of prolonged relaxation times which mean that cases of extinction through habitat fragmentation take many years to reach the levels of diversity predicted by the species-area relationship. This has obvious implications for conservation and management. Of course many of the species concerned are what are referred to as 'functionally dead' and whether action to recover them will be regarded as worthwhile will depend to a large extent on their scientific, economic or other interest.

Conservation management: what action is possible?

It is fairly obvious that although the chance of rescue of fragments and of the species they contain will often exist, it will not be possible to take preventative action to save all of these species: in other words the numbers are greatly in excess of the human, technical and financial resources currently available to deal with them and it is unlikely that massive additional resources will be available in the foreseeable future. Indeed the scale of loss of contemporary species has led many conservation biologists to underline the need to direct conservation efforts to entire communities rather than to single species (e.g. Balmford (1996) who suggests that the current emphasis on fire-fighting in areas where threats are already extensive should be reviewed and directed towards potentially more-rewarding pre-emptive conservation action in relatively untouched and therefore vulnerable areas – see his Box 1). This would clearly argue against efforts to save fragments since they do not meet the criteria of high biological importance and low threat. There are, however, severe limitations to such a strategy and if adopted it would have the effect of condemning large numbers of species that could be saved to extinction. As Turner and Corlett (1996) suggest, 'Fragments are better than nothing'. They can act as last refuges for plant and animal species and may provide an opportunity for conservationists to put in hand attempts to rescue species from extinction. They may also help in the reforestation of landscapes through acting as sources of material for recolonisation and the recreation of forests.

The identification of 'hot spots' or other centres of diversity is one of the approaches to establishing priorities for biodiversity conservation (and other approaches have been proposed based on complementarity, taxonomic or phyletic uniqueness, etc.). In the case of 'Centres of Plant Diversity' (Davis *et al.* 1994–7) 234 major sites of plant diversity of global importance were selected based on their species-richness (even though the number of species might not be accurately known) and the area had to contain a large number of endemic species; additionally other characteristics such as diversity of habitat types present and presence of genetic resources of plant useful to human activities were applied. Other approaches have been suggested, such as the Conservation Potential/Threat Index (CPTI) developed by Dinerstein and Wikramanayake (1993) and Dinerstein *et al.* (1995). However, the success of any of these methods depends on the practicalities of their implementation.

In the case of the 234 sites recognised by the Centres of Plant Diversity, worldwide fewer than one in four (21%) are legally protected in full and only about one third (35%) have more than 50% of their area occurring within existing protected areas. Even more serious is the fact that a large proportion of the sites that are officially protected are not effectively managed and to give one regional example, of the 41 sites in Southeast Asia only 3 are considered to be reasonably safe or secure. There are serious limitations to the adoption of a policy that is based on a requirement for the conservation of large and preferably contiguous tracts of forest (or other types of ecosystem). Even if this were possible, there will be many other endemics and disjunct populations (cf. Koopowitz *et al.* 1994) – and these may represent the majority of plant species – that occur scattered through the world's forests that cannot be preserved *in situ* because of the limitation on the number of sites that can be set aside and effectively maintained.

Even where the protected area system is fairly good, as in Borneo (including Sarawak), because of the many endemic species and high level of diversity of plants and animals, some species will be missed by the parks, occurring in small areas or fragments, or simply not incorporated in the protected areas system (Dinerstein *et al.* 1995). Most tropical moist forest reserves in the Indo-Pacific region are not large enough to conserve entire ecosystems and maintain minimum viable populations of many larger species. Intensive management is therefore needed to deal with demographic, genetic and environmental threats of extinctions associated with isolated populations in small reserves. The dilemmas associated with managing numerous small populations will be the legacy conservationists leave for the next generation unless reserves are incorporated into larger conservation units.

The impact of climatic change on protected areas and on vegetation fragments is likely to be severe in many parts of the world in the coming decades although the details are still uncertain. This will add to the complexities of conservation planning and lead to a further increase in fragmentation of vegetation. What is quite clear is that a more integrated approach than hitherto to conservation will be needed in which protected areas, *in situ*, *ex situ*, reintroduction and restoration will all have a role to play.

Fragments and genetic resources

The genetic resource system, at least in the forestry sector, will view vegetation fragments from a different perspective – how much genetic diversity is there in the target species that occur in such fragments and how may this be maintained in time. For example, recognition that many of the species of genetic resource concern today occupy habitat fragments has led to the commissioning of research into the impact of habitat fragmentation on genetic diversity, mating systems and effective population size in tropical forest fragments in Costa Rica (CIFOR/IPGRI/CATIE) and Sumatra (CIFOR/IPGRI). Indeed the question has been raised – how far can fragments maintain prefragmentation levels of genetic diversity. There is some evidence that in wind-pollinated species such as *Acer saccharinum*, long-range gene flow may even increase certain measures of genetic diversity.

In recent years a great deal of attention has been focused on the conservation needs of the wild relatives of cultivated plants, many of which occur in fragmented habitats, such as the *Brassica oleracea* group (cabbage relatives) (Gustafsson and Lannér-Herrera 1997). A review of the problems involved in the conservation of the wild relatives of European cultivated plants is given by Valdés *et al.* (1997). One of the problems highlighted by Safriel *et al.* (1997) with reference to the management nature reserves for the conservation of wild emmer wheat (*Triticum turgidum* var. *dicoccoides*) in Israel, known as the Ammiad Project, is the difficulty of when to phase out research and when to start conservation and management practices. These problems have had to be faced in a major innovative conservation project supported by the Turkish government and the Global Environment Facility that was initiated in 1993 – the *in situ* conservation of wild relatives of crops and important forest species. It calls for the establishment of *in situ* conservation areas, called gene management zones, and involves a great amount of collaboration between agriculturists, forestry experts, environmental scientists, conservationist, planners and local communities.

Bioregional management

The conservation of habitat fragments and their contained species' populations will only be feasible within the framework of a bioregional approach to the conservation and sustainable use of biological resources, whereby all kinds of land use within the landscape matrix are taken into account. As Miller (1996) in a valuable review of the bioregional approach comments, "Since the landscape is fragmented and much wildland has been converted to other use, the boundaries and coverage of some protected areas may not conform to the size and shape of the ecosystems that are to be maintained and managed. ... Moreover, in landscapes where protected areas have not been established, key genetic, taxonomic, and ecological elements of diversity that once may have been found in wildlands, or extensive farm or forest operations, are now relegated to isolated patches in intensively managed farms, pastures, timber-harvesting sites, and suburban, urban, and industrial areas". Vegetation fragments, as we have seen, are now becoming an almost universal component of our landscapes – "Storm-battered islands in a sea of human settlement" (Lash 1996) – and their management has to be planned and implemented in the context of large biotically viable regions.

Conclusions

With rare exceptions, fragmentation is the commonest state in which the world's ecosystems survive today. The evidence that a majority of the Earth's species, at least in some groups, already occur in fragments is overwhelming. This fact alone must suggest that the conservation of vegetation remnants must be given very high priority – on a par with the effort that goes into establishing protected area systems around the world – if we are not to condemn a large part of our biota to extinction during the coming century or so. There is still time available to take conservation action but the scale of the problem is such that no serious attempt has been made to formulate a viable strategy to achieve this. However, as Frankel *et al.* (1995) comment, pragmatism must temper idealism and we must explore what is possible and not concern ourselves too much with the scientific niceties of conservation biology if they do not provide us with useful tools.

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3.2 Community and Population Perspectives in Ecosystem Management

Monitoring and Restoring Biological Diversity: Case Studies from Eastern United States

Richard B. Primack and Brian Drayton

A major goal of ecosystem management is to ensure viable populations of all species, particularly rare and endangered species, and to maintain representative samples of all biological communities (Grumbine 1994a, 1994b, Noss and Cooperrider 1994). Viable populations are those in which the numbers of individuals do not show progressive decline over time, that have the potential for new recruitment in succeeding generations, and that have a high likelihood of persisting into the foreseeable future (Schemske *et al.* 1994). An ecosystem that is sustainably managed is expected to have viable populations of all species, or at least viable populations of the same percentage of species as nearby control areas in which resource usage is not occurring. In an ideal scenario of ecosystem management, populations of rare and endangered species might even be larger than those in nearby control areas because populations could be managed to facilitate increases in numbers.

Biological communities have characteristics of species composition, vegetation structure and species relationships, all of which must be considered in ecosystem management (Primack 1998). Well-managed biological communities would be expected to have essentially the same characteristics as biological communities undisturbed by human activities. Again, the potential exists for ecosystem management to enhance the diversity of biological communities by creating conditions necessary for all representative communities in an area. In particular, many species that occupy early successional stages within their communities can be encouraged by creating the needed levels of disturbance, using such methods as controlled fires, digging up the ground and selective logging operations.

Case study of an isolated conservation area

A crucial component of ecosystem management is monitoring of populations and communities. Populations and biological communities must be carefully followed over time to determine whether management is effective. As an example, consider the Middlesex Fells, a 400 ha conservation are near Boston, Massachusetts, USA. The park was originally established 100 years ago to preserve open space for recreation and to provide a place for nature conservation. At the time of establishment a detailed census was made of the vascular plants present, with notes on their distribution, abundance and habitat.

In 1894 the Fells bordered woodland areas, but over the last century, the park has become isolated by at least a 5 km wide barrier of roads, housing developments, and urban development, from other conservation lands. In 1993, we conducted a detailed study of how the flora had changed over the last century (Drayton and Primack 1996). To our surprise, we found that out of the 422 species originally present, 155 of them could no longer be found in the park. Some 64 new species are now found on the site, the majority of them being exotic species. So even though there was a turnover of species, the number of species declined over time. Many of the exotic species occur in disturbed habitats, such as parking lots, playing fields, and trails. However some of the exotic species, such as European buckthorn (*Rhamnus cathartica*), garlic mustard (*Alliaria petiolata*) and purple loosestrife (*Lythrum salicaria*), have invaded undisturbed stands of vegetation, displacing native species. As a result of this invasion, the proportion of native species in the flora has gone from 83% in 1894 to 74% in 1993. Overall the number of native species is declining at a rate of 0.36% per year, with roughly one native species being lost from the park each year. In contrast the number of exotic species in the park is increasing over time. Whole elements of the flora are being lost such as orchids and lobeliads. All seven orchid species of moist habitats are no longer found. A number of conspicuous and easily recognised species that were common in 1894 are no longer present, such as virgin's bower (*Clematis virginiana*) and golden thread (*Coptis trifolia*). Other formerly common species are now only found in one or a few sites, often as solitary individuals or tiny populations, such as dwarf dogwood (*Cornus canadensis*) and nodding trillium (*Trillium cernuum*).

In a few cases, the reason for the loss of species can be determined. For example, certain species depended on agricultural or gardening practices which are no longer used. In most cases, the reasons for species decline could not be identified specifically. species were lost disproportionately from moist habitats, such as wet meadows, moist woods, stream banks and swamps. In general it appears that the single most important factor involved in the loss of species is the increased human impact on the site, including intensive recreational use, trampling on the plants, thinning of the forest by the local park managers and an increasing frequency of fires set both accidentally and deliberately by local residents. Fires in particular are probably making the Fells inhospitable for many moist woodland species. The effects of acid rain, nitrogen deposition, ozone and other components of air pollution may be causing the local extinction of many plant species that cannot grow and compete under the new set of condition, but the precise impact of these new factors is not known.

Related to these are the increasing numbers of roads, carriage trails and footpaths that dissect the Fells into ever smaller fragments. The fragmentation has several negative effects on the flora and fauna. First, exotic species that can displace native species typically invade along paths, roads, and forest edges. Second, fragmentation of habitat leads to a decline of certain fruit-eating birds, which are important in dispersing seeds and maintaining viable plant populations. Third, opening of trails can lead to microhabitat changes in the vicinity of trails, such as greater air movement, lower humidity, soil erosion and soil drying. These effects are detrimental to many plant and animal species. And fourth, a greater number of trails and roads means that people can easily reach the most remote areas of the park, so that all areas of the park are exposed to direct human impact.

The Middlesex Fells has also become a habitat island over the last century. Whereas in the past, species from adjacent woodland areas could presumably colonise the Fells, the Fells is now isolated from such colonisation. If a species goes extinct in the Fells because of successional changes, natural fluctuations in population size, or human activity, populations of the same species from outside of the Fells have no way to colonise the Fells. These outside populations of native species are now too far from the Fells and separated by too much intervening inhospitable landscape for colonisation to have much chance of success. This is especially true for species that rely on short-distance seed dispersal mechanisms, such as the ant-dispersal seeds of violet (*Viola*) species.

This study clearly demonstrates the value of long-term monitoring of sites for biological diversity (Goldsmith 1991). Simply establishing and protecting a conservation area does not guarantee that species in the protected area will persist over time. Human activities, natural successional changes, occasional catastrophic disturbances, and air pollution cause both subtle and dramatic changes in biological communities. In addition, the impact of rising carbon dioxide levels and the impending effects of global climate change will cause profound changes in the distribution and composition of biological communities (Gates 1993). Land managers will need to include these inevitable changes in their attempts to conserve the diversity of species and biological communities.

This study of the Fells suggests that ever increasing levels of unregulated human activity will definitely result in a continuing depletion of the original species. To stop the loss of species requires officials, park advocates and local citizens to take an active role in stopping damage to the parks. Obvious first steps are strong public education programs explaining the dangers posed by fires, suppressing fires once they start, restricting walks and bicycles to designated paths and roads, and preventing the opening of new trails. More active steps would involve closing some of the existing trails, particularly those near populations of locally rare species. A program of public education and discussion would have to precede any attempt to close trails, in order for such action to be accepted by the public.

Reintroduction of rare species

Potentially, biological diversity can be restored using a variety of approaches, many of which are in the early stages of development in the relatively new field of restoration ecology. We are currently engaged in a long-term effort to reintroduce many of the species that he been lost from the Fells over the last 100 years, such as cardinal flower (*Lobelia cardinals*), as well as create new populations of species that were formerly common and are now rare, such as spikenard (*Aralia racemosa*). This work is being done with an experimental approach to investigate a variety of factors that affect the creation of new plant populations.

First, for each species we are using at least 9 sites, to determine variation in the rate of population establishment. We hope to provide estimates of the probability of successful establishment. Stating it in another way; how many sites need to be used to have a high probability that at last one new population will be created?

Second, populations are being established using a variety of maternal and population genotypes to determine if genetic effects are important in population establishment, and if so, how important. While genetic effects are certainly known to be important in controlled situations, such as gardens and glasshouses, their importance under natural conditions is still being debated. It may well be that genetic effects are virtually insignificant in comparison with the overwhelming effects of environmental variation. On the other hand, genetic effects could make the crucial difference between which plants survive and which die.

Third, we are investigating whether planting adult plants, seedlings, or seeds is the best way to establish new plant populations. Adult plants have typically been used in reintroduction projects, and they have the advantages of having passed the vulnerable seedling stage and are large enough to flower. However, the disadvantage of using adults is that they are expensive to produce in the glasshouse, difficult to transport, and only represent a limited number of genotypes. Using seeds to start a new population is a more natural approach as it mimics the normal seed dispersal process. Using large numbers of seeds increases the likelihood of an adapted genotype being placed into the site. However, seeds often fail to germinate and grow at sites where they are planted.

Fourth, we are investigating whether site preparation is important in establishing new populations of rare species. For each species at the Middlesex Fells, quadrats selected at random at each site are dug with a shovel to reduce root competition as well as loosen the soil. These quadrats as well as adjacent, undisturbed control quadrats are planted with seeds. The quadrats are monitored to see how the site treatment affects the establishment of new plant populations.

Case study of managed pine forests

We are conducting a similar set of experiments at the Savannah River Site in New Ellenton in southern South Carolina. The Site consists of a large complex of buildings for processing nuclear materials for the US Government, surrounded by a buffer zone of 100,000 ha of sandhills pine forest. The forest has been managed for the last 50 years by the US Forest Service for maximum timber production. The management involves conversion of the native long-leaf pine stands to loblolly pines, which are considered to be faster growing on the sandy, freely draining soils. Ground fires are set at approximately 5 years intervals in these forests to remove the dense understorey of oak (*Quercus* species) shrubs and small trees which compete with the pine trees for scarce soil water. The fire generally does not damage the fire-resistant pine trees. Trees are harvested at the age of 50 to 70 years.

When the Forest Service began this management plan for the Savannah River Site, the goal was to maximise wood production. However, this goal is now being changed to one of ecosystem management. The US Forest Service now recognises that this large block of forest contains many plant and animal species that are listed under both national and state Endangered Species Acts, some of these species, such as the US listed red-cockcaded woodpecker, is unable to adapt to the current forest management practices, because the species requires old long-leaf pine trees in which to build its nest cavities. The US Forest Service now recognises that such forests under its authority need to be managed with a much broader perspective than simply maximising wood production: managing the forest for species diversity has emerged as a major policy goal.

Over the last 3 years, a US Forest Service scientist, Joan Walker, and I have been collaborating on reintroducing 14 rare and declining sandhills wildflower species into managed sandhills pine forest. This study is similar to the experiments conducted at the Middlesex Fells, except that we are experimenting with a wider variety of site treatments prior to planting; removing the oak understorey or leaving it intact, digging up the soil, applying nutrient fertiliser, setting down wire mesh enclosures to keep out herbivores (rodents, rabbits, and deer) that might eat seedlings and transplants, and combinations of these treatments. In contrast to the Middlesex Fells sites that are in moist woodland and where water stress is a relatively rare occurrence, these sandhills sites are often extremely hot and dry. As a result, seedlings and adult plants were planted into the site only during periods of rainy weather or were watered during and after the transplantation process.

These experiments on plant reintroductions will be followed over the next decade or more to determine which species have established new populations and which treatments are most successful. These success of a treatment is judged first by the appearance, size and survival of plants, then by the ability of those plants to flower and set seeds. When a second generation of plants is established on the site and the population is stable or expanding in size, the reintroduction attempt can be judged a success.

The preliminary results from the Middlesex Fells and the Savannah River sites show similar patterns. First, seed germination is dramatically enhanced by digging up the soil. Numerous seedlings are often present on quadrats in which the soil has been dug up, in contrast to adjacent untreated control quadrats. Seed germination in not further enhanced by the additional site treatments of fertiliser, enclosure cages, or combinations of treatments. Second, the growth of seedlings is surprisingly slow. Even after two years, most seedlings at the sites remain of a very small size, in contrast with the large plants of similar age raised in the glasshouse. Under natural conditions, these perennial plants may take many years to reach flowering size. Third, in contrast to these seedlings, adult plants transplant into the sites often flowered in their first or second year, producing a new crop of seed. This ability to flower soon after transplanting, illustrates the great advantage of using adult transplants in reintroduction studies. Fourth, about one-third of the species used in these studies have virtually no seedlings germinating the wild, as far as we could detect. In some cases we were unable to germinate the seeds under lab conditions as well, suggesting that the seeds of some species require treatment or special conditions to germinate. Other species, such as spikenard, germinate readily under laboratory conditions, but seedlings are never found in the wild. For many of these rare species, understanding the factors which inhibit and enhance germination in both the lab and the wild will be crucial to reintroducing these species into their former communities. And fifth, certain species responded well to our experimental methodology, having a good rate of transplant survival and a good rate of seed germination in particular quadrats. However, even in these species, other nearby apparently suitable quadrats showed little or no seed germination and poor adult survival. Environmental heterogeneity is clearly a dominant factor in controlling the establishment of new populations. Understanding this environmental variation and developing predictive models of plant distribution using environmental variables are important research priorities.

Conclusion

An exciting development in ecosystem management at the population and community levels is the restoration of a biological diversity on damaged and degraded sites. Where a rare or endangered species has been lost from part or all of its range, attempts can be made to manage sites to restore lost populations or reverse the trend of declining populations. At the community level, entire assemblages of species can be recreated on damaged or degraded sites. Lakes, prairies, wetlands, and forests are communities commonly targeted for restoration. Such attempts at restoration have had an extremely mixed record of success. Initially promising results are often not borne out by follow-up studies conducted several years later. Rare species in particularly often fail to establish new populations at restored sites. Restoring lost biological diversity is so difficult, slow and expensive that every effort should be made to preserve it in existing locations. At the same time, rigorous, quantitative experiments need to be undertaken to determine how to overcome the problems currently encountered in plant reintroduction work. In this way, new populations of rare and endangered plant species can eventually be established in natural and restored biological communities.

3.3 Resilience, Tolerance and Thresholds: Implications from restoration ecology

Krystyna M. Urbanska

Abstract

The concept of tolerance is basically eco-physiological and operates at the individual level. Tolerance threshold may be defined in terms of the organism's existence and functional capacity: should the disturbance exceed the tolerance threshold, the organism dies.

The concept of resilience is both eco-physiological and demographic and operates at the level of population, community and/or ecosystem level processes. The resilience threshold may be defined in terms of unassisted recovery of structure and/or function after disturbance. This recovery may improve the damaged but still existing population but it may often involve a complete re-building of destroyed population from eggs, larvae, or - in case of plants - from persistent propagule bank in soil.

The resilience concept should be reconsidered; restoration ecology offers some worthwhile insights in this respect. The resilience cannot be defined in the context of historical challenges alone, since both the current environment and the extant ecosystems may already have been subtly altered in comparison with the past. A possible cumulative effect of indirect and direct disturbance has also to be considered. Long-term prognoses of unassisted recovery and reference states based on traditional successional concept of floristic relay may, to some extent, be applicable when resilience of productive ecosystems to a degree of disturbance is discussed, but decidedly do not apply to sites damaged beyond the resilience threshold. This situation is particularly clear in extreme ecosystems where the dynamics of the open vegetation is influenced mostly by abiotic factors and interactions other than competition. The new 'nature-in-flow' paradigm which offers a number of reference states seems to be much better suited as a basis for evaluation of future development and restoration planning than the classical principles of stability and climax.

Introduction

A bit of semantics

The terms: 'tolerance', 'resilience' and 'threshold' imply disturbance or hardship of some kind. In the Shorter Oxford Dictionary, 3rd ed. (Onions 1973), tolerance is defined as "action of enduring or sustaining...hardship", or "the power or capacity of enduring or sustaining". The adjective 'tolerant' is accordingly defined as "capable of bearing or sustaining". In the Oxford Advanced Learners Dictionary published one year later (Hornby 1974), tolerance is a synonym of endurance defined as "ability to last, to continue in existence".

Although the work of Hornby gives more emphasis to persistence as the basic advantage of tolerance, the definitions of tolerance are not very different in the two Oxford Dictionaries. On the other hand, there are some interesting nuances as far as resilience is concerned. The definitions in both volumes clearly include return to an original state as an inherent element, but the time factor is treated differently. The definition of resilience (or resiliency) in the 1973 dictionary reads simply: "power of recovery" and it does not include any reference to the speed of that recovery. In the 1974 volume, definitions of resilience include an explicit temporal component ("resilience = quality or property of quickly recovering the original shape" or "condition or power of recuperating quickly").

Threshold may be understood as the limit beyond which no functional capacity is assured. If the threshold of tolerance is crossed, the organism dies; if the threshold of resilience is crossed, no unassisted recovery of the population, community or ecosystem is possible.

Changing times - a need for reconsidering?

A good understanding of the essential elements of tolerance and resilience is much needed nowadays, because our environment is changing under the influence of human activities. Anthropogenic disturbance proceeds, on the one hand, gradually and affects ecosystems indirectly via global change of climate. On the other hand, direct effect of human activities due for example to resource exploitation often results in a rapid, patchy disturbance. For instance, the intensive industrial development in the area of the Kola Peninsula, northern Russia, where the mean annual nickel emission from two local smelters is 3700 tonnes/year, has caused extensive pollution of soil. The territory affected by only one of the smelters is about 3000 km^2 and the primary vegetation has been completely destroyed within ca. 550 km^2 (Balaganskaya and Lysnes 1995). On a more regional scale, total habitat and vegetation destruction resulting from various human activities in NW Siberia includes ca. 2500 km^2 (Vilchek and Bykova 1992). Damage in western Europe due, for example, to urban development or winter sports, may be less extensive spatially, but equally drastic when the ecosystems are destroyed and the topsoil irretrievably lost.

There is an urgent need to repair, at least partially, the environmental damage and the science of restoration ecology, which attempts to give a scientific base to this repair is developing very rapidly. However, the challenge of restoration ecology is not only to develop ecologically correct approaches to the repair work but also to test important ecological theories (Ewel 1987; Bradshaw 1987, 1995; Harper 1987; Urbanska 1996). This goal applies well to the concepts of tolerance and resilience.

Tolerance and resilience should be taken into consideration in assessment of damage and possible unassisted recovery as well as planning, implementation, and assessment of restoration. I propose to consider here these concepts setting them both in a more general context of disturbance and within a framework of research in restoration ecology.

Tolerance and resilience in ecology

Tolerance

Considered in ecological terms, the term tolerance refers to those physiological properties of an organism which permit its existence in spite of disturbance/stress factor(s). The tolerance threshold is usually linear and if crossed, the organism cannot survive (Figure 3.1).

Since tolerance is a genetically controlled feature of the individual, various tolerance degrees may be found within one population (Jowett 1964; Gregory and Bradshaw 1965). Another interesting aspect is that tolerance may be specifically oriented towards a single factor, or it may be multiple. For instance, a given plant may be tolerant to one heavy metal but not to another

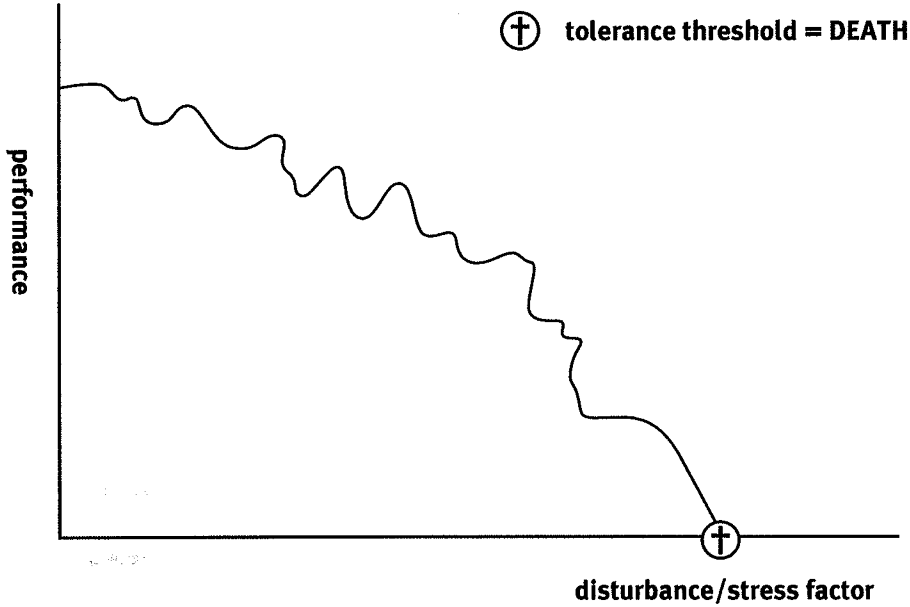

Figure 3.1: An organisms physiological performance decreases until its tolerance threshold is reached (i.e. it dies).

(Neenan 1960; Broker 1963) or it may tolerate various metals. The latter type of tolerance has been directly documented in some studies (see e.g. Prat 1934, Spence 1957, or Gregory and Bradshaw 1965 for data on *Agrostis stolonifera*), but there is also a large body of circumstantial evidence (see e.g. Rune 1951 for data on *Rumex acetosa*, *Lychnis alpina* and other serpentine species).

Tolerance of plants may be revealed not only in direct physiological measurements but also in growth and reproductive patterns. Two contrasting aspects of tolerance are respectively (i) ability to function under stress without loss of extant organs but with a diminished or arrested development of new organs, and (ii) a capacity to regenerate after damage i. e. to replace lost or strongly affected organs.

A good indicator of the first aspect is the root system. Differences in root system development, indicating different tolerance degree to heavy metals in soil, were found in the classical studies of Gregory and Bradshaw (1965) or Clarkson (1966). Relative growth rate, RGR, may be regarded as another indicator (see e.g. Grime and Hunt 1975 for so-called stress tolerators). Differences in reproductive performance also have an important indicative value (see e.g. Wookey *et al.* 1995).

Regeneration at individual level may occur in various forms. For instance, damage-tolerant plants often respond to damage with compensatory regrowth. The pattern of regenerative growth does not seem to be influenced by the nature of damage, rather, it seems to be species-specific. The behaviour of two graminoids occurring together in an arctic site and strongly grazed by snow geese illustrates well those differences: *Puccinella phryganodes* in a grazed site produced significantly more new tillers and leaves than in the ungrazed plot (Bazely and Jefferies 1989), whereas *Carex subspathacea* responded to grazing with a significant increase in number of leaves on the extant tillers (Kotanen and Jefferies 1987).

Curiously enough, tolerance to damage expressed via high regenerative capacity may sometimes result in an increased relative fitness. The well-documented study of Paige and Whitham (1987) on scarlet gilia (*Ipomopsis aggregata*) provides a good example. The authors demonstrated experimentally that under natural field conditions plants can benefit from herbivory effects. When mammalian herbivores removed 95% or more of the above-ground biomass of the iteroparous plants, seed production and subsequent seedling survival averaged 2.4 times that of the uneaten controls.

Tolerance to enforced fragmentation is an important attribute of high-alpine or arctic plants which frequently suffer the damage resulting from rockfall or gelifluxion. Experimental studies of our group consistently revealed that many high-alpine plants are very well able to tolerate a rather extreme damage resulting from single-ramet cloning (Figure 3.2). The tolerance to fragmentation was found not only in graminoids but also legumes and forbs ; the extreme damage apparently stimulated not only a good production of new ramets but often also a rapid flowering (Urbanska *et al.* 1987; Gasser 1989; Tschurr 1990, 1992; Hasler 1992; Wilhalm 1996).

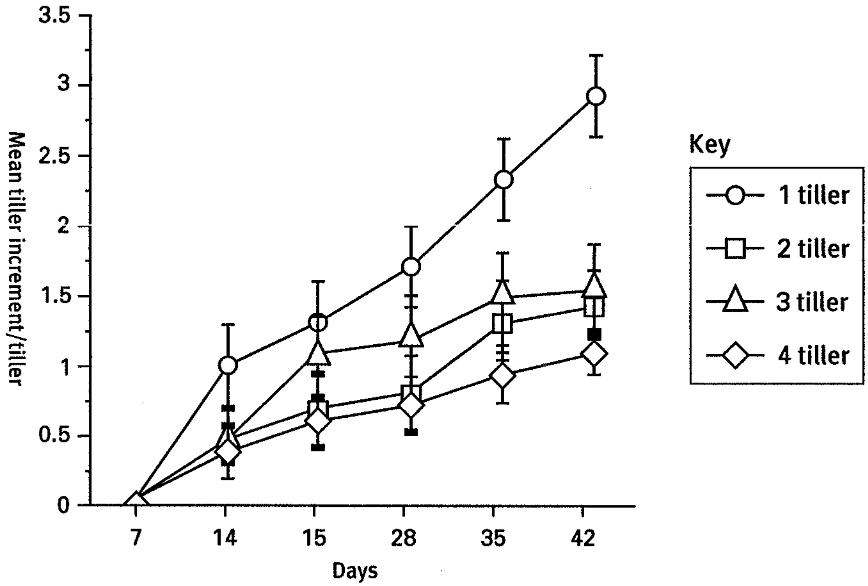

Figure 3.2: Growth of tillers following single-ramet cloning.

Tolerance in plants is very important to restoration. For example, body-damage-tolerance must be known when large amounts of native material are needed for restoration, because only a limited sampling can be done in the donor population so that single-ramet cloning is required. Damage-tolerant ramets should be able to grow well after cloning (see e.g. Urbanska 1995a, b; Keigley 1988). Accelerated flowering is also very important because it permits a rapid population development in restoration site after founders are introduced (see e.g. Urbanska 1995a, b, 1996 for data on *Trisetum spicatum* or *Myosotis alpestris*). Tolerance to herbivory may be a decisive criterion in the choice of plant material for restoration of heavily grazed areas.

Resilience

Ecological concepts of resilience most frequently operates at levels of biological organisation other than the individual viz. populations, communities or ecosystems. Pimm (1991) regarded resilience as the speed with which a system returns to its equilibrium (but see Westman 1991). On the other hand, Holling (1973) referred to resilience as a property of ecosystems that is a measure of their persistence and of their ability to absorb changes and disturbance and still maintain the same relationships between populations or state variables. The resilience of an ecosystem does not depend on its stability i. e. ability to return to an equilibrium state after a temporary disturbance, on the contrary communities may fluctuate substantially, but just this instability confers an enormous resilience (Holling 1973). The ecosystem resilience threshold has been defined by Holling as a limit beyond which a disturbed system will return to a new domain in the vicinity of altered structure characterised by different composition and interactions of species. A resilient ecosystem does not need to include exactly the same populations as before disturbance (see also Denslow 1985).

Resilience thresholds

Similar to the tolerance threshold, the term "resilience threshold" in ecology may be understood as a limit of recovery. However, it should be considered in the hierarchical context of biological organisation and may have different meaning depending on the organisation level. These differences clearly come into view when resilience of population is compared to resilience of community and/or ecosystem.

The resilience of plant populations is based on the tolerance of individuals which form the population. However, there is another aspect to population resilience: even after the extant individuals have been destroyed, many plant populations are able to recover using diaspore reserves in soil (Figure 3.3). If such reserves do not exist, the population resilience threshold has been crossed and the population becomes extinct.

Some populations within a community may disappear but if the resilience threshold of the community has not been crossed, the community is still likely to function. A similar assumption could be made for the ecosystem level: some communities may disappear when their respective resilience thresholds have been crossed but a resilient ecosystem still maintains its function. This may not always be the case if the populations or communities affected are of pivotal importance to the ecosystem and the problem of keystone species (Paine 1969; Bond 1993 but see Mills *et al.* 1993, and Milton and Dean 1995) should be further studied. If the ecosystem resilience threshold is crossed, there is no recovery in the usual sense - only change into another state, or extinction (Figure 3.3). Resilience thus takes a special meaning in conditions of advanced habitat destruction. I shall return to this problem in the following part of the paper.

The resilience thresholds are often linear i. e. approaching with gradually increasing degree of disturbance/stress; however, non-linear thresholds also have to be considered. Their effects,

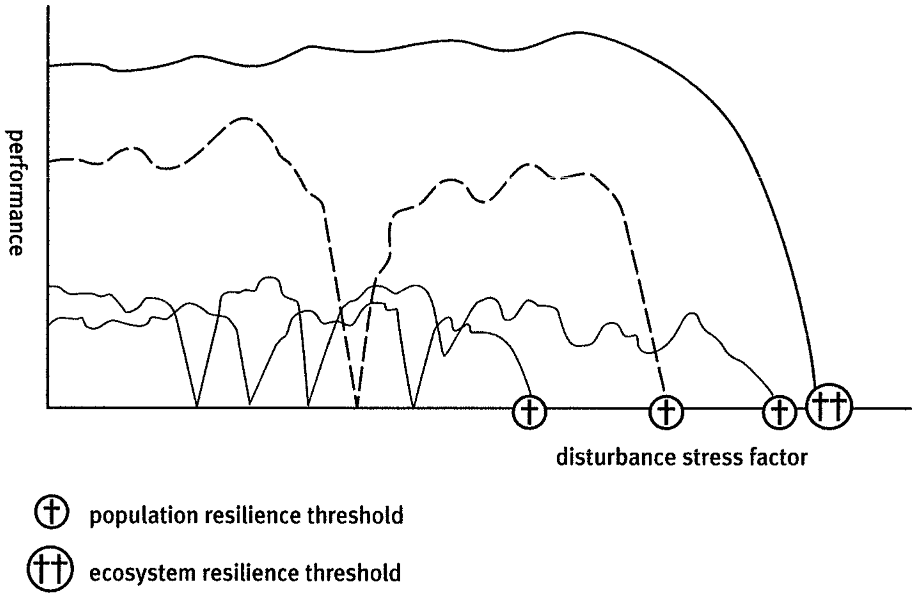

Figure 3.3: Where diaspore reserves exist in the soil, ecosystems can have a higher tolerance to disturbance than individual plants. However, once such reserves used up or where they don't exist, the ecosystem resilience threshold will be reached.

especially those at the ecosystem level, are much more difficult to recognise and to predict as recently reported by Jefferies and his co-authors. These scientists found out that episodic extreme weather events are likely to have a much greater trigger-effect on trophic cascades in arctic coastal ecosystems than average weather conditions (Jefferies *et al.* 1995).

Usefulness of tolerance and resilience concepts in restoration ecology

Concepts of tolerance and resilience are very important in ecological research yet their overall usefulness should clearly be reconsidered in the frame of reference of new ecological findings. The insights from restoration ecology, whose primary research field is a drastically disturbed ecosystem, are helpful in this respect, and the concept of resilience requires a particular focus.

Consider first the extent of damage and resilience. Resilient populations, communities, and ecosystems are able to withstand a degree of disturbance, particularly when the disturbance occurs gradually and on a local scale. On the other hand, even the strongest resilience does not help against a single disturbance event of catastrophic proportions, heavy continuous site exploitation, or frequently repeated, strong disturbance on a large scale. I propose to consider briefly but one example, that of machine-graded ski runs in high-alpine sites.

In the early seventies, the extensive construction of downhill ski runs at high altitudes resulted in removal of the topsoil which was subsequently discarded. Not only the plant cover but also diaspore reserves were thus lost. What remained were erosion-exposed "technogenic barrens" characterised by mineral soil with no horizons and no plant cover (see e.g. Meisterhans 1988;

Flüeler 1992; Urbanska 1995). In many high-alpine areas, spontaneous colonisation of machine-graded ski runs did not succeed, even after ca. 30 years, as it seems that ski run construction destroyed most of the safe sites indispensable to a successful plant establishment (Urbanska 1995, and 1997b). The situation observed in the graded ski runs clearly illustrates the case of destroyed resiliency via the actual habitat (and inhabitants) destruction, irretrievable loss of diaspore reserves in situ, and also elimination of possible safe sites for immigrating diaspores.

The example of high-alpine ski runs might be considered as a relatively "mild" case of ecosystem damage; still more aggravated conditions for unassisted recovery can be easily imagined when the loss of plants and topsoil is compounded by soil toxicity, and no possibility for a colonisation (see e.g. Bradshaw and Chadwick 1980; Bradshaw 1997). The concept of resilience clearly does not apply to dramatically disturbed sites because there are simply no functioning ecosystems anymore, resilient or otherwise.

Another problem with the general usefulness of the established resilience concept is its obvious, but not clearly-defined, temporal component. Both the traditional and more recent definitions are based on the time a variable lasts before it disappears or is changed to a new value, or the speed with which a system returns to its equilibrium. The interpretation of this essential temporal element brings about considerable problems because the time scale is interpreted arbitrarily by various authors in prognoses of unassisted recovery after disturbance and sometimes includes a time-span of centuries. For instance, predictions of Tischkov (1996 and unpublished) on plant cover development in anthropogenic disturbances on Svalbard include several hundred years recovery. Similarly long recovery was diagnosed for some tropical rain forests (Knight 1975 and Whitmore 1975).

A resilience concept based on an excessively long time span and the resulting long-term prognoses are not very helpful at the time being. The current global change of climate is, more frequently than not, reinforced by direct human effect on ecosystems, and land use is likely to be the most significant component of this effect, not only now but also in the foreseeable future (Vitousek 1992). Also, contemporary disturbances are often drastically different from those in the past, both in their magnitude and nature, especially as far as the human influence on terrestrial ecosystems is concerned. For instance, tropical rain forests are, in general, very productive in a natural state and may indeed demonstrate a considerable resilience to local disturbances. But nowadays they are heavily exploited for logging on a large scale, and also there is often also an extensive farming in which the original seed sources and the soil structure are destroyed (Gomez-Pompa *et al.* 1972).

Some authors argue that communities seem to be much more resilient to particular environmental disturbances if they have faced comparable challenges in the past. However, even in the situations where current disturbances are roughly similar to those which occurred in a recent past, no success in direct application of the past data to long-term prognoses for the future is guaranteed. As justly pointed out by Balmford (1996), "contemporary challenges may often be subtly but critically different from historical ones". For instance, a past resilience of ecosystems to e.g. dramatic fluctuations of climate might have been linked to their former overall distribution, and species population size. Nowadays, many of these species have reduced distribution ranges and fragmented habitats; their populations are often small with consequences for gene flow, genetic structure etc. Historical data may thus be considered but predictions based on these data alone (see e.g. Katenin 1995) should not be accepted uncritically.

Pre-disturbance forms an inherent part of the traditional resilience concept. The only paradigm considered a basis for resilience is thus the ecosystem stability, and the principle of succession is routinely used.

It is still not unequivocally agreed whether the ecosystem stability should be principally assessed in terms of species composition i. e. alpha diversity, or should focus rather on species abundance (Rahel 1990, see also comments on ecosystem fragility, Nilsson and Grelsson 1995). In cases of a strong disturbance, the principle of stability does not seem to be always applicable on account of the context (Parker and Pickett 1997). The aim of restoration clearly has its focus on ecosystem function rather than the exact pre-disturbance structure (Bradshaw 1997). The new paradigm of "nature-in- flow" (Botkin 1990; Pickett *et al.* 1992; Pickett and Parker 1994; Parker and Pickett 1997), offering a number of reference states instead of a unique climax, may thus prove to be more suitable in restoration (see also Urbanska 1996).

Succession is supposed to be generally characterised by progression i. e. development from simple to more complex structures with increasing species and community diversity (van der Maarel *et al.* 1985). However, is it not always clear whether a community mosaic really and always corresponds to successional stages or perhaps it may sometimes reflect underlying differences in the physical or chemical environment, as suggested by Roozen and Westhoff (1985).

The model of vegetation dynamics *sensu* Pickett does not need to be directional, stage-wise, or terminate in a repeatable state (Urbanska 1996); for this reason, it may be a better option for interpretation of community development after restoration than the deterministic model of succession. A more general term "vegetation development" suggested by Pickett, might thus be more adequate than "succession", especially in situations where no floristic relay occurs over longer time. Our restoration research in the Alps (see for example Urbanska 1996), and also recent reports from other areas (e.g. Andersen 1995) suggest that the species diversity in restored sites may be similar to that of natural areas but the floristic composition of communities is different. I recently questioned a direct application of the traditional concept of succession based on floristic array to evaluation of restoration in extreme alpine and arctic areas (Urbanska 1997a). Apart from more basic reasons, an unconditional adherence to the succession principle may lead sometimes to terminology which is paradoxical, both semantically and ecologically, e.g. "permanent pioneer stage".

Many readers may disagree with the views expressed in this paper. Fine, discussion on the subject would be most welcome, but before it starts, the obvious should be stated: revision of established concepts and/or terminology clearly does not take away the merit of the pioneer work done by many great ecologists. On the contrary, it clarifies the meaning of numerous concepts. The purpose of the First Sibthorp Seminar is to challenge traditional ecological thinking and to explore the relevance and application of recent scientific advances; the insights gained in the field of restoration ecology call for attention in this respect. Drastically disturbed sites cannot be regarded anymore as just a small number of exceptional cases; there are simply already too many of such exceptions, and their number grows continuously.

Discussion of ecological concepts should not only result in giving them a more precise meaning for purely scientific purposes. Clear concepts are also needed urgently for communication among specialists involved in applied ecology and the ecosystem management. Soulé (1986) argued that "the risk of non-action may be greater than the risk of inappropriate action". I am afraid that the opinion of the well-known conservationist does not apply to many dramatically damaged areas where "trial and error" manipulations (e.g. seeding of exotic plant species with massive fertiliser supplements) often seem to have heavily affected an unassisted recovery process so that new measures are indispensable. Should it not rather be said that "the risk of inappropriate action may be as great as the risk of non-action"?

Conclusions

Human intellect favours connections over disjunctions. It is therefore not surprising that data which appear to generate patterns and interpretations which take those patterns for granted may often be privileged over data and interpretations which call patterns into questions. However, the number of exceptions from established rules seem to be steadily increasing, especially when extreme disturbance is concerned. New approaches, reference states, and a revised terminology are thus urgently needed.

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3.4 What are the Principles for Managing Ocean Ecosystems?

Philip C. Reid

Abstract

The oceans comprise 70.8% of the surface of the world and have very different scales governing their physical and biological variability compared to terrestrial systems because of the contrasting density and viscosity of their enclosing media. An immediate and obvious difference is in the size of the primary producers, trees and grass on land with fixed roots compared to single celled free floating phytoplankton in the sea. A new understanding of the spatial scales of change in the ocean has become possible through the development of new sophisticated global ocean models and from a global coverage of multispectral satellite imagery. Together this new information has helped define the complex ecological environments of the oceans and emphasises the need for special management structures to be established. This paper draws attention to progress that has been made in the subdivision of the ocean's ecosystems and proposes that a quantitative classification system created by Longhurst (1995), which bears similarities to the hierarchical system of biomes defined for terrestrial environments should be adopted as a structure for regional management. The scarcity of information that is available on marine organisms and their habitats is emphasised and at this time a focus can only be made on the pelagos. Special mention is made of the management of fisheries and biodiversity. To evaluate seasonal and year to year variability within the defined Biomes and Provinces, a system of long-term in-situ monitoring needs to be superimposed on the spatial regions. Only two such programmes exist at the present time, The Continuous Plankton Recorder Survey and the CalCOFI Survey off California, USA. Developments in new instrumentation that could be used in such surveys are outlined. It is proposed that existing or planned Regional Marine Conventions such as OSPAR, Helsinki, Barcelona and CCAMLR could provide a framework within which the holistic management of marine ecosystems could be embraced. The Intergovernmental Oceanographic Commission (IOC) is encouraging coastal states to develop regional approaches to management of coastal seas where these do not already exist. With time a global coverage of regional conventions should exist to implement the UNCED declaration on the oceans.

Introduction

At 361.059 million km^2 and a volume of 1370.323 million m^3 the oceans comprise 70.8 % of the surface of the world. The deepest trench extends to a depth of 11,034 m which compares to Mount Everest at 8,848 m high. Because of the difficulties and expense of sampling such large spatial and depth scales knowledge of the pelagic environment of the top 200 m is greatest, but is still very limited compared to terrestrial systems. The vast majority of species have not yet been properly described and the ecology of those that have is poorly understood. While some representative pelagic ecosystems and other systems such as coral reefs have been defined, knowledge of the processes involved and their extent and geographical 'boundaries' is limited. Information on the planktonic organisms living in deeper water and their role in the breakdown of sinking organic detritus is very limited. The benthic organisms living on and in the bottom are dependent for their nutrition on a rain of detritus from the plankton which may be highly seasonal in its timing. They are structured by depth zones but are likely to reflect patterns of productivity seen in the overlying pelagos.

The physical basis of terrestrial and marine ecological systems differs greatly (Steele 1985). The large heat capacity of the oceans damps out short term variability in the 'climate' of the oceans compared to land. Because of this thermal response and the long exchange rate between deep and near surface water (10^2 - 10^3 years) large amplitude environmental changes may occur at long time intervals in the oceans. Thus to quote from Steele "at very long time scales the system is inherently unpredictable and must be considered in the evolutionary, rather than the ecological, context". This latter observation has important management and risk analysis connotations.

Different ecological strategies have been evolved by marine and terrestrial organisms to respond to their contrasting environmental forcing. For example, poikilotherms dominate in the relatively constant conditions of marine systems. In a further distinction marine primary producers only have short life cycles (~ one day) compared to trees and grasses on land. Perhaps most fundamental is the contrast in size structure against generation time seen in the two systems i.e. phytoplankton with a length of ~ 40 µm and a generation time of one day against trees with a length of 108 (m and a generation time of 104 days, (figures 1-3, Steele 1991). Studies of marine systems have emphasised that physical processes govern population variability in contrast to terrestrial ecosystems where density dependent, predator/prey interactions are thought to be the most important forcing parameter. The oceans and seas of the world are thus a very different ecological environment and their ecosystems will require the establishment of special management structures.

All planktonic living matter in the oceans is in continuous motion in a viscous and turbulent aqueous system that is part of a long-term, circumglobal circulation of sinking cold water at the poles and returning easterly flowing warm surface water at the equator. The main surface currents are forced by dominant wind regimes such as the trades, westerlies and monsoons. These currents show complex eddy structures in satellite imagery. At continental margins, dominated by strong offshore winds, upwelling of nutrient rich deep water may occur. These areas are the most productive regions of the oceans. On the shallow (< 200 m) shelves (and adjacent seas) of the continents, which only comprise 7.6% of the surface area of the world, tidal currents create a further agitation of the water. As a response to seasonal surface heating and the input of 'lighter' less dense freshwater inputs a density layering (thermocline, pycnocline) forms within the upper layers. This layer acts as a boundary and limits transfer of nutrients from deeper water. Primary production is confined to only a small portion of the ocean's volume as light only penetrates to ~ 100m depth, further reducing its penetration in polar and turbid coastal waters.

The oceans thus exhibit complex vertical and horizontal patterns of motion and all planktonic ecosystems are mobile and have a continually changing structure and boundaries. The dominant features are forced by an atmospheric circulation that appears chaotic. How climate change will affect oceanic circulation is far from clear and management at an oceanic scale is totally impractical until the forcing parameters of climate change and anthropogenic influence of this change through the production of greenhouse gases are understood and harnessed.

Given the above quotation from Steele, developing a full understanding of the processes governing oceanic circulation is a matter of urgency as any major change must impact marine and terrestrial systems.

An approach to ecosystem management of the oceans needs, in the first instance, to define the areas to be managed. The great variability in physical structure seen in satellite imagery and the latest generation of global ocean models emphasises that the oceans cannot be managed as one entity. It is proposed that a hierarchical classification of oceanic ecosystems, based on differing seasonal cycles of primary production, is established which can be compared to the biomes used in terrestrial systems. This could form the basis for a geographical breakdown of management regions. A long-term monitoring programme should be superimposed on the ocean Biomes or Provinces to establish seasonal, interannual and long-term changes in pelagic systems (IOC and SAHFOS 1991). Some single site vertical profile stations should be occupied with a network of continuous plankton measurements along standard ship of opportunity routes using instrumented CPRs or a new generation of Continuous Plankton and Environmental Recorders, CPERs, (Quartley and Reid 1996). Regular (5 yearly) assessments of changes in the different Provinces/Ecosystems could be undertaken as part of the remit of existing Regional Sea Conventions, within Large Marine Ecosystem studies (Sherman 1994) or under the umbrella of the UNEP Regional Sea Programme. Implementation of recommendations from the assessments would be the responsibility of signatory national and supranational (e.g. The European Union) authorities.

Satellite Imagery

The advent of satellites and sophisticated mathematical models has revolutionised our understanding of oceanic variability. Primary production is the source of carbon (with the exception of mid ocean ridges and similar methanogenic/sulphur systems) for all living organisms in the sea. Between November 1978 and June 1986 the Nimbus 7 Satellite carried an experimental multispectral 'Coastal Zone Colour Scanner' (CZCS) which measured backscattered radiance at 433, 520, 550 and 670 nm (Feldman *et al.* 1989). This instrument only operated for a short period of time and there were difficulties in developing algorithms to differentiate between chlorophyll, suspended sediment in shallow water and Dissolved Organic Matter (DOM). A large number of images have been processed to produce a composite global view of chlorophyll distribution on a world scale from both CZCS and the successor programme from September 1997 Sea WiFS. A set of composite images can be downloaded from the NASA world wide web site. The images display high phytoplankton biomass near the poles, in temperate west wind zones, in upwelling regions and along the equatorial counter currents in the Pacific and Atlantic oceans. Seasonal composites for the North Atlantic show a northwesterly extension of chlorophyll maxima in the Spring with a later autumnal regression to the south. AVHRR imagery, primarily used to measure changing sea surface temperature can also reveal coccolithophore blooms from their white reflectance, (Holligan *et al.* 1989). CZCS, Sea WiFS and AVHRR imagery clearly demonstrates the great spatial variability of marine ecosystems, the large contrasts that exist between different oceans and the extensive seasonal changes that occur. The images have provided a new set of tools that can be used to develop a classification of the world's oceans.

Biogeographic Classification

A management approach to oceanic ecosystems will need to represent the full diversity of the fauna/flora and take note of unique, endemic or special characteristics. As a first stage a classification representative of the different geographical regions of the oceans needs to be established with a superimposed monitoring strategy to evaluate changes with time. Periodic regional assessments should follow as part of a management cycle. Proposals are under development to establish an operational Global Terrestrial Observation System GTOS and a Global Ocean Observation System, GOOS (IGBP 1992, OECD 1994) that will embrace aspects of this structure. Terrestrial ecologists have divided the continents into a number of 'Biomes' that are representative of different vegetation types, e.g. Tundra, Chaparral, Grassland, Tropical Rainforest, Temperate Evergreen Forest, Taiga, Desert, Mountain Zones, Temperate Deciduous Forest and Polar Ice. Research has demonstrated that the boundaries between these biomes have a physical basis and can be defined by a combination of a limited set of measurements such as temperature, slope, elevation, latitude, solar radiation, rainfall and soil type. The advent of satellites has greatly facilitated the regional mapping of these biomes as a basis for modelling production and carbon flow.

Longhurst (1995) has recently developed a classification of the worlds oceans that is based on planktonic plant biomass and production and can be considered as equivalent to the terrestrial biomes. His system was established on a knowledge of the physical features of the oceans and how physical processes influence the growth of planktonic algae. To define the seasonal phytoplankton growth cycle he compiled global data sets, at a resolution where possible, of one degree latitude x one degree longitude for: mixed layer depth, temperature, density, surface nutrient concentrations and solar irradiance. Surface chlorophyll fields on a one degree grid were obtained from monthly mean CZCS satellite data for the period 1978 - 1986, again on a one degree grid. Photic depth, primary production integrated for photic depth and a phytoplankton loss term were calculated from the above variables using a mathematical model (see Sathyendranath *et al.* 1995). With the above information Longhurst (1995) proposed a division of the oceans into four Biomes and 56 Biogeochemical Provinces. Characteristics of the Biomes are as follows (after Longhurst);

Westerlies Biome:	Regions with a mixed layer depth that is controlled by local winds and irradiance.
Trades Biome:	Regions where mixed layer depths are adjusted at ocean wide scales, often by distant winds.
Polar Biome:	Regions influenced by a surface brackish layer from ice melt (includes Polar Coastal).
Coastal Biome:	Shallow regions (< 200 m) dominated by the effects of coastal processes on mixed layer depth.

Once the varying seasonal boundaries of these Biomes and Provinces are established they could form a quantitative basis for the management of marine systems, including primary and secondary producers, fishery carrying capacity and biodiversity.

Coastal ecosystems are most under threat from habitat degradation, pollution, eutrophication and overexploitation of marine resources. Close to 95% of the yield from the fisheries of the world is taken from coastal waters. Some of these fisheries, especially in upwelling regions, have shown a rapid change in the dominant pelagic species, termed 'biomass flips' that have had major economic and biological effects because of reductions in yield. The true causes of these changes are not fully understood, but may be a combination of overfishing, natural environmental change and possibly climate change. To tackle the problems of coastal environments a qualitative division into approximately 50 Large Marine Ecosystems (LMEs) on the basis of their similar bathymetry, hydrography, productivity and trophically dependent populations has been proposed (IOC 1993). These LMEs were defined specifically to enable a regional, multinational approach to be applied "to research, monitoring and stress mitigation". The proposal recognised that the management of LMEs is an evolving scientific and geopolitical process, but felt that enough information was available to understand some of the driving forces behind the large scale changes in the biomass yields and long-term sustainability of LMEs. Longhurst's approach means that a quantitative structure is now available for LMEs and that it should eventually be possible to determine the fishery carrying capacity of each LME.

Zooplankton

The ecosystem classification of Longhurst is based on relatively coarse estimates of algal biomass. It is known that the type of zooplankton present may greatly influence phytoplankton composition and levels of primary production achieved, and yet our knowledge of the zooplankton grazers is at a much lower level. Longhurst (1985) synthesised information on global variability in the relative contributions of different zooplankton for 14 different regions of the world. Longhurst's figure 1 (1985) emphasises the marked differences that are found in 'average' zooplankton composition in different parts of the world's oceans. A vastly improved knowledge of the ecology and characteristic seasonal succession of zooplankton in the Biomes and Provinces identified by Longhurst is urgently needed for any management system. Recognition of their central role within marine ecosystems as grazers and the main food source for fish larvae and pelagic fish is being addressed in a number of national programmes under the international GLOBEC umbrella, (GLOBEC 1997).

Long-term Monitoring

Long-term monitoring covering all the major Provinces identified by Longhurst is an essential requirement in any developing ocean management system. Tinker (1994) has defined the following uses for a monitoring network:

- To parameterise and validate ecological models of biomes.
- To detail changes of value to governments in the development of policies and regulations.
- To provide information to include in socio-economic studies and models.
- To monitor changes in biodiversity.

The value of these exercises will increase as each years' data are accumulated, in the first instance providing a baseline against which natural and anthropogenically forced change can be evaluated. A continuation of standard quantitative measurements through time is needed to establish trends, cyclic phenomena and lag effects from environmental forcing. The results can be used to test hypotheses and as input to mathematical models. Monitoring plankton changes may provide an early warning of climatic or contaminant impacts, as the plankton are integrators of a wide range of forcing parameters (e.g. Taylor 1995). The objectives of any programme need to be well defined and the strategy and equipment to be used well thought out before any programmes start. There will need to be a continual reappraisal of these objectives as long-term programmes are not favoured by funding agencies. As Duarte *et al.* (1992) has shown many long-term monitoring programmes are started, but few survive more than four years.

Only two extensive and long-term programmes sampling zooplankton (CPR and CalCOFI) exist that can define seasonal and interannual variability. The CalCOFI Survey off the west coast of California has sampled zooplankton monthly since 1948 by vertical tows at standard sites along a series of transects out from the coast. Species have not been identified in these samples, settled biomass volume only is measured. The results have demonstrated, however, a marked downward trend in abundance (Roemmich and McGowan 1995). This sizeable change emphasises the need to establish long-term monitoring programmes at key locations within each of the oceanic Biomes and especially in the southern oceans. These programmes are needed to determine if the observed decline also occurs in the southern hemisphere and could be caused by climate change.

The Continuous Plankton Recorder Survey which started in 1931 is the only basin wide Survey in the world which provides quantitative counts of zooplankton (and phytoplankton) abundance at intervals of 10 nautical miles, (Gamble 1994). The machines used in this Survey are towed by merchant ships on their normal route of passage. Close to 200,000 samples have been analysed since the Survey started, identifying up to 400 different taxa of zooplankton and phytoplankton. The Survey has demonstrated consistent patterns of spatial and temporal change in the plankton over large spatial areas and long periods of time. A general downward trend in abundance in the CPR data shows similarities to the CalCOFI results (Reid and Hunt 1998). The importance of density independent factors in governing plankton abundance has been confirmed by recent results using CPR data (Planque and Ibanez 1996; Taylor 1995). The CPR approach to monitoring plankton in situ by using voluntary ships of opportunity has been identified as a way forward in the plans to establish a global operational observation system within GOOS, (OECD 1994).

New Technology

To assess and measure in the future the oceans at the spatial and short temporal scales used by Longhurst will require the development and deployment of a range of new instruments. Because of the high cost of time on research vessels, towed systems operated by ships of opportunity provide the best platform for this approach. New towed bodies are now available that can operate at fixed or varying depths (Aiken *et al.* 1995, Quartley and Reid 1996) and resemble the space shuttle in being able to carry a wide range of instruments. The data can be logged internally in the machines for later retrieval or passed up a cable in real time mode to an onboard computer that could also transmit via satellite to a shore laboratory. The first results are being obtained from such Continuous Plankton and Environmental Recorders

(CPERs) which also carry 'continuous' plankton sampling mechanisms. After appropriate intercomparisions it should, in time, be possible to integrate the measurements from these new machines with the existing CPR dataset. Fixed moorings that can take vertical profiles of ecosystem variables and associated instrumentation to measure key physical and chemical parameters are also under development or being deployed in pilot form and are likely to contribute important new information for oceanic management.

Fisheries Management

Management of marine living resources within the 200 n miles limit is the responsibility of nation states or multinational entities such as the European Union under the United Nations Convention on the Law of the Sea (UNCLOS). This convention contains specific provisions for the sustainable management of fisheries, for the protection of habitats and species and for the protection and preservation of the marine environment from pollution (Ducrotoy 1996). UNCLOS also covers fish stocks and marine mammals that migrate between Exclusive Economic Zones and the open ocean. However, how all these varying requirements should be enacted, which international organisations should have responsibility and what management system should be put in place is not clear. Whaling is covered by the International Whaling Commission and some other migratory stocks by specific conventions, such as the North Atlantic Salmonid Convention. The UN is developing an 'Agreement on Straddling and Highly Migratory Fish Stocks' and FAO is preparing a 'Code of Conduct on Responsible Fisheries'.

The oceans are the world's last true wilderness and yet even here, because of the mobility and shoaling characteristics of many species, man has exploited a number of fish and marine mammals to near or real extinction. World-wide fish stocks have declined dramatically (FAO, 1995, House of Lords SCST 1996) with associated huge changes in the contribution of different species to stocks. For example, fish landings in the North Sea in 1988 (North Sea Task Force 1993) totalled 2,680,705 metric tonnes, a tonnage which was largely made up of small species caught in an industrial fishery, a considerable change from the post World War II fishery for pelagic herring that were used directly for human consumption. Discards (generally undersize fish that are thrown back for regulatory reasons) of fish and benthic organisms (most of which die) add a sizeable additional tonnage. At the present time between 30-40% of the biomass of the North Sea stocks is caught each year. In terms of carbon flow in the ecosystem one would expect a density dependent effect on the planktonic food resources of these fish. Natural mortality is of course high and despite the scale of the exploitation there is no clear evidence for any negative or positive feedback from lower down the food chain.

Existing fisheries management in most areas of the world are established on a system of 'Total Allowable Catches' (TACs) which are based on stock assessments for stocks of individual species. In reality fisheries are based on multispecies catches; and existing models are not capable of recommending catch quotas for multispecies stocks. In many fisheries a large proportion, (in some over 50%) of the catch comprises industrial species with a 1-2 year life span; the impact this has on longer lived more valuable species is currently impossible to determine. Of the 103 species for which quotas are set in Europe it is recognised that 'sufficient scientific data' is only available to set safe catches for 39 species, (House of Lords SCST 1996).

Existing systems of management do not take into account the fishermen, their boats and the catching ability of the fleet. A decommissioning system has been operating under the EU Common Fisheries Policy. Any reduction in fishing pressure that may have arisen was negated by the use of new larger boats equipped with sophisticated electronic fish finding gear. A European Commission Fisheries Policy based on natural quotas has proved unworkable. For sustainable fisheries a 'Common Fishery Protection Policy' with a precautionary approach to management is needed. This policy needs to be based on an integrated management of the fishing ability of the boats in the fishing fleet and estimates of stock size of the different species to be exploited. Sustainability of all stocks including sharks, rays and other by-catch species needs to be guaranteed and incidental capture of other organisms should be limited by selective restrictions on time and place of fishing. It is recognised that deep ploughing by beam trawls and discards may be impacting the benthos (North Sea Task Force 1993). Approaches to reduce discards and the impact of beam trawls need to be developed. Establishing permanent reserves and no fishing zones that may vary in size at different times of the year should be considered as well as gear and net developments as possible ways forward.

Biodiversity

UNCED 1992 recognised a basic lack of knowledge of the diversity of living organisms and the urgent need to develop strategies to tackle this problem. A Global Biodiversity Assessment (Heywood 1995) has reviewed the problem and defined four different types of biodiversity, ecological (biomes to populations), organismal (kingdoms to populations), genetic (populations to nucleotides) and cultural diversity. The report recognises the limited knowledge that exists for the world's biomes and ecosystems and that no single system of classification exists to measure ecological community diversity. Our knowledge of the biodiversity of the seas is far less than for terrestrial systems, but it is less threatened by man's activities. In the marine environment so far exploitation has been limited to larger species, fish, shellfish and marine mammals. A focus has recently been placed on the effects of introduced species. Some algal blooms/toxic algae are thought to have been introduced via resting cysts in ships ballast water. Other apparently introduced species may dominate their new environment and totally change the local ecosystem, e.g. in the Black Sea. The extent to which these events are introductions or a response to changing environments from human activities e.g. eutrophication is still not clear. A strategy for marine biodiversity should be integrated into management structures that are being developed for regional seas.

Regional Conventions

Regional Marine Conventions and areas covered by the UNEP Regional Seas Programme provide a framework under which the holistic management of marine ecosystems could be embraced. The Helsinki Convention on the Baltic includes a responsibility to assess biodiversity and nature conservation and a new Annex on species and habitats was added to the OSPAR Convention (north east Atlantic) in 1998.

Problems of managing the world's oceanic ecosystems are further compounded by socio-economic factors. Only a relatively small part of the oceans and seas are within the control of coastal states out to 12 n miles and to 200 n miles under the Law of the Sea, (OECD 1994).

Very few states have the ability, or in many cases inclination to manage these 'national' waters. Biota and ecosystems, because of their mobility, do not respect these national boundaries; a population that might receive protection from one state may pass into the waters of another where it is not protected. Some countries, such as Japan, place a greater emphasis on the exploitation of marine resources and will travel great distances to harvest high value products.

Reid (1994) described how North Sea Ministerial Conferences, their associated preparatory groups and the North Sea Task Force (NSTF) proved to be an effective management tool which produced substantive progress on measures to improve the North Sea environment. While these improvements focused on e.g. reductions in nutrients and pesticides and termination of dumping at sea, the Ministers did establish procedures to protect habitats and species and recognised a need to identify an ecological network of habitats and initiate long-term monitoring of habitats. The management cycle for the North Sea has been repeated through four North Sea Ministerial Conferences in 1984, 1987, 1990 and 1995 (Reid in press). Under the revised OSPAR the successful management procedures used in the North Sea will now be applied to the whole of the north east Atlantic, stretching from Gibraltar in the south to the North Pole and from the Kattegat in the east to 430W by the year 2000.

Conclusion

The UNCED declarations on the oceans, summarised below, identify an urgent need to develop sustainable management systems for the world's oceans and seas. UNCED also recognised the general importance of capacity building, as well as the important linkage between monitoring and the achievement of marine resource development goals.

- Prevent, reduce and control degradation of the marine environment so as to maintain and improve its life-support and productive capacities.
- Develop and increase the potential of marine living resources to meet human nutritional needs, as well as social, economic and development goals.
- Promote the integrated management and sustainable development of coastal areas and the marine environment.

Regional sea conventions such as OSPAR, Helsinki, Barcelona and CCMLR are the most appropriate bodies to take these declarations forward as part of an integrated management approach since many of the decisions that will need to be taken will have political and resource connotations. Management approaches to reduce pollution/eutrophication, to maintain biodiversity and to conserve habitats are already in place for some of the regional conventions. While considerable progress has been made within existing conventions there are still many problems to be overcome, for example, it is not yet possible to assess the impact of the very many contaminants entering the marine environment as a cocktail. New biological effects techniques are being developed in an attempt to overcome this problem and are being applied to provide an alarm system and to help prioritise action. The management of marine living resources out to the 200 n miles contour of coastal states is a national responsibility and not currently covered by regional conventions. Many coastal states do not have the resources to implement protective management schemes and there is a need for a regional approach to be applied to UNCLOS provisions on the sustainable use of living resources.

At present only a few regional conventions have been established with responsibility for protection and preservation of the marine environment. Until recently most of these conventions concentrated on pollution issues. For areas of the ocean not covered by conventions, UNEP co-ordinates a Regional Seas Programme. The activity and resources devoted to different regions under this programme varies greatly. There is a need for the Regional Seas within the UNEP programme to formalise their role as Regional Conventions. Where Large Marine Ecosystem (LME) programmes are being implemented for regional seas these could form the basis for incipient regional agreements that may later be developed into Conventions. When new regional conventions are being developed they should build on the experience gained by existing equivalent organisations.

As new conventions are established they should have realistic natural boundaries that should take note of the classification progressed by Longhurst (1995). The use of ships of opportunity and Continuous Plankton and Environmental Recorders, combined with information from satellites is seen as a cost effective method of acquiring long-term and spatially extensive information on such provinces. All regional conventions should have full responsibility for the management and protection of their defined marine environment in a sustainable way and contain specific provisions covering habitats, species and their conservation, water quality, coastal management and living marine resources. Mechanisms to liaise with relevant global conventions should be included. A management cycle including a well planned monitoring programme and the production of regular assessments at ~five year intervals should be integral to each convention. If it were possible to establish standard reporting dates for all regional assessments world-wide an integrated global assessment could be made available at regular intervals.

4. UNDERSTANDING ECOSYSTEM FUNCTIONING AND REVIEWING WHAT IS OR SHOULD BE MANAGED OR CONSERVED AND HOW SCIENCE CAN HELP

4.1 Ecological Science and the Management of Terrestrial Ecosystems

Phil Ineson

Introduction

Man has had a dramatic impact on the Earth, with the consequences of his activities being detectable at both the local and global scales. If the management regimes imposed by man are to be sustainable at both these scales, then a basic understanding of the ecological impacts of these activities is vital. Additionally, unless this basic knowledge is incorporated into management at the local scale, then the scientific information may be of questionable value.

Evidence of the need for such information, and the speed of the political and social changes which may result from it, is not lacking; for example, the formulation of a sound scientific case has resulted in dramatic changes in the global use of chlorofluorocarbons. Regrettably, there are many more examples where scientific doubts, or the inability of scientists to convince 'managers', has resulted in inactivity, to the detriment of both society and the ecosystems over which man has stewardship. I provide the worked example of 'acid rain', below.

At the heart of any discussion of the value of science to ecosystem management is the question of what we consider to be 'science'. There are many philosophical and practical works devoted to just this question but the working definition which I find most useful is 'the formulation and testing of hypotheses'. The origins of ecology lie within the study of 'natural history', which has lead to a science which leans towards observation, rather than to the translation of these observations into testable hypotheses.

The importance of the pioneering work of such groups as Likens *et al.* (1977) must be recognised in having brought ecology into a new era of experimental manipulation at the ecosystem scale. However, some commentators fear that the trend to experimental manipulation in ecology has gone too far, to the detriment of basic observation and pattern analysis (Lawton 1996). There has to be both approaches; without careful observation, hypotheses may be flawed or irrelevant; without the testing of hypotheses, ecology cannot be considered a science. Unfortunately, the scientific approach is frequently confused with the use of technology, and the quantification of any ecosystem parameter easily becomes labelled as 'ecosystem science'; a pH meter reporting to several decimal places does not suddenly render the measurements 'scientific'. Similarly, the use of principal components analysis does not make the observation stage any more 'scientific'; all too frequently they substitute for the second phase of the scientific process, becoming the 'last resort of the man with no hypothesis'.

Managers will not take action on the basis of the subjective suspicions of ecologists - they require hard evidence, together with information on the costs and benefits of available options. Unfortunately, the evidence required to prove cause and effect for environmental change or damage is frequently difficult to obtain, particularly when interested parties may prefer the case to remain unproven. The need for a sound scientific footing becomes clear and, although the refereed scientific publication is central in this respect, it necessarily begins to become less accessible to the manager or layman.

In order to demonstrate how the results from basic scientific research can, and are being incorporated into the management of ecosystems, a review of the progression from the 'acid rain' debate from the middle of this century to formulation of critical loads for acidifying pollutants is provided. However, first it is necessary to get things into the right, global, perspective.

Global perspectives

Atmospheric composition

The chemical composition of the Earth's atmosphere has not always been as it is today; in addition to the recent alterations due to anthropogenic activities, the most dramatic changes occurred as a result of the development of life on the planet (Table 4.1). This table shows that the development of the biosphere is linked to major alterations in the concentrations of all the major atmospheric constituents and that this has had profound consequences for the physical environment of the Earth. For example, if the 'greenhouse gas' CO_2 was absent from the atmosphere then the surface of the earth would be frozen (Ramanathan 1988); if the O_2 composition were to decrease from today's 21% to below 15% then fires would simply not ignite; at O_2 concentrations above 25% it would be possible to readily burn wet organic matter (see Schlesinger 1991). The current composition of the atmosphere must be maintained within important, and seemingly fragile, limits, if life is to continue on the planet as we know it.

The composition of the atmosphere is in a state of dynamic tension, with CO_2 providing a clear example of the impact of man, through industrialisation and land-use change. The rapid increases in CO_2 concentrations to the current levels of around 0.035% are set against a background of marked seasonal and latitudinal variation (Conway *et al.* 1988), tensioned by the interplay between photosynthesis and respiration at the global scale. The anticipated rise of CO_2 concentrations to 0.060% by the middle of next century gives cause for concern when set

Table 4.1. Some characteristics of the atmospheres and environments of Mars, Venus and the Earth. Data are from Owen and Biemann (1976), Nozette and Lewis (1982) and Lovelock (1989).

	Mars	Venus	Earth (before life)	Earth (now)
Surface Temperature(°C)	-53	474	240 to 340	16
Atmospheric pressure (bars)	0.007	92	60	1
ATMOSPHERIC COMPOSITION				
CO_2	95	96.5	98	0.035
CH_4 (ppm)	0.0	0.0	0.0	1.7
N_2 (%)	2.7	3.5	1.9	79
O_2 (%)	0.13	trace	0.0	21
Ar(%)	1.6	70.0	0.1	1.0

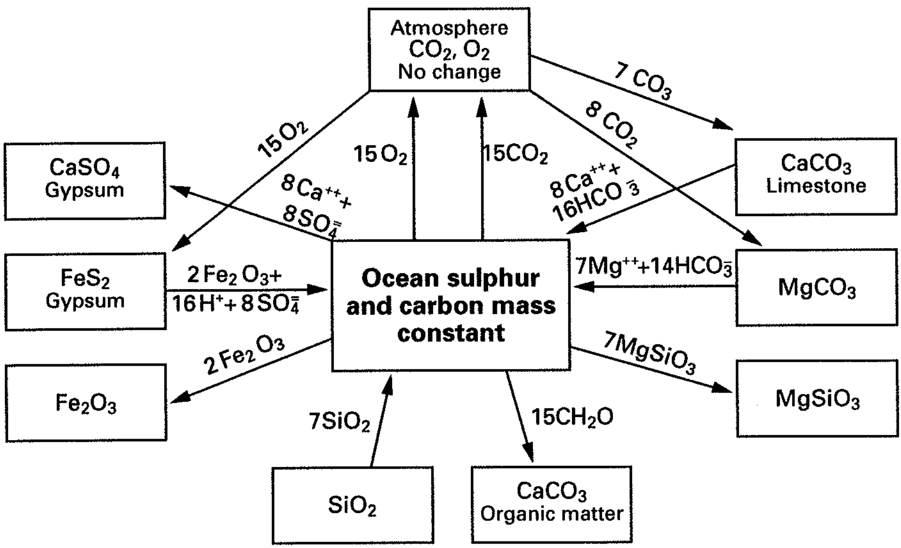
in the context of the data presented in Table 4.1, and the stated importance for global temperatures of the difference between 0.000% and 0.035%. The full implications of doubling these concentrations can only be currently guessed at (IPCC 1995).

A closed system

Since the Earth is basically a closed system, with the exception of the movement of a few meteors, missiles and an occasional light atom across its boundaries, we have a framework for budgeting the movements of elements around the planet. Garrels and Lerman (1981) presented a model of the sedimentary reservoirs and transfers in a model of the surface of the Earth, and Figure 4.1 provides a representation of this model. The transfers shown are the consequences of increasing the biosphere store (CH_2O) by 15 moles. The key assumption in this figure is that the pools of major elements in the oceans and atmosphere have not experienced significant changes, when compared to transfers between crustal minerals. The consequences of this assumption are reflected in the transfers outlined in Figure 4.1, with the C and O bound into organic matter being largely derived from the carbonates of magnesium or calcium. Since these transfers necessitate the release of Ca^{2+} from limestone into the oceans, and the ocean contents remain roughly constant, this results in the deposition of gypsum and the mobilisation of pyrites; the underlying message is that the Earth is a closed system and that adherence to the Laws of Thermodynamics necessitates the chain of reactions illustrated in Figure 4.1.

It is clear from such an analysis that any major anthropogenically-driven transfers will lead to similar 'knock-on' effects for other elements, and this is all too clearly demonstrated by the data arising from the scientific study of the phenomenon of 'acid rain'.

Figure 4.1: The major pools and transfers in the biogeochemical cycles on the surface of the Earth. Example transfers are given for an increase in the biosphere of 15 moles (from Garrels and Lerman 1981).



The acid rain issue

Global sulphur cycle

The term 'acid rain' was first coined more than 100 years ago by the British chemist, R. A. Smith, when describing the air and rainfall quality around the industrial city of Manchester (Smith 1872). Despite these early warnings, the atmospheric transport and deposition of pollutants was largely ignored until the 1950s when environmental chemists again began to conclude that sulphur (S) derived from industrial areas was affecting natural systems, both close to source and across national boundaries. The subsequent work of a number of Swedish scientists led to a report which was presented to a UN Conference on the Human Environment in 1972 (Sweden's Case Study 1971), condemning S pollution and subsequent acidification as one of the major international pollution problems facing the industrialised world.

Despite much initial scepticism the ecologists of the world began to piece together an environmental jigsaw of gigantic proportions, revealing the large scale of impact due to acidification resulting from acid rain. Strong evidence, including many elegant studies in lakes (see Batterbee *et al.* 1990) and forests (Hallbacken and Tamm 1986), was assembled on both sides of the Atlantic and led to the frightening conclusion that, through the large-scale combustion of fossil fuels, man had unwittingly damaged many of the natural ecosystems of the northern hemisphere. The true scale of the problem can best be appreciated when the global S budget is considered, and the scale of anthropogenic influences placed in perspective (Granat *et al.* 1976; see Figure 4.2).

One of the features arising from this synthesis was the importance of the anthropogenic emissions within the global transfers; fossil fuel represents a massive transfer of S in global terms, overshadowing volcanic emissions and matching global weathering transfers (Figure 4.2).

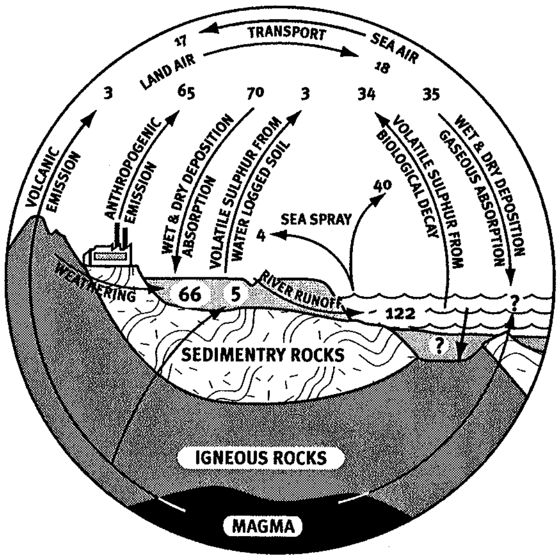


Figure 4.2: Major transfers in the global sulphur cycle (Tg S a⁻¹). Redrawn from Granat et al. 1976.

Man's activities have effectively reversed the S cycle, resulting in a contemporary net transfer of S from land to sea, in direct contrast to the net transfers occurring prior to the industrial revolution.

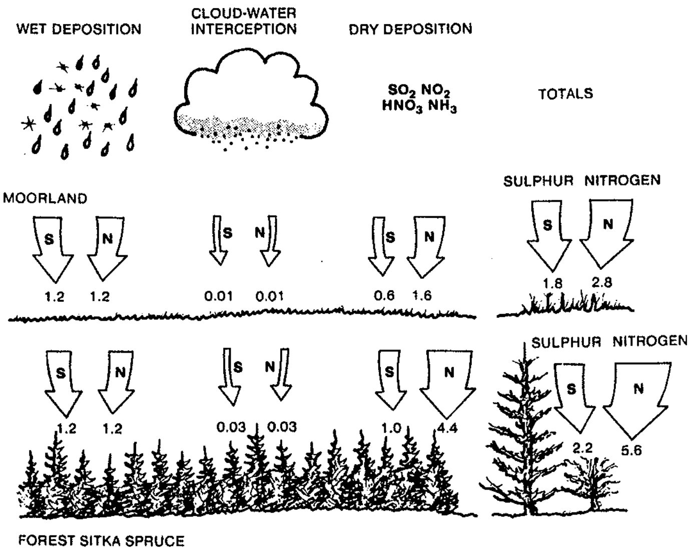
Clearly, when considering the world as a closed system (see Figure 4.1), such enormous transfers must give rise to consequences for the cycling of other elements. During the scientific evaluation of these effects, several fundamental and important biogeochemical principles were clarified, which now assist us in our understanding and scientific management of ecosystems. It is these kinds of principles which need to be communicated to managers, if knowledge generated from scientific research is to be integrated into environmental management. As an example, four of these principles are briefly outlined below, and are subsequently applied in a sample case studies.

Principle 1: The importance of the plant canopy

During the construction of detailed S budgets for terrestrial ecosystems it became obvious from the assembled data that these budgets rarely balanced, requiring revised estimates for S inputs to many ecosystems. Earlier work had identified the importance of tree canopies in the 'catching' of marine salts such as Na^+ and Cl^- (Carlisle *et al.* 1966), and it became evident that dry and occult deposition of S frequently equalled, and often exceeded, wet deposition inputs (see DoE 1990).

Figure 4.3 shows the estimates for N and S deposition to adjacent moorland and forest areas in the uplands of the UK, emphasising how vegetation change can influence the deposition of these elements to natural ecosystems, with attendant consequences for soils and waters. Land

Figure 4.3: Inputs of sulphur and nitrogen to a moorland and forest canopy at High Muffles in the United Kingdom. Units are g m^{-2} a^{-1}.



use and the importance of dry deposition of air pollutants of N and S in the total budgets are now an essential component of critical load calculations (Hornung and Skeffington 1993).

The plant canopy is also instrumental in controlling physical conditions at the soil surface, and can act as an insulating layer, protecting the soil from extremes of temperature, and modifying the moisture status through a combination of evapotranspiration and canopy interception.

Principle 2: Importance of the nitrogen cycle

We now know that the nitrogen (N) cycle is important in controlling acidification processes in natural and agricultural ecosystems, largely through the conversion of NH_4^+ to NO_3^-, a reaction which results in the production of two moles of H^+ ions for every mole of NH_4^+ converted (see Figure 4.4). This process, called nitrification, is one of the key reactions within the N cycle, and has important consequences for soil acidification, water pollution, N loss from terrestrial ecosystems, and for the production of the trace gas, N_2O. An understanding of the responses and functioning of the N cycle has been found necessary to assess the full implications of ecosystem management regimes on acidification.

A simplified N cycle is shown in Figure 4.4, with the movement of N from atmospheric pools of N_2, through organic N forms and back into the atmosphere, forming the framework for the cycle. Of particular importance are the reactions which release N from organic matter (mineralisation), and the inorganic conversions of N within the soil. It is in it's inorganic forms that N plays a vital role in plant nutrition, being incorporated into plant tissue, thus returning to organic form (Figure 4.4). The 'short-cut' across this cycle via atmospheric N_2 controls the amounts of N involved in life on the Earth, with N being fixed from N_2 from the atmosphere by a limited number of bacteria (N fixation) and returned to the atmosphere by a much more diverse group of bacteria through the process of denitrification. From the point of view of the

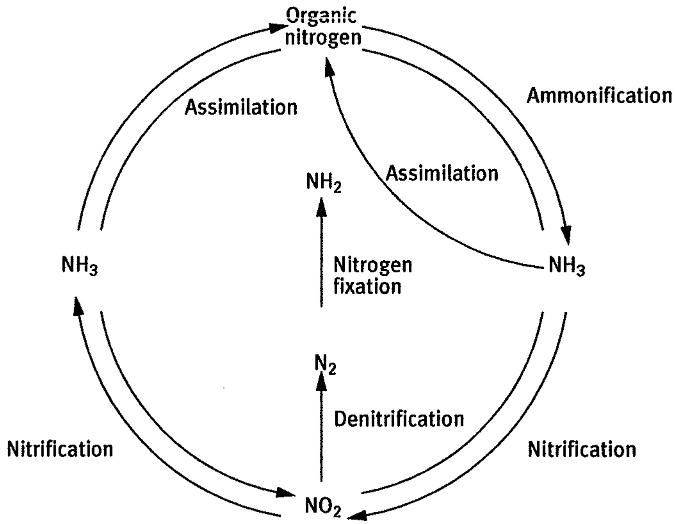


Figure 4.4: A simplified diagram of the nitrogen cycle.

current analysis, it is sufficient to remember that when organic matter decomposes it produces (in the main) ammonium which can then be transformed by nitrifiers to NO_3, with associated production of acidity.

Principle 3: Cations and anions must balance

It may seem only too self-evident that ecosystems are unable to transcend the laws of physics and chemistry, but it took a surprisingly large amount of research to prove that the well-established principle of cation and anion balance applied equally to ecosystems and their compartments and fluxes. The addition of large quantities of a mobile anion, such as SO_4^{2-}, to terrestrial ecosystems, has inevitable consequences for cation/anion balance, in much the same way as global elemental balances must be maintained (see Figure 4.1). The input of H_2SO_4 as 'acid rain' to forests and natural ecosystems, must result in either increased H_2SO_4 transport through the canopy and soil or, more likely, to the retention of H^+, with the loss of cations, such as Ca^{2+} or Mg^{2+}. Whatever the fate of the inputs, the cations and anions must be balanced in all parts of the system, including tree uptake, soil exchange and runoff chemistry.

One demonstration of such a cation/anion balance was provided by the experiments of Wookey and Ineson (1991) in which decomposing leaf litter was placed in a field fumigation system with differing treatment levels of SO_2. A synthesis of the results from this work are presented in Figure 4.5, which shows the concentrations of anions and cations in the leachate beneath leaf litter exposed to increasing concentrations of atmospheric SO_2. The data show that increasing dry deposition of SO_2 at higher SO_2 concentrations resulted in increased leaching of SO_4^{2-} In order for the litter and runoff waters to maintain charge balance, the litter

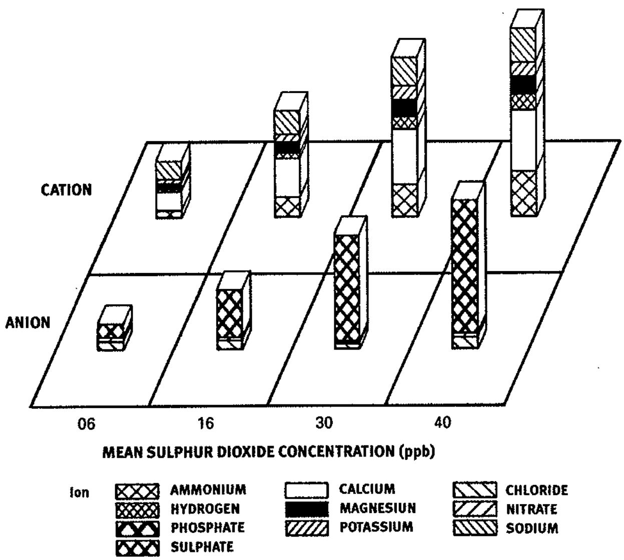


Figure 4.5: Cation and anion leaching beneath the leaf litter of soils exposed to elevated concentrations of SO_2. Note the increases in leachate solutes, and the balance between cations and anions (full details in Wookey and Ineson 1991).

had to lose cations to balance the SO_4^{2-}, with increases in Ca^{2+}, $Mg^{2+}$and H^+ leaching being the response. This led to associated reductions in litter quality and decomposition rates; quite subtle changes in the chemistry of the atmosphere can have major impacts on ecosystem processes, exerted through biogeochemical stress as seen here, but also through toxicity to a few 'keystone' species (see Newsham *et al.* 1992).

Principle 4: The importance of soil buffer zones

Not all soils are alike in terms of their response to acidification and Ulrich (see Ulrich 1986) was instrumental in formalising a classification for soils, with three major groups, according to the soils' response to acidification stress. The first buffer zone contains alkaline soils which have a large amount of cation reserve and remain largely unaffected by acidic inputs. The second group are in the cation exchange buffer range (CEBR; pH 4.2 - 5.0), and respond to acidic inputs by increasing losses of cations from exchange sites in the soil. This is similar to the behaviour shown by the leaf litters, described above. The third group of soils, those in the aluminium buffer range (ABR; pH 3.0 - 4.2), have very little cation exchange capacity, and release aluminium as a result of acidification. The pH_{H2O} values which divide these groups of soils are critical, and the physico-chemical boundaries between these groups are almost 'switch-like' in their behaviour. The importance of these distinctions is critical in determining the impact of acidification on Al^{3+} concentrations in soil solution and stream water and, hence, toxicity to sensitive ecosystem components, such as tree roots (Ulrich *et al.* 1980) and fish populations (CLAG 1995).

Synthesis and worked example

The four principles outlined above provide a framework for understanding and predicting how management practices will impact biogeochemical cycles in ecosystems. They have been deliberately generalised from the body of scientific investigation in order to enable them to be readily assimilated into management thinking. There are major difficulties in 'delivering' the results of ecological science to those involved in the day-to-day management of complex ecosystems. I have found that these four principles are accessible, relevant and understandable, offering an important means of communicating fundamental ecological research to managers.

Given these four working principles, we will now apply them to the ecosystem management example of forest felling. This routine forestry operation has major implications for water quality, including the threat of damage to fish populations. Using the principles, we can systematically consider the biogeochemical impacts:

Principle 1: The importance of the plant canopy. Felling causes major changes in the structure of the canopy of the forest, with mature trees often being replaced immediately by minimal vegetation and, in the short-term, with opportunists or regenerating seedlings. The initial impact that this canopy removal has is to reduce the amount of pollutants 'trapped' by the system, and this is reflected in changes in soil and stream water chemistry. There is a reduction in aerosol deposition, accompanied by a reduction in the dry deposition of N and S. Results of experiments in which the consequences of felling plantation forests have been monitored confirm predicted reductions in stream and river concentrations of Na^{2+}, Cl^- and SO_4^{2-} (see, for example, Reynolds *et al.* 1992; Adamson and Hornung 1990). Clearly, the quality of the atmosphere around the forest is critical in determining this outcome; forests distant from maritime aerosol inputs, or close to industrial areas, will result in very different streamwater quality changes. These differences can, to a large extent, be predicted.

Principle 2: Importance of the N cycle. The second major implication of felling is for the organic matter stored on the forest floor beneath the trees. This material has usually accumulated because of the cooler, drier micro-climate beneath the trees, which limits decomposition.

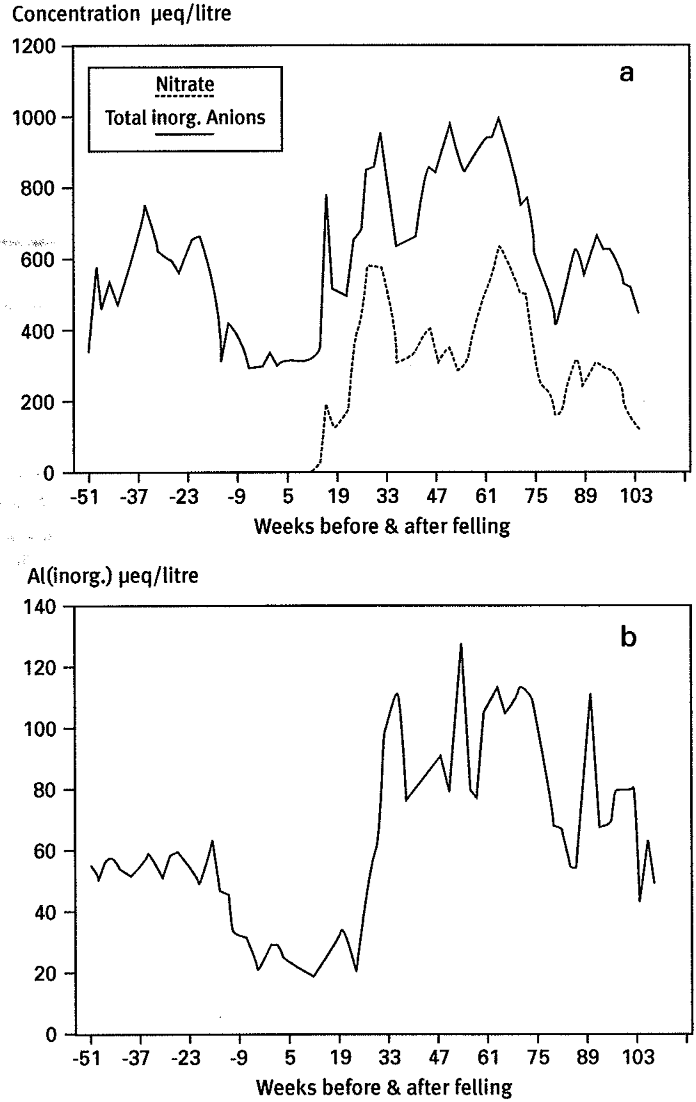


Figure 4.6: The impact of felling a plantation forest on soil water chemistry, showing changes in a) mean nitrate and b) aluminium concentrations. Data are from Reynolds et al. 1992.

Removal of the canopy places this organic matter in new physical surroundings, ones which are more conducive to decomposition, leading to the mineralisation of N from the organic matter, with the potential for nitrification and associated acidification. Hence, in Figure 4.6a, we see that the stream water draining the felled forest has a higher concentration of NO_3. The amount of NO_3 released is very much a result of the N fertility level of the site.

Principle 3: Cations and anions must balance. Applying Principle 3 in this situation leads to the conclusion that the felling operation must cause the loss of cations to accompany the NO_3 being generated, since cations and anions must balance. However, predictions about which cation will dominate this balancing of charge requires a knowledge of the soil buffer zone.

Principle 4: The importance of soil buffer zones. The buffer zone of the soil will dictate which cations are available to balance the NO_3 generated from the decomposition and nitrification of the organic N held in the soil. In the case of the example shown in Figure 4.6, the soil in question has a pH_{H2O} in the aluminium buffer range and we would predict a loss of Al. Figure 4.6b clearly confirms this, and suggests that felling at this site may lead to impoverishment of freshwater invertebrate communities and a decline in fish stocks, largely through aluminium toxicity.

Conclusions

These principles are the product of basic research into the biogeochemistry of ecosystems, stemming largely from concerns over the damaging effects of acid rain to forest ecosystems. Each year I teach these principles to practical forest managers, who readily appreciate these concepts then them into making field decisions about the risks of felling in particular regions. They give consideration to soil pH, the N status of the site, and the local levels of air pollution before operations begin, and make decisions based on this knowledge. However, if simply left with a collection of research papers, these important ecological findings may miss the audience best suited to applying them.

The 'acid rain' debate and the science resulting from it have been essential in alerting the world to some of the problems arising from the combustion of fossil fuels. This is leading to legislation which will incorporate many of the concepts generated from the research, resulting in the setting of critical loads for deposition of acidifying pollutants to ecosystems. As a consequence, sensitive natural populations will be protected from pollutant damage. The additional, secondary, products of this research, are a deeper understanding of how ecosystems function, leading to more sound, holistic management strategies. The principal opponent in the protection of natural ecosystems appears to be ignorance; science has a crucial role in overcoming it.

Acknowledgements

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4.2 Species Distribution and Environmental Change

Considerations from the site to the landscape scale

Brian Huntley

Abstract

Quaternary palaeoecological evidence indicates that species' principal response to environmental change is change in their distribution. The magnitude of forecast future climate changes is comparable to changes since the last glacial maximum. The rates at which potential range limits will shift, however, may be as much as 10x faster than in the Quaternary record.

A key feature of species spatial responses is their individualism. As a consequence, a species assemblage that at one time and place co-exists as a community will at a different time and/or place participate separately in distinct communities differing in composition. Species communities and thus ecosystems have no long term persistence.

As the environment changes ecosystems within one site change in composition as new species enter and species previously present no longer can persist. At a landscape scale, species move between sites and even the structure of ecosystems on the landscape changes. Ecosystem management often aims to conserve rare or threatened species. At a site scale these species often are the first for whom the site becomes unsuitable as the environment changes; they may be near their range margin and hence vulnerable to a small change. Other rare species, however, may be favoured by the environmental change; they too may have been near their range margin but the environmental change expands their potential range. Site management must adapt to accept losses and gains as an inevitable consequence of environmental change; management must be planned in a context of dynamic ecosystem responses to change rather than being designed to limit or prevent dynamic responses.

At the landscape scale too, ecosystem management must take account of the inherent dynamism of species distributions. A landscape parcelled up into units used for different purposes may substantially inhibit such dynamism if the scale of the landscape texture is inappropriate. No one scale, however, can be appropriate for all classes of organism. Ecosystem managers thus must seek to develop landscapes that have 'grain' at a hierarchy of spatial scales. Only in this way can resilience to change be achieved.

Introduction

Most ecologists and biogeographers would agree that species' distributions generally are determined by their environmental tolerances and/or requirements (Good 1931; Woodward 1987). Furthermore, many also would accept that this generalisation is true at a hierarchy of spatial scales, albeit with different aspects of the environment becoming of relatively greater importance at different scales. Given the perspective of Quaternary palaeoecology, it also is apparent that this generalisation applies to the temporally changing patterns of species' distributions (Huntley and Webb 1989). From this it follows that any future environmental changes will lead to changes in species' distributions.

In this paper I first shall review briefly the evidence of species' responses to the changing environmental conditions of the late Quaternary, drawing out the key characteristics of these responses. I then shall consider what are the likely responses of species to forecast future environmental changes and how these may be modelled. Finally I shall discuss the implications of these responses with respect to ecosystem management aimed at the conservation of biodiversity before presenting some general conclusions. At each stage I shall examine the importance of spatial scale, from that of the individual site, which I take to be of the order of 1 ha or less, to that of the landscape, which I take to be of the order of 10^2 - 10^5 km^2.

How species responded to past environmental changes

That there have been global environmental changes of large magnitude during the relatively recent geological past of the late Quaternary now is well known (Wright *et al.* 1993). 20,000 years ago the world was in the grip of the most recent of the Quaternary glacial stages with massive ice sheets developed especially in north-eastern North America and Fennoscandia (Denton and Hughes 1981); global mean temperatures are estimated to have been about 5°C less than today (Folland *et al.* 1990), sea level was lowered ca. 120 m (Fairbanks 1989), the atmospheric concentration of CO_2 was ca. 190 ppmv (Barnola *et al.* 1987) and that of CH_4 was ca. 350 ppbv (Chappellaz *et al.* 1990). Furthermore, these global conditions resulted in circulations both of the atmosphere (Kutzbach and Guetter 1986) and of the oceans (Imbrie *et al.* 1992) that differed from those of today; these differences in turn resulting in regional and seasonal climate differences that sometimes, of course, were much less. Extreme temperature differences were seen in the North Atlantic region where the sea surface temperature in summer was as much as ca. 15°C less than today (CLIMAP Project Members 1984) and in north-western Europe where winter temperatures may been >30°C less than today (Atkinson *et al.* 1987; Guiot *et al.* 1993).

Subsequently, changing seasonal insolation (COHMAP 1988) triggered melting of the bulk of the glacial ice sheets between ca. 15,000 and 10,000 years ago. During the same interval, sea level consequently rose more or less to its present level whilst atmospheric concentrations of CO_2 and CH_4 rose to ca. 280 ppmv and ca. 600 ppbv respectively. Global mean temperature rose to more or less its present value, although this disguises greater rises in some regions whilst other regions remained cooler than today. Atmospheric and ocean circulation patterns were re-arranged during this time (Kutzbach and Guetter 1986; Imbrie *et al.* 1992; 1993; Huntley in press), some aspects of the atmospheric circulation being intensified relative to today as a result of the maximum in northern hemisphere summer insolation between ca. 12,000 and 9,000 years ago. The African and Asian monsoons, in particular, were intensified during this interval, leading to increased precipitation in many arid areas compared to that of today (Street-Perrott and Perrott 1993).

Since deglaciation seasonal insolation regimes have continued to change, being now more or less as they were at the time of the last glacial maximum. This has let to continuing changes in atmospheric and ocean circulation patterns and hence to ongoing regional and seasonal climate changes. Overlaid upon these smaller magnitude changes are other climate fluctuations resulting from such factors as fluctuations in solar output, explosive volcanic eruptions and internal instabilities of the climate system such as the El Niño - Southern Oscillation (ENSO).

The responses of species to these complex environmental changes are well documented by the late-Quaternary fossil record of several major taxonomic groups of terrestrial, freshwater and marine organisms. A principal line of evidence records the response of terrestrial higher plants and is provided by palynology (Huntley 1990), although this is complemented by other sources of palaeovegetation evidence, especially that of plant macrofossils. Palynological data for the period since the last glacial stage are especially numerous in North America and Europe and in each case have been used to draw so-called isopoll maps portraying the past patterns of distribution and abundance of individual pollen taxa (Bernabo and Webb 1977; Webb 1988; Huntley and Birks 1983; Huntley 1988). Sequences of such maps then can be used to deduce the rates and magnitudes of the shifts in overall distribution of the plants that produce the pollen; an example of such a sequence is shown in Figure 4.7. Such studies have shown that many tree taxa of these two regions exhibit range margin shifts of 2000-3000 km between the glacial maximum and their Holocene maximum range limit. Range margins advanced typically at rates of 250-500 m yr^{-1} as a long-term average (Huntley and Webb 1989). However, some exceptions, including for example *Picea*, advance more quickly, at rates of up to 2 km yr^{-1} (Ritchie and MacDonald 1986; Huntley 1988).

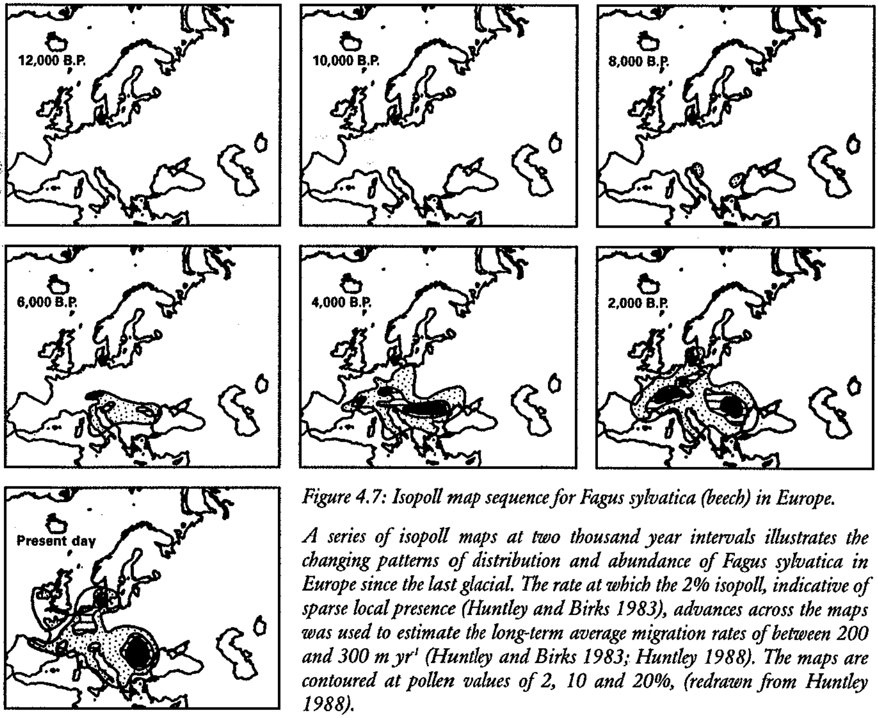

Figure 4.7: Isopoll map sequence for Fagus sylvatica (beech) in Europe.

A series of isopoll maps at two thousand year intervals illustrates the changing patterns of distribution and abundance of Fagus sylvatica in Europe since the last glacial. The rate at which the 2% isopoll, indicative of sparse local presence (Huntley and Birks 1983), advances across the maps was used to estimate the long-term average migration rates of between 200 and 300 m yr^{-1} (Huntley and Birks 1983; Huntley 1988). The maps are contoured at pollen values of 2, 10 and 20%, (redrawn from Huntley 1988).

with past changes, there are likely to be pronounced impacts upon the circulation both of the atmosphere and of the oceans. Changes in atmospheric circulation will result in altered precipitation patterns as well as regional temperature changes. Changes in ocean circulation first are likely to alter phenomena such as the seasonal extent and duration of sea-ice cover; they have the potential, however, to produce major feedbacks upon the atmosphere if, for example, important features of the thermo-haline circulation are altered as many simulations indicate is likely (Houghton *et al.* 1996).

Finally, and perhaps most importantly, the increase in global mean temperature will be faster and larger than any during the Holocene (the last 10,000 years) and is likely to take place faster even than that during the last deglaciation. Current forecasts are of a rise of between 1.0 and 3.5°C between AD 1990 and 2100 (Houghton *et al.* 1996); the best estimate of a 2.0°C rise during that interval amounts to a rate of ca. 0.2°C per decade. The global mean temperature increase during the last deglaciation is difficult to estimate because the majority of palaeoclimate records represent local or regional changes. Although some of these changes were extremely rapid, the evidence does not yet indicate a rise in global mean temperature at a rate of much more than ca. 0.2°C per century. Even if it is in due course demonstrated that global mean temperature rose much faster than this during deglaciation, however, there seems little doubt that global mean temperature has at no time during the Quaternary (the last 2.4 Ma) risen as high as it is forecast to rise during the next one to two centuries.

Although the Quaternary record cannot tell us precisely what will be the responses of individual taxa or of the major ecotones to the forecast changes, it does provide perhaps the only evidence we have as to the underlying principles and mechanisms of these responses. Succinctly, species will tend to shift spatially so as to track the environmental conditions that they require and they will do so individualistically. The magnitude of the potential changes in range can be estimated by modelling the climatically-determined ranges of species (Huntley *et al.* 1995; Huntley 1995), whilst the dynamics of species adjustments to these changes in their potential ranges can be simulated using models of dispersal and migration (Collingham *et al.* 1996). The former type of static models, developed by relating species' present ranges to present climate, can be used to simulate species' potential ranges for alternative climate scenarios. Such simulations show the potential range of boundary shifts for many species to be of the order of hundreds or even a thousand kilometres or more for the equilibrium climate changes associated with a doubling of the radiative effect of greenhouse gases in the atmosphere (Huntley *et al.* 1995; Huntley 1995); an example is illustrated in Figure 4.8. Such a doubling is expected to have occurred by the middle of the twenty-first century and the latest estimates are of global mean warming relative to 1990 or between 1.0 and 3.5°C, the 'best estimate' being 2.0°C, by AD 2100 (Houghton *et al.* 1996). Many species' capacity for dispersal and migration will be insufficient for them to maintain their ranges in equilibrium with such rapid shifts in their potential ranges (Davis 1989). Dynamic modelling studies show that habitat fragmentation further will hinder the migratory response of species, rendering them even less able to keep up with the changes in their potential range (Collingham 1995; Collingham *et al.* 1996); an illustrative example of the results of such a modelling approach is presented in Figure 4.9.

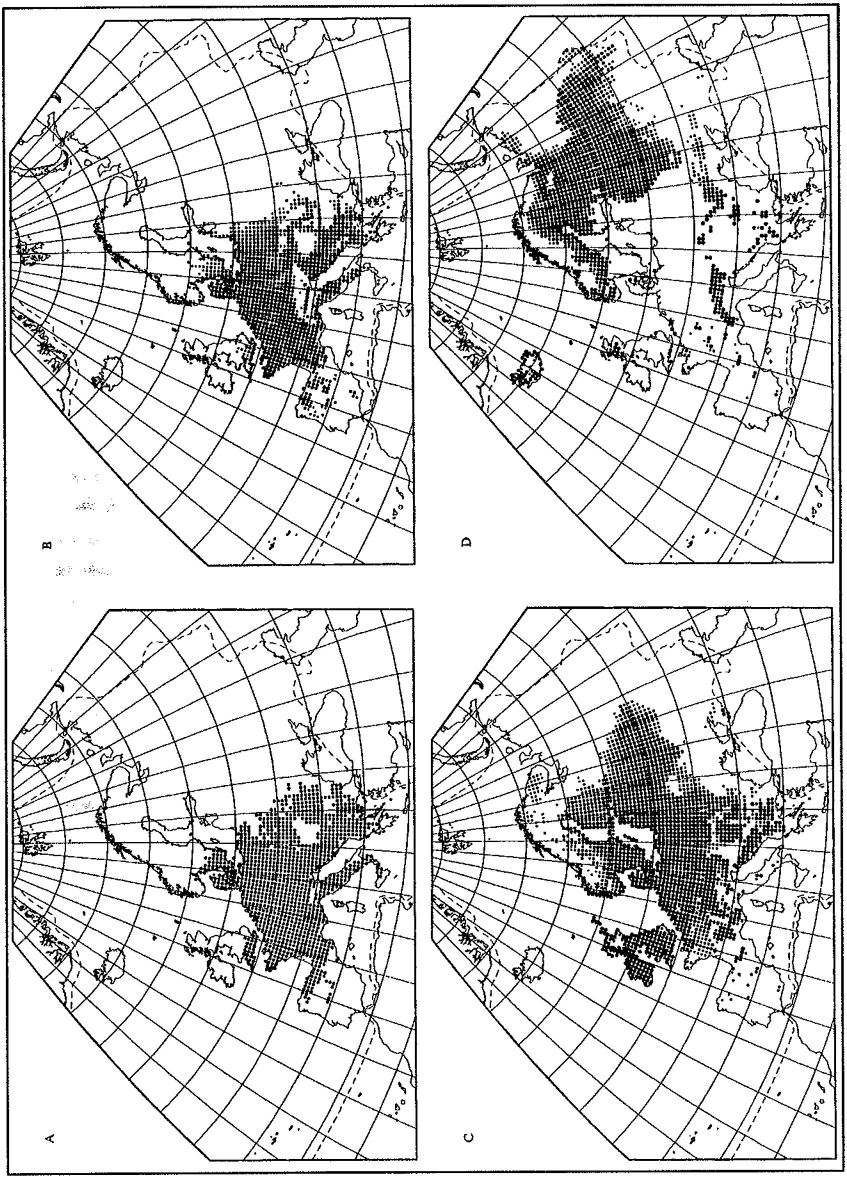

Figure 4.8: Observed, simulated and potential future distributions of Fagus sylvatica (beech). All four distributions are mapped using the ca. 50 km UTM grid adopted by Jalas and Suominen (1972) for their Atlas Florae Europaeae.

(A) Observed distribution (Jalas and Suominen 1976);

(B) Distribution simulated for present climate (see Huntley et al. 1995) for details of the method and Huntley (1995) for an illustration of the climate response surface for Fagus sylvatica);

(C) distribution simulated for 2 x CO^2 scenario by the Oregon State University general circulation model (Schlesinger and Zhao 1989):

(D) distribution simulated for 2 x CO^2 scenario by the UK Meteorological Office general circulation model (Mitchell 1983).

The varying size of dots used on B, C and D indicate the probability of occurrence with larger dots indicating higher probabilities.

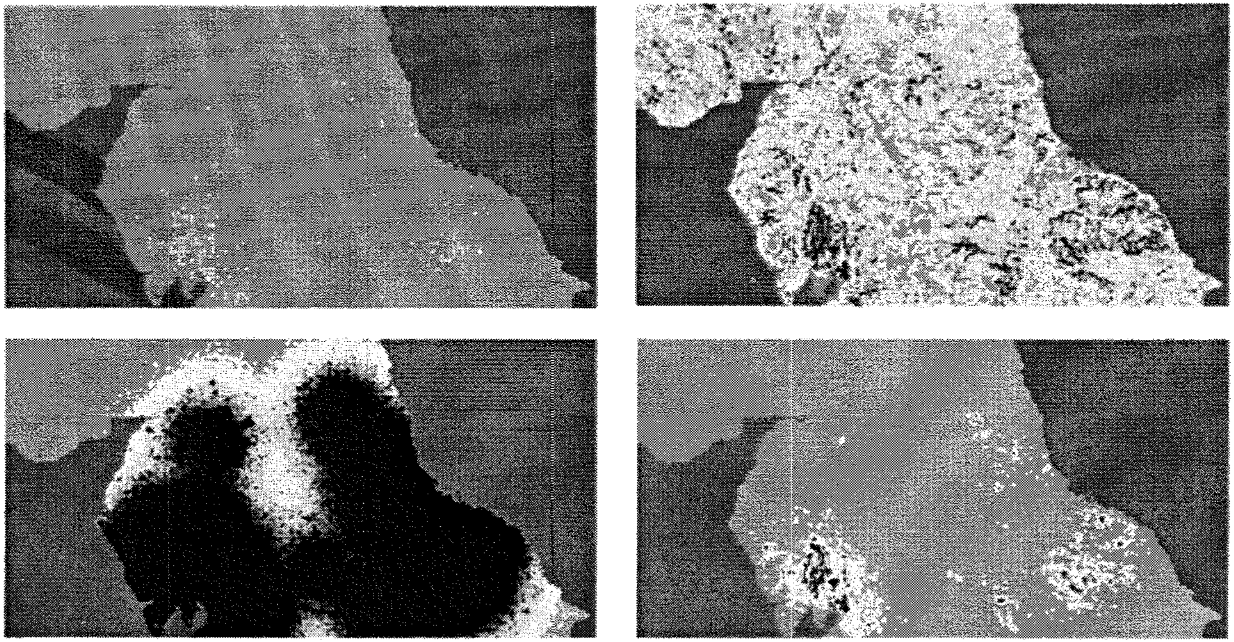

Figure 4.9: Effect of habitat fragmentation upon the migration of Tilia cordata (small-leaved lime) (Redrawn from Collingham 1995).

An illustration of the use of the MIGRATE model (Collingham, et al., 1996) to simulate the dispersal and migration of the tree Tilia cordata across northern England and southern Scotland following a favourable climatic change. The tree currently reaches its climatically-determined northern limit in Britain in this region (Pigott and Huntley 1981; Pigott 1991).

(top left) Initial distribution of Tilia cordata in northern England mapped on a 1 km grid across an overall area of 280 x 140 km (after Pigott (1992) with refinements following Paice 1974);

(top right) Proportion of deciduous woodland mapped on a 1 km grid using the Institute of Terrestrial Ecology Land-cover Data and transformed into a 'carrying capacity' for T. cordata by assuming that this tree might occupy a maximum of 80% of the area of deciduous woodland;

(bottom left) simulated distribution and population density of T. cordata 250 year (10 generations) after an environmental change enabling expansion from the present relict populations and assuming continuously available habitat;

(bottom right) simulated distribution and population density of T. cordata 250 year (10 generations) after an environmental change enabling expansion from the present relict populations with present fragmented habitat availability as determined by the extent of deciduous woodland.

We can predict two general consequences for ecosystems as a result of such rapid climate changes. The extent and intensity of these consequences will vary amongst taxa according to the stages of their life cycle that determine their various climatic limits and, in the case of sessile organisms, according to their longevity and the frequency of disturbance of their habitat. Taxa whose 'warm' or 'dry' limit is determined by the physiological tolerances of the established individual can be expected to be eliminated quickly near to these limits of their range as climate warms and/or dries. Conversely, mature individuals of taxa whose range is limited because of sensitivity during some part of their regenerative cycle may persist for the remainder of their life span under conditions no longer favourable for regeneration so long as they are free from disturbance; as single disturbance or extreme event, however, then may lead to the widespread eradication of such taxa whose range margin no longer is in equilibrium with climate. At their 'cool' climate limit the response of taxa will be constrained by their ability to disperse into and colonise areas rendered newly suitable as a result of climate warming.

The two general consequences are, firstly, that many ecosystems will lose species at a much faster rate than the potential new species can enter. This will lead to an overall reduction in ecosystem diversity and will provide opportunities for opportunist, weedy species to occupy the resulting gaps. As a result there is likely to be a general trend towards more uniform ecosystems dominated by widespread opportunists of a more ruderal strategy (*sensu* Grime 1978). Although, given sufficient time, the potential new ecosystem components with more competitive or stress-tolerant strategies may colonise, and diversity be restored, this is a much longer-term process and may fail as a result of the second consequence of the rapidity of climate change. This second consequence is the likelihood of at least severe population reduction, probable loss of intra-specific genetic diversity and possible extinction of taxa as a result of their potential range under changed climate conditions having little or no overlap with their present range. The species most at risk or extinction as result of this are those with relatively limited climatic and hence geographical ranges; however, as Figure 4.8 illustrates, even a species as widespread as the European beech (*Fagus sylvatica*) has only a ca. 11% overlap between its present range and that simulated for a more extreme warming scenario. Given the extent to which many taxa show environmentally-related genotypic variation, even taxa with modest overlaps between their present and potential future ranges are at risk of extinction if some alleles have limited spatial distribution within the overall range of the species and the species' rate of gene flow is limited.

Viewed at the scale of an individual site, the magnitude and rate of the changes likely to occur will depend upon the location of that site. Although it is likely that most sites will experience some species losses and some gains as a consequence of spatial adjustments to the changing environment, sites close to major ecotones will be much more severely effected in the short term than will sites near the middle of the environmental extent of a biome or ecosystem complex. The former sites may experience a turnover of the majority of their species within a relatively short time period of between a few decades and one to two centuries. Sites more remote from ecotones, however, although initially buffered from the impacts of the changing environment, may in the longer term of a few centuries suffer principally species losses as the environment no longer is suitable for species that initially were present but the migration of potential colonisers lags behind the expansion of their potential ranges. Sites set within landscapes dominated by agriculture, plantation forestry or other human-altered areas will suffer the additional consequences of the impact of habitat fragmentation upon species' rates of migration. Such isolated habitat fragments will at best experience increased lags before species reach them; in many cases their degree of isolation virtually may preclude the dispersal of propagules or individuals of some potential colonists from the nearest source population.

Viewed at the landscape scale the impact of the environmental changes will greatly depend upon the extent to which that landscape exhibits varied relief. Low-relief landscapes, such as we find in most of southern Britain, offer little in the way of topographically-determined micro-habitats that would allow species to make only small spatial adjustments in order to continue to meet their environmental requirements. Thus in these landscapes the predominant mechanism of species' response will be by spatially-extensive range adjustments. As a result, the same general consequences will be seen as for individual sites, depending upon the location of the landscape with respect to any major ecotone.

In landscapes of higher relief the predominant impacts, at least in the short term will be a tendency for species' elevational ranges to shift upwards and a pattern of small-scale spatial adjustments so that species formerly found on sunnier, warmer exposures will be displaced onto more shaded, cooler sites. Those species that already occupied the highest elevations and/or the coolest, most shaded slopes, however, will have nowhere to move to within the landscape and so they are likely to become at least locally extinct. At the same time, more lowland species are likely to spread into the lower elevation areas of the landscape. Once again, the rapidity with which the latter occurs will depend upon the proximity of these potential colonising species as well as upon the degree of habitat fragmentation resulting from human management and alteration of the landscape.

Conserving biodiversity - the implications of species responses to environmental changes

Before we can consider the implications of species' responses to environmental change for the conservation of biodiversity we first must define our usage of 'biodiversity'. This term is bedevilled by three problems of scale; taxonomic, spatial and temporal scale each influence our concept of biodiversity. In terms of taxonomic scale, it sometimes has been suggested that we must clarify whether it is species or genotypes whose diversity we wish to conserve. Most commonly discussions of biodiversity focus upon species. However, in the context of a changing environment our attention more readily becomes focused upon intra-specific genetic diversity and the role that this plays in enabling species to occupy a range of environmental conditions. If species lost significant proportions of their genetic diversity then they may become constrained to a much narrower range of environmental conditions; if this happens then biodiversity effectively has been reduced under at least some combinations of environmental conditions. Given the fundamental response of species to a changing environment, namely shifts in their geographical range, then the rate at which the environment changes in relation to the rate at which spatially-restricted alleles can flow through the species' population becomes of vital importance. If gene flow is insufficient then even a species with a substantial overlap between its present and potential future range may become extinct as the genotypes present throughout the range become inappropriate to the changed environmental conditions. Thus it is essential to the conservation of species diversity in a changing environment that we ensure the conservation of intra-specific genetic diversity. The two should not be seen as alternatives; the one is a prerequisite for the success of the other.

The problem of spatial scale is often one of perspective; it would be possible for the entire biota of a single site to change as a consequence of species' ranges shifting in response to an environmental change, yet the species-level biodiversity of the site might be maintained if it contained the same number of species after the transformation as before. At landscape scale the problem is even more complex; some landscapes may contain more species following adjustment to a given environmental change, whereas others may contain the same or a smaller number as a consequence of the same change. At the global scale, however, at least the short-term consequences for species-level biodiversity only can be neutral or negative; negative consequences will arise from any change that either is too rapid for species to make the necessary spatial adjustments of their ranges, or else that leads to the loss of some combinations of conditions that have prevailed in at least some areas for a period long enough for species to have evolved that are adapted to those combinations. Forecast global warming threatens biodiversity on both counts; it will be too rapid for many species to maintain their ranges in equilibrium with the changing environment, and it will lead to globally warmer conditions than have occurred during the Quaternary so that the environmental conditions occupied by specialised polar and high mountain species are likely to disappear.

The issue of temporal scale is relevant when we consider the potential scale of anthropogenic extinction, both that occurring due to habitat destruction and that which is likely to be a consequence of global warming. It is clear from the geological record that, given enough time, global biodiversity can recover, through evolution, from episodes of mass extinction. Thus it is possible to argue that anthropogenic extinction is unlikely permanently to alter global biodiversity. However, sustaining such an argument requires a very long-term perspective; the recovery of global biodiversity following mass extinctions in the geological record took periods of 10^7 years or more. On the other hand, it is essential also to recognise that there is a background level of extinction that is the result of the continuous process of evolution and the continuing environmental change; thus to expect to conserve all of the species present on Earth at any one point in time is unrealistic. The appropriate temporal scale and view to adopt is one that recognises that evolutionary replacement of biodiversity lost as a result of an unusually rapid extinction rate is a very long-term process; thus we must assess anthropogenic extinctions with respect to the background rate of extinction. Any excess extinction caused by humans in the short term of the twentieth and twenty-first centuries is likely to take millions of years to be redressed through evolution. From a human perspective loss of biodiversity that persists over such a timescale effectively may be considered permanent; *Homo sapiens* has after all existed as a species for only ca. 100,000 years (see e.g. Malassé 1993), whilst 'anatomically modern' human forms, as opposed to Neanderthals, arrived in Europe only during the last glacial stage some 40,000 years or so ago (see e.g. Stuart 1993).

In summary then, conserving biodiversity only is a useful concept at a global scale and to be meaningful must be considered in terms of human timescales; conserving species-level biodiversity will not be possible without also conserving intra-specific genetic diversity. However, it also is apparent that some loss of biodiversity probably is inevitable, because even if stringent controls upon emissions were implemented now there is a commitment to some global warming over the next century as a consequence of the climate system, which is lagging behind the emissions to-date, reaching a new equilibrium. Globally, the most threatened organisms are those limited to high latitudes and/or altitudes, i.e. the cold extremes that will be the first to disappear as the Earth warms. Of the two it may well be that montane species are at greater risk for two reasons; firstly, the polar regions may warm but will continue to experience the same low insolation intensity, permanent winter darkness and summer light that together may exclude many potential invaders, and secondly, montane species often are found in isolated areas of high elevation and thus experience more severe restrictions upon their ability to migrate tan do many high latitude species that may not have significant barriers to their poleward migration.

Putting this into a British context and considering the landscape scale, we can predict that the Cairngorms National Nature Reserve is likely during the next century to lose a number of the higher elevation species for which it today is noted. Plant species such as *Luzula arcuata*, *Saxifraga rivularis* and even *Juncus trifidus* that are restricted to high north-facing or to summit areas are at risk, as are other high-elevation species such as *Plectrophenax nivalis* (Snow Bunting) and perhaps even *Charadrius morinellus* (Dotterel) and *Lagopus mutus* (Ptarmigan). It is unlikely that such losses can be avoided and so conservation managers must take a dynamic view, and a wider view, as they plan the conservation management of reserves and of the landscapes within which they are located. A parochial concern to conserve species in Britain that, if we take a wider and more practical viewpoint, are and are likely to remain much more abundant in the Scandinavian mountains, is a distraction from the more pressing issue of how to manage reserves and the landscape in a way that will enable the greatest number of species overall to survive the inevitable climate changes.

At the site scale that corresponds to many individual nature reserves, at least in the highly anthropogenically-altered areas of northern Europe and large parts of North America, the problem becomes even more focused. Because such sites once again often are identified using rather parochial criteria, they often lie close to the margins of the ranges of the species and/or ecosystems that they are established to conserve. Where they lie close to a margin that potentially will expand as a result of climate change, then they may assume great importance as a source from which this expansion will be fuelled. However, many of them will lie close to a potentially retreating margin; such sites may experience a rapid species turnover and thus might be deemed to lose their 'conservation value' if a parochial view is taken and/or if the legislative framework within which they are established identifies 'conservation value' with the continued presence of particular species. Once again it is essential that a wider view is taken; the value of a nature reserve lies in the opportunities that it provides for species to respond dynamically to the changing environment rather than in the species that it contains at any one moment - to cite a trite but relevant example, the Royal Society for the Protection of Birds reserve at Loch Garten does not lose its 'conservation value' when, as winter approaches, the breeding *Pandion haliaetus* (Osprey) and their offspring embark upon their annual migration to Africa. We need to take the same view with respect to the longer-term migrations of species responding to long-term trends of climate change as we do toward annual migrations of species in response to the annual climate cycle. We can extend this analogy even further; it already is well established that the successful conservation of many annual migrants requires favourable conditions along their migration route, whether that be protection from hunting, as in the case of raptors and passerines migrating through some areas of southern Europe, or the availability of staging posts where migrating shorebirds such as *Calidris canutus* (Knot) can refuel before subsequent legs of their overall migratory flights between the temperate latitudes, where they spend the winter months, and the high Arctic, where they breed. The same is true of the landscape across which species must make their longer-term migrations in response to the changing environment; species must be given equal protection whether in reserves or the wider landscape, and the landscape must be managed in such a way as to provide suitable habitat patches sufficiently close together that they may be sued as 'staging posts' or 'stepping stones' between the reserve areas that likely will remain the principal secure areas within which wild species may maintain the populations needed to fuel their migrations.

Conclusions - towards a strategy for conserving a dynamic world

In a changing world it must be apparent that establishing a network of nature reserves is a necessary but not a sufficient condition of the successful conservation of biodiversity. Species will respond to the changing global environment with individualistic migrations as they strive to 'realise their potential' in terms of the climatic conditions to which they are adapted, these conditions themselves shifting across the landscape inexorably at rates that are likely to outstrip species' ability to keep pace. Successful conservation during the coming centuries of anthropogenically-accelerated global climate change will require that species are afforded equal protection wherever in the landscape they find themselves. Without such overall protection those individuals that are the potential parents of future generations will not be able to make the necessary migrations as the environment changes.

However, extending equal protection to species whether they are within reserves or not will again, whilst necessary, be insufficient to ensure successful biodiversity conservation. In many landscapes the areas at present identified as reserves are relatively isolated and often separated from their neighbouring reserves by considerable expanses of unfavourably managed landscape. Thus it will also be necessary to mange the intervening landscape in such a way as to provide patches of habitat that will facilitate the migration of species. Just what pattern of landscape management is most appropriate depends upon the biological characteristics of the organism upon which we focus. If we consider a seasonally-migrant song-bird or even butterfly, or a resident but wide-ranging large carnivore, then it is likely that patches of habitat isolated by tens or even, in the case of seasonal migrants, hundreds of kilometres often will be colonised as the climate changes. At the other extreme, if we consider a sedentary or sessile organism with limited tendency to disperse and/or with propagules of limited dispersal power, and especially if the organism in question has relatively specialised habitat requirements, then even a habitat discontinuity of a few hundred metres may limit severely the rate at which any long-term migration may be achieved.

So as to facilitate the migration of organisms whatever their powers of dispersal the landscape must be planned in such a way as to provide habitat patches and heterogeneity at a wide range of scales. The overall aim must be to provide a connected network of patches of a diverse range of physical habitat types through which species will dynamically adjust their ranges in response to changes in the environment. For some organisms features such as hedgerows, roadside verges, railway margins and river corridors may provide an essential degree of continuity, whether as protected routes along which more mobile organisms may disperse or as linear habitats along which sessile organisms may spread from one larger area of suitable habitat to another. For other organisms, with greater capacities to disperse across unfavourable areas, habitat effectively may be continuous even if there are gaps of a kilometre or more.

This leads to a model of landscape planning and management that, although continuing to place emphasis upon reserves, sees these as nodes in an overall network. These nodes, and their connecting network, must have 'grain' at a wide range of spatial scales - isolated hedgerow trees may link small copses of ca. 0.1 ha that in turn link woodland patches of ca. 1 ha, whereas the woodland reserves forming the nodes in this network may be 10 ha or more but be separated by distances of ca. 10 km or more. A similar pattern of patches of other habitat types, for example grassland, heathland, freshwater and waterside areas, etc., also should be embedded in the same landscape. Evolving toward such a model of landscape planning and management will, in many areas that are strongly influenced by human activities, require the designation of land that currently may be used for agriculture for the establishment of the necessary patches of habitat. Often these habitats may initially have to be regenerated or 'created'. The current trend toward removing land from intensive agriculture in areas such as the European Union provides an ideal, but generally as yet unrealised, opportunity to take the steps needed to develop the more resilient landscape compatible with the conservation of wildlife in a dynamic environment.

Ecosystem management within reserve areas also needs to be reconsidered; 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 species assemblages. Many conservation bodies' management plans for their nature reserves have as their foundation an identification of certain organisms or communities that are regarded as of particular value; the plan then identifies a strategy that aims to ensure the continued presence of these valued facets of the reserve. In future the emphasis should shift to the types of physical habitat the reserve is able to offer (Hunter *et al.* 1988), and to sustaining these whilst at the same time managing the reserve in a manner that will facilitate the establishment of new species as well as sustaining the output of offspring and propagules by those species already present.

The alternative, of transplantation of species so as to artificially enable them to 'migrate' to newly suitable habitat areas, must be seen only as a last resort. Principally this is because of our ignorance of organisms and ecosystems. We could not effectively transplant the component species of the invertebrate fauna, or of the soil fauna and flora of most areas simply because we don't know what is there. We also know so relatively little about the environmental determinants of the distribution of most species that we either must adopt a 'carpet bombing' approach, in which we introduce species throughout vast areas, most of which will be unsuitable, or else undertake extensive 'gardening' in an effort to ensure the success of more limited introductions that often ultimately will fail. One of the few groups of organisms for which transplantation may prove to be the only option are those limited to isolated mountain tops or to other habitat islands between which even the most favourable landscape management cannot provide the necessary 'stepping stones'.

Finally, it is worth noting that although this discussion has tended to focus upon Britain and Europe for its examples, the principles are applicable world-wide. All organisms will tend to shift their geographical ranges as the environment changes. Human activity, whether it takes the form of hunting, poaching or other forms of exploitation of wild species outside designated reserves, or of the destruction of habitats, either to provide agricultural land or as a consequence of an unsustainable exploitation of their natural products, also is a world-wide phenomenon. Unless we can learn to manage the world around us in a manner that enables the survival of wild species and is appropriate to their needs for dynamic range adjustments, as well as modifying our actions so as to limit the impact that they have upon the global environment, then the prospects for the successful conservation of global biodiversity over the next one or two centuries are grim indeed.

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4.3 Wetland Ecosystem Functioning: An expert system-type approach to support decision making

Edward Maltby

Abstract

It has been impossible to conserve and manage wisely the wetland ecosystems of the world exclusively by means of a traditional approach to natural conservation. It is recognised increasingly that the highly sensitive rapidly diminishing wetland resources must be managed on the basis of their wider significance for human as well as wildlife welfare and on the basis of their important role in maintenance or enhancement of environmental quality.

Research has focused increasingly on the investigation of wetland functioning and it is demonstrated that these include a wide range of functions which are significant to many sectors of society. A fundamental problem of application of this process is that wetland ecosystems are extremely diverse, not all perform the same functions and functions are not performed to the same extent. In the light of this diversity and variation, it is necessary to develop procedures by which it is possible a) to predict the likelihood of a particular wetland performing a particular function, b) to assess the magnitude to which that particular function is being performed and c) to evaluate the extent to which a function may be affected by a range of human or environmental impacts. The scientific community is increasingly aware of the important processes controlling functions within wetlands and there have been significant attempts also, to place economic values on the performance on these functions. However, it is quite insufficient simply to have this information within the realm of scientific literature.

The most important need is to translate the science base into procedures which can be used by decision-makers and non-experts in order to improve the strategy for wetland resource management. This approach has been adopted at a general level in the United States, but until recently has been absent from other parts of the world and overall has lacked the strength of empirical scientific verification. Recent work funded largely by the European Commission, and in part by the UK Environment Agency and NERC has focused on the development of such procedures for river marginal wetland ecosystems in Europe.

Introduction

There is growing scientific evidence for the importance of wetland ecosystems in the maintenance of environmental quality together with the direct health and welfare of both humans as well as wildlife. However, neither science nor policies dependent on the traditional approach to nature conservation based on site protection and criteria such as biodiversity, naturalness, uniqueness, rarity or typicality has been sufficient to maintain the quantity or quality of the world's wetland ecosystems (Maltby *et al.* 1994). This implies *inter alia* that (i) the scientific research is deficient in range and depth, emphasis and relevance or applicability; (ii) policies are too weak in competition with other societal priorities; or (iii) the mechanisms for implementing either scientific advances or policy objective are lacking.

Loss or degradation of wetland ecosystems often hinges on the inherent conflicts perceived between ecology or nature conservation and the socio-economic needs of society. It is essential to break down this dualism of vision if the fullest possible benefits of wetland ecosystems are to be achieved by society. Both deliberate and intended actions such as drainage, mining and construction as well as the effects of non-intended actions such as agricultural and industrial pollution, river regulation and groundwater obstruction have had a major effect on the extent and character of the European, as well as global wetland resource. Only a relatively small proportion (probably no more than half) of the medieval extent of European wetlands probably remains and very little if any of the remainder can be considered pristine. The pressures leading to wetland alteration and loss remain.

New applications for permission to abstract more surface and groundwater continue throughout Europe and falling groundwater levels compounded by recent drought conditions have led to desiccation of river marginal wetlands in southern and eastern England. The Tablas de Damiel National Park also designated under the Ramsar Convention as a wetland of international importance is frequently a virtual desert as a result of groundwater lowering (Llamas 1988). Plans to divert the River Acheloos in Greece will inevitably impact the coastal wetland complex of Messolonghi. Proposals still abound to erect dams and regulate river flow throughout Europe including major rivers such as the Danube and the Allier. The new dam affecting the Szeged-Fherto area in the Hungarian-Slovakian border region has caused immense controversy over impacts well beyond nature conservation to issues of the recharge, movement and quality of groundwater.

The traditional measures of nature conservation importance will always succeed in Europe in the identification and 'protection' of the high profile wetlands 'jewels'. However, even the strongest site protection may be frustrated because water supply and quality may be impacted far upstream or simply just outside the wetland boundary. These effects are impossible to control without strong management of the water resources and wider landscape processes in the entire river basin. The European wetland resource has been and continues to be at the mercy of decision-making which takes no account of their wider functional importance. Neither politicians nor society as a whole have recognised the role of wetlands in:

1. the provision of goods and services for direct or indirect human use (as well as wildlife)
2. the maintenance of environmental quality

An ecosystem-based approach to wetland conservation and management recognises the importance of the flows of materials and the dynamics of individual species, communities and populations in determining their functioning and character. Effective and sustainable management of natural resources needs to recognise the complex linkages which occur through food webs, nutrient cycles and water movement. These linkages occur at different scales, but can be investigated effectively by means of ecosystem models.

Wetland Functioning

Hydrological, biological, chemical and physical processes occurring naturally in wetlands result in ecosystem functions such as groundwater discharge or recharge, flood control, nutrient transformation, productivity and habitat development or maintenance. Process interactions maintain ecosystem elements or components such as the water regime, character of soil, interstitial water, plant and animal populations, nutrient pools and soil or sediment properties. The ecosystem may generate products such as forest, wildlife, fisheries and grazing resources used by both human as well as wildlife populations. The ecosystem itself possesses attributes which include biological diversity and cultural uniqueness or heritage. All three of these aspects of ecosystem dynamics may result in benefits to human society whether directly or indirectly (Figure 4.10). However, wetlands vary in process dynamics and this gives rise to significant variation in these functional characteristics. This will result in differences in the nature of human values which can be ascribed to any particular wetland (e.g. for water quality maintenance). The science base is still very inadequate in explaining how different wetland ecosystems work and how different environmental factors and processes interact to control functioning (Maltby 1991). Meanwhile economists are only just beginning to wrestle with the possibilities and implications of assigning values to wetland functions, products and attributes (Barbier *et al.* 1997; Costanza 1984; Turner 1991, Crowards 1995).

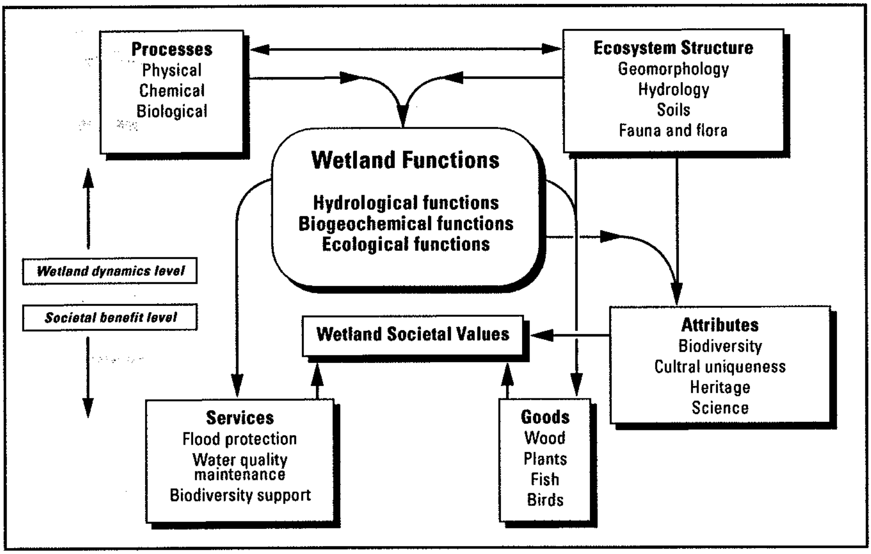

Figure 4.10: Interactions between ecosystem functioning, processes and ecosystem structures and societal values, services, goods and attributes (Maltby et al. 1996).

More effective conservation and sound management of wetlands hinges on predicting how particular wetlands actually work and the recognition of the way in which functions, products, and attributes may be modified by human activities which may adversely affect these ecosystem features and the societal values they maintain. This rationale underpins the development of robust techniques for the functional analysis of wetlands and more effective arguments for their conservation and management.

Rationale of a functional approach to wetland ecosystem conservation and management

A number of arguments support the notion that a functional approach is pivotal to developing systems for the resolution of decision-making and planning dilemmas:

1. It should allow better (e.g. more efficient/appropriate) use of increasingly scarce resources by determining such relationships as the compatibility and intensity of land use activities with functioning, the capacity of particular wetlands to tolerate impacts such as flood or fertiliser loads and the resilience of the wetland to human disturbance. This is consistent with the need to define 'wise use' as necessary for full implementation of the Ramsar Convention.

2. By being ecosystem rather than habitat-led the approach recognises a wide range of both ecological and environmental interactions and is not restricted to a narrow view of conservation.

3. The approach is amenable to economic analysis. This enables translation of the effects of ecosystem dynamics into economic terms and for the consequences of impacts to be more properly accounted. These terms may be more appropriate for the public and politicians to understand than more ethical/scientific arguments relating for example to biodiversity.

4. The implications and benefits of a functional approach extend to better use of water and land resources and improving environmental quality together with human health and welfare. These aspects are highly appealing to the political agenda and thus may be more likely to gain financial support.

5. Functional analysis is attractive to policy innovation. It is already identified explicitly in the EC document *Wise Use and Conservation of Wetlands* (COM(95)0189). Adoption as a key planning instrument at the EU level will be an important driving force in promoting use of the approach at national levels.

6. If the analysis is founded on sound science the assessment of wetland functioning must inevitably lead to more effective protection of the environment. This stems from the ability to target more precisely the systems responsible for particular benefits. There are at least two important dimensions to this:

- optimising the use of limited financial resources i.e. obtaining the greatest benefit for the least spend.
- identifying priority areas for protection, rehabilitation or restoration which stems for example from having some real measures/forecasts of likely success.

Development of procedures for the functional analysis of European wetland ecosystems (FAEWE)

Procedures of functional analysis ideally should fulfil the following:

1. provide a synthetic (and where necessary rapid assessment) tool to assist planners to resolve decision-making dilemmas in regard to land use allocation, permitting of activities and catchment zoning;

2. in doing so provide some guidance on the optimum (or perhaps minimum) conditions necessary for the support of wetland functions;

3. identify levels of impact which alter functioning;

4. provide indications of ecosystem stability, levels of tolerance and resistance to change.

It is evident, however, that the use and implementation of procedures may require new or more comprehensive interpretation of the statutory duties of implementing authorities, new or modifications to existing legislation as well as comprehensive training. The extent to which these additional requirements need to be met depends on the specific user, e.g. non-governmental organisation (NGO) or science/academic community versus statutory or regulatory body. In the case of the United States, for example, procedures have been developed in clear response to a national wetland policy and legislation focused on Section 404 of the Clean Water Act. A choice exists of tailoring a product around the specific needs of a particular user with very straightforward objectives and need for discrete and usable products or of formulating an overall structure which might satisfy a broader user constituency and within which individual protocols can be selected. Inevitably it will be necessary to satisfy initially only a restricted number of such protocols.

A major international interdisciplinary project *Functional Analysis of European Wetland Ecosystems* (FAEWE) funded by the EC DGXII aims to develop procedures for the assessment of wetlands initially in river marginal environments (Maltby *et al.* 1996). The overall internal structure of the FAEWE procedures is represented diagramatically in Figure 4.11. An 'expert' system type approach has been adopted which draws on both empirical research and specialist interpretation to provide the knowledge base which underpins the analysis. The procedures are designed to assess either one particular function or a range of functions and to forecast impacts on single or combinations of functions. Three levels of assessment have been developed: qualitative, quantitative and detailed monitoring/modelling. Once the user has identified the purpose and hence the pathway through the procedures, information needs to be gathered and synthesised in an appropriate format to complete an assessment.

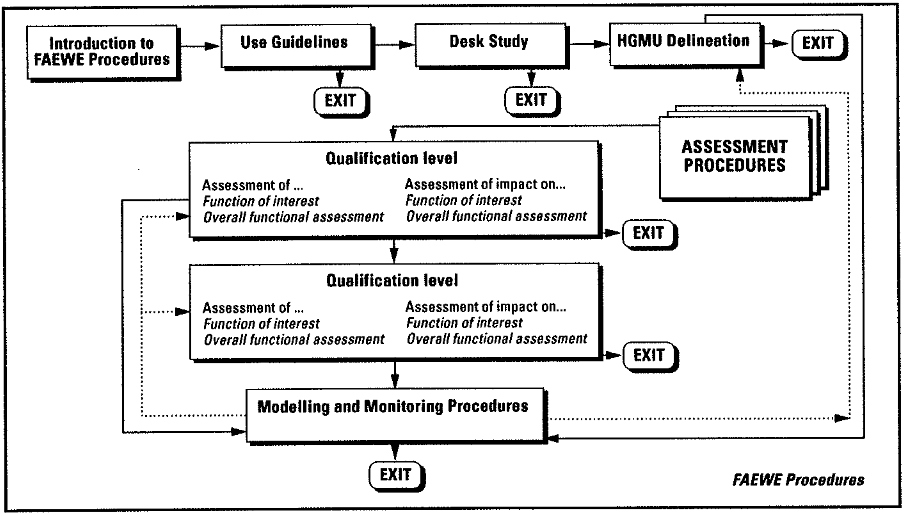

Figure 4.11: Diagrammatic representation of the internal pathways through the functional assessment procedures.

Database establishment provides the user with the methods and protocols to assemble the required information. This section of the procedures is sub-divided into desk studies and field studies. The desk study directs the user to existing data sources that are essential/useful to the study, for instance topographic maps, soil maps, hydrological data and aerial photographs. Details are given on the precise nature of data requirements, and appendices give details of the sources of data sets in each of the EU member states. The data required in the desk study is collated into a preliminary landscape assessment, allowing the user to develop a basic understanding of the study prior to the field based assessments. 'Red flags', indicative of protected sites or areas which are designated already as having a high conservation or environmental value, are identified. In some cases their identification will terminate the functional assessment. Gaps in information identified in the desk study may preclude the successful completion of the assessment and it is possible that for some users the assessment procedure will terminate at this stage.

The basis of the field assessment is the identification and delineation of hydrogeomorphic units (HGMUs). These are areas of homogenous geomorphology, hydrology and/or hydrogeology and under normal conditions homogenous soil. The key principle behind the HGMU delineation methodology is that is allows non-expert users to divide a wetland site into more or less homogenous units. The products resulting from the methodology are a completed field map and annotated recording sheets on which each HGMU is assigned a specific identifying code. HGMUs are considered basic landscape building blocks representing the smallest identifiable area which can be defined in functional terms.

On completion of the site HGMU delineation, the user can progress to undertake the functional assessment procedures. The qualitative assessment will provide a simple outcome statement from one of three alternatives:

- the function is definitely being performed;
- the function is being performed, but only to a small degree;
- it is very unlikely the function is being performed.

For instance, the user wants to assess the flood control function (for short and long term storage) and is interested only in a simple qualification statement of the ability of the wetland to perform the function. Three hypothetical outcomes of the assessment are:

- the function is definitely being performed - the wetland stores water both short and long duration, thus reducing flood peak discharges and helping to attenuate and desynchronise flooding from rainfall downstream;

- the function is being performed, but only to a small degree - the wetland stores flood waters but the overall effect on the reduction of flood peak discharges and downstream attenuation is negligible;

- it is very unlikely that the function is being performed - the wetland does not store flood waters.

An assessment is made for all the processes that maintain the function of interest, based on an evaluation of each of the 'controlling variables' identified for each process. The actual assessment takes the form of a 'decision tree' that allows the user to take a variable route through its branches depending on the answers provided to a series of assessment questions. The final assessment of the 'function' is based on an evaluation of the combined 'process' assessments.

Current research is developing the protocols and tools necessary to translate the value of wetland functions into economic terms and also to extend the approach from the site-specific to the wider catchment scale. The emphasis now is to develop the procedures into practical tools for better decision-making. This phase is being carried out under a new project Procedural *Operationalisation of Techniques for the Functional Analysis of European Wetland Ecosystems* (PROTOWET). Decision-support systems such as these are an essential requirement not only to meet the conservation and management challenges presented by wetlands, but also for other ecosystem types.

Applying a functional approach to ecosystem management and rural development in the Mekong Delta

Melaleuca forests are wetlands which have long supported human communities in the Mekong Delta (Duc 1991). The direct uses of products from this ecosystem are summarised in Table 4.2 and are discussed in more detail in Maltby *et al.* 1996 and Safford and Maltby 1997. Large tracts of *Melaleuca* forest were destroyed during the Vietnam-American war by chemical defoliants, napalm and bombing. Other areas were drained by canals for agriculture, mainly rice cultivation. Much of the drained land has been abandoned subsequently because of the effects of soil acidification resulting from oxidation of potential acid sulphate soil materials (Maltby *et al.* 1996).

New plantations have been established to replace the lost forests, but uncontrolled fires and over exploitation have substantially reduced the effectiveness of the restoration effect. In the province of An Giang the area of *Melaleuca* forest is reported to have declined from 40,000 ha to less than 4,000 ha in the last decade alone. The loss of *Melaleuca* results in economic. environmental and ecological changes:

- loss of economically important resources;
- increased pressure in local climate upland forests and mangroves;
- alternation in local climate especially higher temperatures;
- reduced water quality;
- increase in water borne diseases;
- loss of important wildlife habitat.

Melaleuca species are flood tolerant which means that potential acid sulphate soil materials can be maintained in a reduced condition and material already strongly acidified by oxidation can be returned to a state in which acidification is arrested. The forest is habitat for a wide range of vertebrate and invertebrate species including fin-fish and freshwater prawns. Duc (1991) has identified 23 mammals, 91 birds, 38 reptiles and 11 frog species from *Melaleuca* forest. Endangered bird species include the Eastern Sarus Crane, Painted Stork, Less Adjutant Stork,

Table 4.2 Direct uses and values of Melaleuca ecosytems. MD = Mekong Delta (Safford and Maltby 1997)

Use or Value	**Product or Attribute**	**Special properties**	**Importance during 1990's**
Fuel • firewood • charcoal	Melaleuca wood	High-quality charcoal	Up to 78% of energy consumption in the MD met by firewood and charcoal; much comes from Melaleuca.
Construction	Melaleuca wood	• ideal for posts • durable under water • resists attack by termites	Much used locally and in cities
Fishing	Fish populations	Melaleuca areas provide spawning and nursery grounds for fish caught elsewhere	32% of MD inland capture fishery (14,500 t) caught in Melaleuca areas; importance of Melaleuca probably much higher than this
Essential oil • medicine • insect repellent • soap	Melaleuca leaves and stems	• Oils isolated by steam distillation • Antibacterial and other medicinal properties well documented	Estimated value $40/ha of Melaleuca. Important industry in Long An province, MD, but not elsewhere
Honey, wax and brood production	Native bee species	• High yields: Melaleuca flowers profusely, and native bee species are abundant • Melaleuca honey highly prized in MD	• Traditional, non-destructive bee-keeping methods practiced in Melaleuca in MD • Less sophisticated methods widely operated in Melaleuca forests elsewhere in the region
Food	Harvestable species of fauna and flora	• Some species (besides fish) in Melaleuca ecosystems are important protein sources • Sustainable harvests generally not known.	Intensively harvested where available. Examples: snakes, turtles, crustacea, rodents and wild rice.
Medicine	Plant species besides Melaleuca	Associated species provide range of traditional medicines	Few data available, but production increasing
Livestock fodder	Melaleuca and other plants	• Melaleuca shoots suitable for goats • Marsh sedges fodder for buffalo	
Other uses	Melaleuca bark	Fuel, insulation, caulking, torches, poultices	No data available

Asian Openbill Stork, Milky Stork and Black-necked Stork. Much of the real biodiversity of these ecosystems has yet to be documented, but Table 4.3 summarises its significance.

Table 4.3. Values arising from Melaleuca ecosystem biodiversity. MD=Mekong Delta.

Value	Reason for value	Current importance
Gene poll	Source of genotypes of Melaleuca for rehabilitation programmes	Source of genotypes of Melaleuca for rehabilitation programmes
Scientific research and Education	• Formation of the habitat not understood • Biota and functioning poorly known	Important research sites in Vietnam; elsewhere largely ignored
Wildlife conservation	• One of richest wildlife habitats in MD • Can be rehabilitated; rare species may recolonise	• Biota poorly known (e.g. no data on invertebrates) • Considered unimportant over much of region • many rare species declined in MD through habitat destruction and hunting
Pollination and Pest control	Insectivorous or nectarivorous birds and mammals control pests and pollinate flowers outside Melaleuca areas.	Importance not assessed
Tourism	Old-growth Melaleuca and associated wetlands are spectacular and often bird-rich.	Very low, but tourism in Vietnam and ecotourism in general are increasing rapidly

Much of this biodiversity already has been reduced by the loss of extent and quality (e.g. structural alterations) of the *Melaleuca* habitat. Whilst re-establishment or maintenance of *Melaleuca* is critical for biodiversity, the compelling case for restoration is based on socio-economic grounds. Economic analysis of the value of wood products alone shows far higher returns than that which can be achieved from rice. The yields of honey, fish, essential oils and other non-timber products further widens the differential. If the value of water quality, flood control and water supply together with the direct and indirect benefits of biodiversity are taken into account then the potentially vital role of restored *Melaleuca* forest ecosystems in rural development becomes indisputable.

Lack of success to date in maintaining the area of restored *Melaleuca* forest simply serves to underline the magnitude of the basic social and institutional structure/organisational problems in implementation of sustainable land use activities. The sheer poverty of local populations coupled with the problems of landlessness means that (a) it is often impossible to wait the 5-10 years before realising the greatest benefits from re-established *Melaleuca* and (b) illegal, uncontrolled exploitation destroys the forest.

The Darwin *Melaleuca* wetlands project is linking scientists in the UK with those at the University of Cantho, Vietnam to build an integrated functional approach to ecosystem management which will support rural development (Safford *et al.* 1997). Its logic rests on the multiple benefit to be obtained from the functioning of intact, but accessible and usable *Melaleuca* forest. Implementation depends on extension work with both individual farmers and agencies to establish small *Melaleuca* plots on land holdings or to create mechanisms where the landless poor can benefit from the products of larger areas of forest currently 'protected' and designated as legally inaccessible. It is clear that unless the basic source problem of poverty is at least partly alleviated and more institutional mechanisms established the real potential and value of ecosystem functioning will be impossible to realise. The inevitable consequence of this will be continued land and water degradation and loss of biodiversity as well as reduced human welfare.

Some conclusions

There is increasing recognition of the importance of an ecosystem based approach to conservation. One of the significant advantages of this change from a more traditional species or habitat orientation is to demonstrate the wider benefits resulting from ecosystem functioning. Such environmental benefits as water quality, flood control or fisheries maintenance relate much more readily to human values and are amenable to economic analysis. This is not to relegate the fundamental value of biodiversity and its general expression as species or wildlife. However, the functional approach can be more readily understood by the public and used by politicians in arriving at more informed conservation decisions. In order to translate this new approach into better decisions it is essential to develop appropriate tools. The procedures for functional analysis of European Wetland Ecosystems are an example of the type of tool required. They link fundamental science with decision-making and provide a model for an approach which can be adopted for other ecosystems.

Such procedures may eventually gain acceptance in the formal planning process of developed countries, but in the case of developing countries such application is still impossible to contemplate before more fundamental social and institutional problems are overcome. In such cases the management of ecosystems must adapt to the special circumstances of poverty and landlessness. Science alone cannot provide solutions to the conservation of important natural resources. However, in co-operation with other efforts to relieve ecosystems of overriding human pressures science can help to provide managers with a rationale for maintaining or enhancing the functioning of ecosystems for both people as well as wildlife.

Acknowledgements

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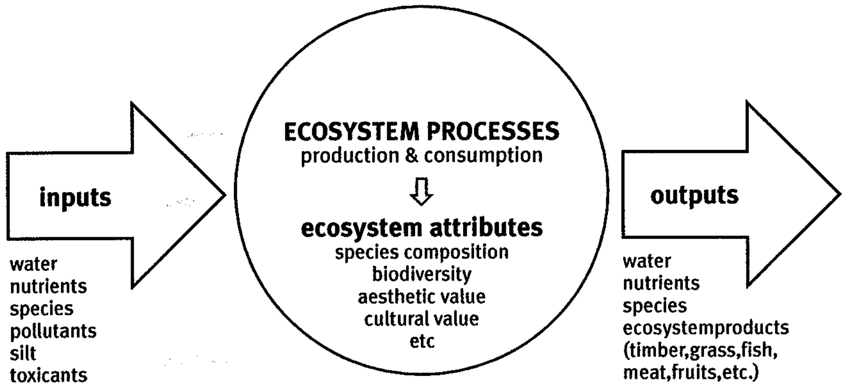
4.4 Ecosystem management - a view from the South

Hillary M Masundire

Introduction

All ecosystems are basically comprised of both biotic and physical attributes. All ecosystems have inputs, perform processes within themselves and produce outputs. This scenario describes the structure and functioning of ecosystems (Figure 4.12).

Figure 4.12: A simple ecosystem model.



Managing ecosystems implies deliberate suppression or amplification of aspects of ecosystem structure or function for a specific purpose. Such purpose is defined by humans, i.e. ecosystems are managed to satisfy specific human needs or desires. Management could involve controlling levels of input of materials into an ecosystem, inputs such as nutrients, pollutants, gases, water or silt because such inputs would in some way alter the structure and/or proper functioning of a given ecosystem. Ecosystem management may also involve deliberate manipulation of processes that take place within ecosystems so as to either promote or suppress some of the processes leading to a manipulation or regulation of quality and/or quantity of outputs from ecosystems. Ecosystem management may sometimes refer to deliberate manipulation of ecosystem products i.e. selective harvesting of ecosystem products. All these aspects of ecosystem management are universal in principle.

Differences between north and south arise from a number of sources. Some arise from differences in both the physical and biological components of ecosystems. The forcing functions which may control, regulate or alter the rates of ecosystem functional processes may differ in kind or in extent or both. Differences with respect to ecosystem management arise from differences in the objectives of management or differences in perceptions of ecosystem values and attributes.

For example wetland ecosystems have received a lot of attention in recent years especially after the Ramsar Conference of 1971. There is a large volume of literature on wetland conservation (Naiman and Decamps 1990; Dugan 1990; Maltby 1991; Matiza and Chabwela 1992; Masundire *et al.* 1995). In the south, southern Africa in particular, wetlands are sources of water, timber, thatch grass, fish, meat (wildlife), recreation, etc. They must be conserved so that they can continue to provide these benefits to the communities that depend on them. The northern view on wetland conservation has been very much influenced by wetlands as habitats for waterfowl. It is not clear to me whether habitats for waterfowl should be conserved because of the functional importance of waterfowl, their aesthetic value or purely for the purpose of species preservation. The specific objectives of ecosystem management, therefore, vary between the north and the south.

This paper presents an argument that the north wishes to manage or influence the management of ecosystems of the south for reasons alien to the peoples of the south. Reasons that may be seen as trivial to the peoples of the south. The paper focuses on sub-Saharan Africa in general with more specific examples from southern Africa.

Historical background

Ecosystem management may be equated with management. It has been argued that it is unsustainable to exploit resource without resource management (Murphree 1991). Resource management is difficult in situations where resource ownership is either uncertain or totally absent. Under such circumstances, resources are 'common property' the management of which has been referred to as the tragedy of the commons (Hardin 1968). This is a scenario where, because resource users do not own the resources, each user aims to obtain maximum benefit at any point in time as there is no guarantee that the resource will still be available in future. The problems experienced in fisheries management, where every fisherman is out to get maximum catches at any point in time as someone else may catch what one has left behind for future harvest is a case in point. In southern Africa, it is apparent that resources whose ownership is not clearly defined are more prone to over-exploitation and/or to degrade more rapidly than individually owned resources.

It is pertinent to look at the modern history of Africa if one is to understand the present demise of resource/ecosystem management in Africa south of Sahara in general and in southern Africa in particular (Figure 4.13).

Four hundred years ago, Africa was no more than a source of slave labour for Europeans to a greater extent and to Asians to a lesser extent. As the European slave traders moved more inland in search of slaves, they realised that there were other valuable tradable resources such as gold and ivory. Thus by 1700 trade had diversified to include gold, ivory and other valuable resources. The abolition of slavery in 1833 and the discovery of mineral and other natural resources led to a desire for more permanent presence by the Europeans. By 1800, European explorers and missionaries penetrated deep into Africa's interior. They brought reports of 'untold riches' waiting to be tapped. By 1900, European states were engaged in a "scramble for Africa". Africa could then be equated to an openly accessible common property resource with no particular owner. The disorderly scramble gave way to a more structured 'partitioning of Africa'. Partitioning was a way of averting the tragedy of commons.

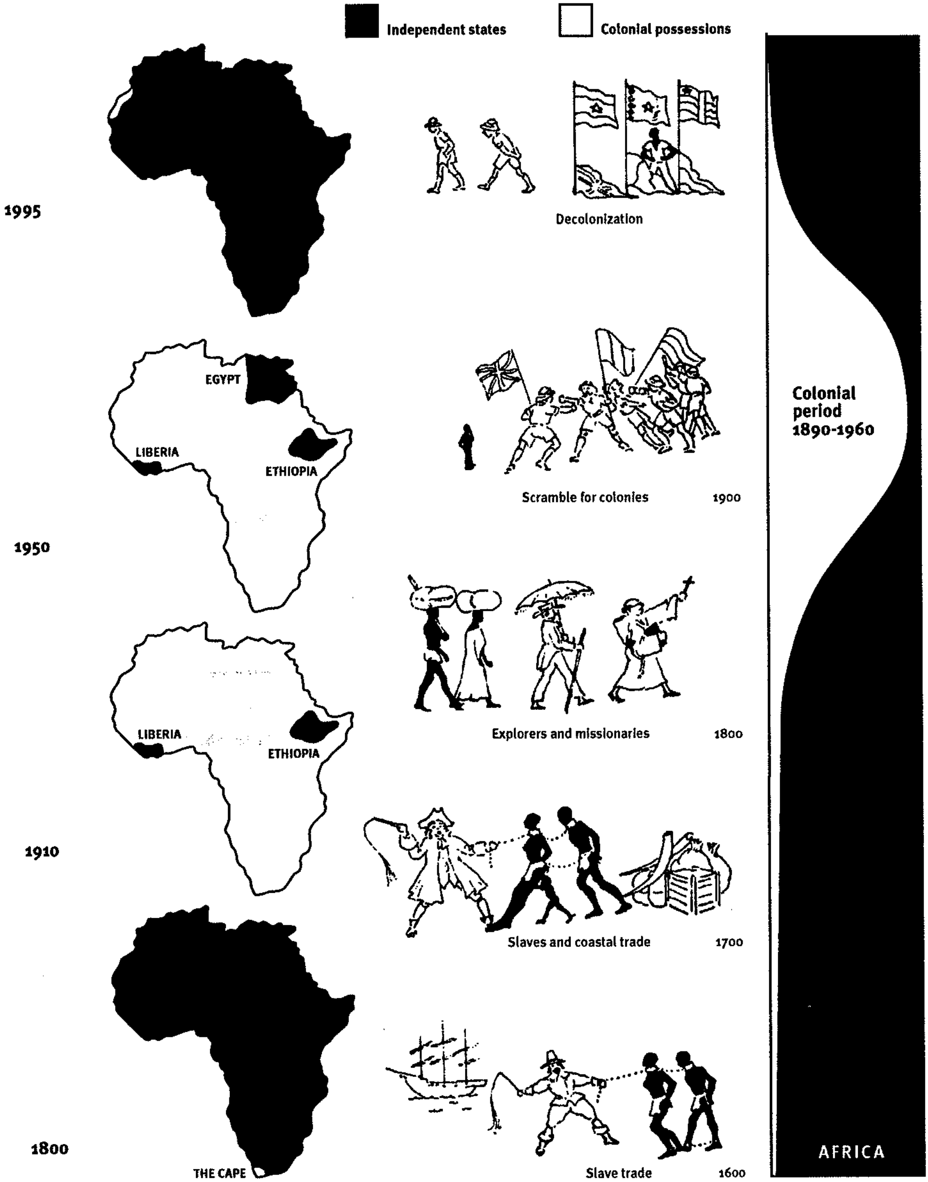


Figure 4.13: Africa: 1600 - 1995 (modified after Moss 1978).

The partitioning of Africa culminated in the Berlin Conference of 1914 which produced much of the present day 'national' boundaries between African countries. Moss (1978) describes the boundaries created as being haphazard and in some cases, bisecting local communities who found themselves in two different states following different sets of foreign laws. Interests of the local communities were not taken into consideration.

The European states owned the territories they acquired courtesy of this partitioning exercise. They owned the land, the resources and even the people! The purpose of colonisation, as viewed from the south, was to gain access to free resources, both material and human. The ultimate results were that the settlers took possession of the means of production, imposed their values, laws, culture and traditions. They discouraged or even banned cultural practices of the indigenous peoples. The indigenous people lost possession of their most valuable resource - land. They also lost a lot of their traditions and culture, among them, those cultural practices that were designed to promote sustainable use of resources. They could no longer practise ecosystem management in the ways they were accustomed to.

Forty years ago, African colonies began to fight for freedom from colonial rule. The nationalist leaders appealed to their people for support. Their rallying points were, *inter alia*, that the settlers/colonialists had (i) deprived the masses of their means of production (ii) deprived the masses of access to resources with which to produce wealth (iii) put in place rules and regulations that suited the needs of the colonialist and had no relevance to the locals and (iv) completely dehumanised the natives by despising, suppressing or even banning of some of the their cultural practices, etc.

Struggles for independence were motivated by a desire by the liberators to return the means of production which, in many African countries meant land, back to the African people, thereby restoring some of the lost dignity. Methods of achieving independence from colonial rule included negotiations, civil disobedience, and, in other cases, protracted armed conflicts. These struggles for independence were accompanied by a process of politicising the African peoples. Basically all that the colonialist had put in place was wrong and should be disobeyed until the colonialist gave up political power to the masses. The laws of the settler governments were to be disobeyed unselectively.

Laws and regulations that had relevance to ecosystem management were not exempt from the general civil disobedience as part of the independence struggles. If the colonialist ordered that peasant farmers should dig contour ridges in their fields, the liberators told the people not to. If the colonialist ordered people not to till land sloping into river banks, the liberators told the people exactly the opposite, and so on. The liberators instructed the peasantry to fill up dip tanks with soil because the colonialist wanted the peasants to dip their cattle. According to the colonialist, dipping cattle controlled ticks, while according to the liberators, dipping cattle gave the colonial administrators data with which they would order de-stocking. In countries like South Africa, Zimbabwe and Namibia which gained independence more recently, the native people's attitude towards resource conservation and ecosystem management are still very much influenced by the fervour of the liberation struggles.

Against the political background briefly summarised above, the settlers came with ideas, concepts and practices of resource use and management which did not necessarily blend with those traditionally practised by the natives. To their credit, the governments did try to implement conservation and ecosystem management strategies. However, it appears that in the majority of cases, native populations were not educated on the merits of the new technologies. It also appears that the settlers totally ignored whatever conservation practices native populations had. Natives hunted wildlife for meat. Hunting expeditions were organised in groups with the approval of the Chief. In a lot of cases there were well defined hunting seasons, the beginning of which was marked by elaborate ceremonies. The Lozi people of western Zambia under Chief Lewanika still carry out this practice. In most other southern African communities, hunting traditions were completely banned.

Land tenure

The land tenure system introduced by the settlers has been at the core of almost all liberation struggles in Africa. In Zimbabwe and South Africa the natives (blacks) were separated into 'their own areas' called Tribal Trust Lands in Zimbabwe or Homelands in South Africa. Elsewhere in colonial Africa the term 'Reserve' was commonly used to refer to those parts of the country 'reserved' for Africans. The areas into which the natives were confined were mostly characterised by infertile unproductive soils and/or unfavourable climatic conditions. The creation of these reserves for the natives invariably meant displacement of the native people. Any resistance to these forced migrations and displacements were dealt with ruthlessly by the settler government authorities.

Table 4.4. Pre-independence land policy in Zimbabwe (modified from Rukuni, 1994)

Year	Land Act	Purpose	Summary of Effects
1889	Limpert Concession	White settlers to acquire land rights from natives	British South Africa Company (BSACo) bought concession and used it as a basis for land appropriation
1898	Native Reserves Order in Council	To create Native Reserves in the face of mass land acquisition by white settlers	Native Reserves created haphazardly in areas of low productive potential
1930	Land Apportionment Act (LAA)	Legally separate land for white settlers and land for black natives	Majority of the best productive land became white-owned large scale commercial farms
1951	Native Land Husbandry Act	Enforce private ownership of land, destock & introduce conservation practices to native small land holders	Mass resistance by natives, fuelling nationalist politics. Law was scrapped in 1961
1965	Tribal Trust Lands	Change Native Land to TTL and create trustees for the land	High human & livestock populations in TTLs result in massive land degradation
1969	Land Tenure Act	Replace LAA (1930). Divide land 50/50 between minority whites and majority blacks	This, combined with the TTL Act was equivalent to the apartheid policy of South Africa

Table 4.5. Agro-ecological classification, Zimbabwe					
Region	**Area (km^2)**	**% of total**	**Annual rainfall (mm)**	**Production**	**Farming system**
1 Specialised and diversified farming	7000	2	1000	forestry, intensive livestock, tea, coffee, macademia nuts and other plantation crops	74% large scale commercial, 24% communal land, 2% small scale commercial.
2 Intensive farming	58600	15	750 – 1000	intensive livestock	74% large scale commercial, 22% communal land, 4% small scale commercial.
3 Semi-intensive farming	58600	15	650 – 800	cash crops (tobacco cotton), livestock	49% large scale commercial, 43% communal land, 8% small scale commercial.
4 Semi-intensive farming	58600	38	450 – 600	drought resitant crops, livestock	34% large scale commercial, 62% communal land, 4% small scale commercial.
5 Extensive farming	58600	27	too low and erratic even for drought resistant crops	extensive livestock production or game ranching	35% large scale commercial, 45% communal land, 20% national parks and game reserves

The land tenure system in Zimbabwe, then Southern Rhodesia, provides a case example. Table 4.4 summarises the history of the pre-independence land policy in Zimbabwe. Much of the problems associated with sound natural resources/ecosystem management stem from this history. The greatest post-independence expectation by the native population was the redress of this very unequal distribution of land between natives and settlers which resulted out of this land policy. The struggle for independence had access to land and return of land to the natives as the key issues.

The Land Apportionment Act of 1930 legally separated land between black and white. In the post-independence era, the government has passed two Acts of Parliament in attempt to address the land issue. The first is the Communal Lands Act of 1981 which changed the Tribal Trust Lands to Communal Lands and transferred authority over land from traditional chiefs to elected District Councils. The second is the Land Acquisition Act of 1985 and subsequent amendments. By this Act, the government has the right to purchase land from large scale commercial farms and use it to resettle natives. The problem of land ownership still exists in the sense that communal land dwellers do not individually own the land they use. This still belongs to the state.

Productivity of tracts of land can be deduced either from soil fertility, or from levels of precipitation or both. On this basis, Zimbabwe is divided into five agro-ecological regions (Table 4.5). Agro-ecological region 1 is potentially the most productive, while region 5 is potentially the least productive. The areas of high productive potential were reserved for whites, who constituted less that 3% of the population while the majority blacks were restricted to the marginal areas.

Up until independence in 1980, large scale commercial farms invariably meant land privately-owned by white settler farmers. There were no white people who lived in the land described in Table 4.5 as 'communal land'. It is quite clear from Table 4.5 that most of the more productive agro-ecological zones I and II were reserved for white settlers while the majority black population lived communally in the less productive agro-ecological zones IV and V.

Ownership

The land tenure system described allowed individual ownership of land, and resources therein, in the large scale commercial farming areas while the state owned all the land, and resources therein, in the communal areas. The latter remained an open access system while the former was structured to give exclusive rights to owners. With regards to resource exploitation, the communal areas therefore suffered what has been earlier referred to as the tragedy of the commons. In the absence of clearly defined proprietorship over resources, resource/ecosystem management was virtually impossible. No single user, or even group of users, could take responsibility of managing the resources/ecosystems on which they depended. A general consequence of this tenure system was that resource (ecosystem) management was the responsibility of the resource owner who, in the case of the large scale (white-owned) commercial farms, was also the resource user while in the communal lands the resource users were not, *sensu strictu*, the land owners.

The communal areas were characterised by highly degraded environments manifested by massive soil erosion, overgrazing of pastures and excessive deforestation. Wildlife, in particular were severely depleted in the communal areas. The natives, in the communal areas looked with envy at what appeared to be abundant resources in the white-owned large scale commercial farms. Natives poached wildlife, fuelwood and other natural resources in the large scale commercial farms. In 1975, the Government of Rhodesia gave land owners (white) authority to own and utilise wildlife on their land which had hitherto belonged to the state. Private land owners could then manage their lands and other resources for their direct benefit. Some large scale commercial farmers adopted wildlife production as an addition to their normal/usual farming. At present, there are numerous private game parks in Zimbabwe where hunting and photographic safaris are carried out as a major economic activity. There was an incentive to properly manage their resources/ecosystems.

As no single person or group of persons legally owned land in the communal areas, it was difficult to identify management units that could enjoy benefits from similar incentive of ecosystem management. This notwithstanding, the Department of National Parks and Wildlife Management introduced a programme called Communal Areas Management Programme for Indigenous Resources, now widely known by its acronym, CAMPFIRE. The objectives of this programme are, *inter alia*, to restructure the control of Zimbabwe's countryside giving people alternative ways of using their natural resources, to give communal people resource ownership

In order to contain and halt the spread of the cattle ling disease, the government ordered the total destruction of all cattle in the district. They were shot, doused in kerosene, burnt and buried in mass graves. At least a quarter of million cattle were to be killed. Some political activists suggested to the people that the government was eradicating cattle so as to promote expansion of wild animal populations in the Okavango Delta and so promote tourism.

The way forward

Only when the local people realise direct economic benefit from the ecosystem in which they live will they accept, appreciate and participate in natural resources/ecosystem management that has tourism as one of its objectives. They must be empowered, through proprietorship of the resource/ecosystems, to manage their ecosystems. They must also be empowered, through education and training at various levels, to be able to make, understand and implement technical decisions on ecosystem management.

The CAMPFIRE programme is one of several community based natural resources management programmes that have been developed in post-independent Africa. They have their strengths in being participatory and in giving a measure of proprietorship to the ecosystem users. The ecosystem users define their own management objectives and set out to achieve them. Such programmes also have strength from recognising traditional knowledge and practices and blending these with scientific theory and modern scientific advice. A key concept in these efforts is that of sustainability.

The success of these ecosystem management programmes derives from increasing and improving awareness about the consequences of various ecosystem use options, giving proprietorship to the ecosystem users who develop accountability for their action or lack thereof, and that ecosystem users derive direct benefit in economic terms from managing their ecosystems. The ecosystem users and managers must view themselves as integral components of the ecosystems that they manage.

REFERENCES

Åberg, J., Jansson, G., Swenson, J. E., Angelstam, P. 1995. The effect of matrix on the occurrence of hazel grouse (*Bonasa bonasia*) in isolated habitat fragments. *Oecologia* 103:265-269

Acreman, M. C. In Press. Principles of water management for people and the environment. American Association for the Advancement of Science.

Acreman, M. C. and Lahmann, E. 1995. Water management and protected areas. *Parks* 5(12): 1-5

Adamson, J. K. and Hornung, M. 1990. The effect of clearfelling a Sitka spruce (*Picea sitchensis*) plantation on solute concentrations in drainage water. *Journal of Hydrology* 116: 287-297.

Aiken, J., Williams, R., Bellan, I. and Burt, R. 1995. Measurements of upper ocean structure using aquashuttle from ships of opportunity and research vessels. Oceans 95: Challenges of our changing global environment. Marine Technology Society, Washington DC, USA 1, 502-519

Alverson, D. L., Freeberg, M. H. Murawski, S. A. and Pope, J. G. 1994. A global assessment of fisheries by catch and discard. *FAO Technical Report* 339. FAO, Rome, Italy

Akeroyd, J. 1995. Towards a spirit of wilderness. *Plant Talk* 3: 2

Andersen, U. V. 1995. Succession and soil development in man-made coastal ecosystems at the Baltic Sea. *Nordic Journal of Botany* 15: 91-104

Arrow, K., Bolin, B., Costanza, R., Dasgupta, P., Folke, C., Holling, C. S., Jansson, B. -O., Levin, S., Maler, K. -G., Perrings, C. and Pimentel, D. 1995. Economic growth, carrying capacity, and the environment. *Science* 268:520-521

Atkinson, T. C., Briffa, K.R. and Coope, G. R. 1987. seasonal temperatures in Britain during the past 22,000 years reconstructed using beetle remains. *Nature* 325: 587-593

Balaganskaya, E. D. and Lysnes, H. 1995. Restoration of polluted soils. *Ecosystem Research Report 10: Global Change and Arctic Terrestrial Ecosystems.* Office for Official Publications of the European Union, Luxembourg

Balmford, A. 1996. Extinction filters and current resilience: the significance of past selection pressures for conservation biology. *Trends in Ecology and Evolution* 11: 19–196

Balmford, A. and Long, A. 1994. Avian endemism and forest loss. *Nature* 372:623-624

Barbier, E., Acreman, M. C. and Knowler, D. 1997. *Economic Evaluation of Wetlands: a guide for policy makers and planners.* Ramsar, Gland, Switzerland

Barnola, J. M., Raynaud, D., Korotkevich, Y. S . and Lorius, C. 1987. Vostoc ice core provides 160,000-years record of atmospheric CO_2. *Nature* 329: 408-414

Battarbee, R. W. 1995 (Editor). *Acid Rain and its Impact: The Critical Loads Debate.* Ensis Publishing, London

Batterbee, R. W. , Mason, J., Renberg, I. and Talling, J. F. (Editors). 1990. *Paleolimnology and Lake Acidification.* Royal Society, London, UK

Bazely, D. R. and Jefferies, R. L. 1989. Leaf and shoot demography of an arctic stoloniferous grass, Puccinella phryganodes, in response to grazing. *Journal of Ecology* 77: 811-822

Beerling, D. J., Chaloner, W. G., Huntley, B., Pearson, J. A. and Tootly, M. J. 1993. Stomatal density responds to the glacial cycle of environmental change. *Proceedings of the Royal Society of London, B* 251:133-138

Bell, G., Lechowicz, M. J., Appenzeller, A., Chandler, M., DeBlois, E., Jackson, L., Mackenzie, B., Preziosi, R., Schallenberg, M. and Tinker, N. 1993. The spatial structure of the physical environment. *Oecologia* 96:114-121

Bell, J. N. B. 1994 (Editor). *The Ecological Effects of Increased Aerial Deposition of Nitrogen.* British Ecological Society Ecological Issues No. 5. Field Studies Council, Montford Bridge

Bennett, K. D. 1990. Milankovitch cycles and their effects on species in ecological and evolutionary time. *Paleobiology* 16: 11-21

Bernabo, J. C. and Webb, T., III. 1977. Changing patterns in the Holocene pollen record from northeastern North America: a mapped summary. *Quaternary Research* 8: 64-96

Bierregaard, R. O. 1996. Biological dynamics of forest fragments. In *The Conservation Atlas of Tropical Forests: The Americas* edited by C. S. Harcourt and J. A. Sayer. Simon and Schuster, New York and London

Bond, W. J. 1993. Keystone species. In *Biodiversity and Ecosystem Function* edited by E. D. Schulze and H. A. L. Mooney. Springer, Berlin, Germany

Botkin, A. D. 1990. *Discordant Harmonies: a New Ecology for the Twenty-first Century*. Oxford University Press, New York, USA

Bradshaw, A. D. 1987. Restoration: an acid test for ecology. In *Restoration Ecology - a Synthetic Approach to Ecological Research* edited by W. R. Jordan, M. E. Gilpin, and J. D. Aber. Cambridge University Press, Cambridge, UK

Bradshaw, A. D. 1995. Introduction. In *Restoration Ecology in Europe* edited by K. M. Urbanska and K. Grodzinska. Geobotanical Institute SFIT., Zurich, Switzerland

Bradshaw, A. D. 1997. Importance of soil in restoration science. pp. 33-64 in *Restoration Ecology and Sustainable Development* edited by K. M. Urbanska, N. R. Webb and P. J. Edwards. Cambridge University Press, Cambridge, UK

Bradshaw, A. D. and Chadwick, M. J. 1980. *The Restoration of Land*. Blackwell Scientific Publications, Oxford, UK

Broker, W. 1963. Genetisch-physiologische *Untersuchungenm über die Zinkvertr_glichkeit von Silene inflata Sm. Flora* 153:122-156

Brooks, T. M. and Balmford, A. 1996. Atlantic forest extinctions. *Nature* 380(6570): 115

Brown, K. S. and Brown, G. G. 1992. Habitat alteration and species loss in Brazilian forests. In *Tropical Deforestation and Species Extinction* edited by T. C. Whitmore and J. A. Sayer. Chapman and Hall, London, UK

Budiansky, S. 1994. Extinction or miscalculation. *Nature* 370: 105

Caldwell, L. 1970. The ecosystem as a criterion for public land policy. *Natural Resources Journal* 10:203-221.

Carlisle, A., Brown, A. F. H. and White, E. J. 1966. The organic matter and nutrient elements in the precipitation beneath a sessile oak (*Quercus petrae*) woodland. *Journal of Ecology* 54: 87-98

Carson, R. 1962. *Silent Spring*. Hamish Hamilton, London, UK

Cebrian, J. and Duarte, C. M. 1995. Plant growth-rate dependence of detrital carbon storage in ecosystems. *Science* 268:1606-1608

Chambers, R. 1993. *Challenging the Professions. Frontiers for rural development*. Intermediate Technology Publications, London, UK

Chappellaz, J., Barnola, J. M., Raynaud, D., Korotkevich, Y. S. and Lorius, C. 1990. Ice-core record of atmospheric methane over the part 160,000 years. *Nature* 345: 127-131

Chesson, P. and Huntly, N. 1989. Short-term instabilities and long-term community dynamics. *Trends in Ecology and Evolution* 7: 107-108

Child, B. 1996. The practice and principles of community-based wildlife management in Zimbabwe: the CAMPFIRE programme. *Biodiversity and Conservation* 5: 369 - 398

Clarkson, D. T. 1966. Aluminium tolerance in species within the genus Agrostis. *Journal of Ecology* 54: 167-178

CLAG. 1995. Critical Loads of Acid Deposition for United Kingdom Freshwaters. Critical Loads Advisory Group, Department of the Environment, London, UK

CLIMAP Project Members 1984. The last Interglacial Ocean. *Quaternary Research* 21: 123-224

COHMAP 1988. Climatic changes of the last 18,000 years: Observations and model simulations. *Science* 241: 1043-1052

Collar, N. J., Crosby, M. J. and Stattersfield, A. J. 1994. *Birds to Watch 2. The World List of Threatened Birds.* BirdLife International, Cambridge

Collingham, Y.C. 1995. *The development of a spatially explicit landscape-scale model of migration and its application to investigate the response of trees to climate change.* Ph.D. thesis, Durham, UK

Collingham, Y. C., Hill, M. O., Huntley, B. 1996. The migration of sessile organisms: a simulation model with measurable parameters. *Journal of Vegetation Science* 7:831-846

Conway, T. J., Tans, P., Waterman, L. S., Thoning, K. W., Masarie, K. A. and Gammon, R. H. 1988. Atmospheric carbon dioxide measurements in the remote global troposphere 1981-1984. *Tellus* 40B: 81-115

Coope, G. R. 1978. Constancy of insect species versus inconstancy of Quaternary environments. In *Diversity of Insect Faunas.* Symposia of the Royal Entomological Society of London 9 edited by L. A. Mound and N. Waloff. Blackwell Scientific, Oxford, UK

Coope, G. R. 1995. The effects of Quaternary climatic changes on insect populations: lessons from the past. In *Insects in a Changing Environment* edited by R. Harrington and N. E. Stork. Academic Press, London, UK

Corlett, R. T. and Turner, I. M. 1997. Long-term survival in tropical forest remnants in Singapore and Hong Kong in tropical forest remnants In *Ecology, Management and Conservation of Fragmented Communities* edited by W. F. Laurence, R. O. Bierregaard and C. Moritz. University of Chicago Press, Chicago, USA

Craighead, F. 1979. *Track of the Grizzly.* Sierra Club Books, San Francisco, California.

Critical Loads Advisory Group. 1995. *Critical Loads of Acid Deposition for United Kingdom Freshwaters.* Critical Loads Advisory Group, Department of the Environment, London, UK

Czech, B. and Krausman, P. R. 1997. Implications of an ecosystem management literature review. *Wildlife Society Bulletin* 25:667-675.

Davis, M. B. 1981. Quaternary history and the stability of forest communities. In *Forest Succession. Concepts and Application* edited by D. C. West, H. H. Shugart, and D. B. Botkin. Springer-Verlag, New York, USA

Davis, M. B. 1986. Climatic instability, time lags, and community disequilibrium. In *Community Ecology* edited by J. Diamond and T. J. Case. Harper and Row, New York, USA

Davis, M. B. 1989. Lags in vegetation response to greenhouse warming. *Climatic Change* 15: 75-82

Davis, S. D., Heywood, V. H. and Hamilton, A. C. (Editors) 1994–97. *Centres of Plant Diversity– a Guide and Strategy for their Conservation.* Vol. 1 (1994) Europe, Africa, South West Asia and The Middle East. Vol. 2 (1995) Asia, Australasia and the Pacific. Vol. 3 (1997) The Americas. WWF/IUCN, IUCN Publications Unit, Cambridge, UK

De-Beaulieu, J.-L. and, Reille, M. 1984. A long upper Pleistocene pollen record from Les Echets, near Lyon, France. *Boreas* 13: 111-132

Denslow, J. S. 1985. Disturbance-mediated coexistence of species. In *The Ecology of Natural Disturbance and Patch Dynamics* edited by S. T. A. Pickett and P. S. White. Academic Press, Orlando FL, USA

Denton, G. H. and Hughes, T. J. (Editors) 1981. *The Last Great Ice Sheets.* Wiley, New York, USA

Diamond, J. M. and May, R. M. 1981. Island biogeography and the design of natural reserves. In *Theoretical Ecology. Principles and Applications* edited by R. M. May. Blackwell Scientific, Oxford

Didham, R. K., Ghazoul, J., Stork, N. E. and Davis, A. J. 1996. Insects in fragmented forests: a functional approach. *Trends in Ecology and Evolution* 11:255-260

Dinerstein, E. and Wikramanayake, E. D. 1993. Beyond 'hotspots': how to prioritize investments in biodiversity in the Indo-Pacific region. *Conservation Biology* 7:701-704

Dinerstein, E., Wikramanayake, E. D. and Forney, M. 1995. Conserving the reservoirs and remnants of tropical moist forest in the Indo-Pacific region. In *Ecology, Conservation, and Management of Southeast Asian Rainforests* edited by R. B. Primack and T. E. Lovejoy. Yale University Press, New Haven, USA

DoE. 1990. *Acid Deposition in the United Kingdom 1986-1988.* Department of the Environment, London, UK

Drayton, B. and R. Primack. 1996. Plant species lost in an isolated conservation area in metropolitan Boston from 1894 to 1993. *Conservation Biology* 10:30-39

Duarte, C. M., Cebrian, J. and Marba, N. 1992. Uncertainty of detecting sea change. *Nature* 356:190

Duc, L. D. 1991 *Proceedings of the Workshop on Melaleuca Rehabilitation and Management, Long Xuyen, An Giang, 14-18 May 1991.* Agricultural Publishing House, Hanoi, Vietnam

Dudley, N. 1992. *Forests in Trouble: A review of the status of temperate forests worldwide.* Worldwide Fund for Nature, Gland, Switzerland

Dugan, P. J. 1990. *Wetland conservation: a review of current issues and required action.* IUCN, Gland, Switzerland

Ehrlich, P. R., Ehrlich, A. H. and Holdren, J. P. 1977. *Ecoscience: Population, resources, environment.* Freeman, San Francisco

Ehrlich, P. R. and Wilson, E. O. 1991. Biodiversity studies: Science and policy. *Science* 253:758-762

Elias, S. A. 1991. Insects and climate change. *BioScience* 41:552-559

Ewel, J. J. 1987. Restoration is the ultimate test of ecological theory. In *Restoration Ecology - a Synthetic Approach to Ecological Research* edited by W. R. Jordan, M. E. Gilpin, and J. D. Aber. Cambridge University Press, Cambridge, UK

Fairbanks, R. G. 1989. A 17,000-year glacio-eustatic sea level record: Influence of glacial melting rates on the Younger Dryas event and deep-ocean circulation. *Nature* 342: 637-642

FAO 1995. The state of world fisheries and aquaculture. FAO, Rome, Italy

Feldman, G. *et al.* 1989. Ocean color. Availability of the global data set. Eos, Trans Am geophys. Union 70:634-641

Flüeler, P. R. 1992. Experimentelle Untersuchungen über Keimung und Etablierung von alpinen Leguminosen. *Ver_ffentlichungen des Geobotanischen Institutes der ETH Stiftung Rübel Zürich*, No. 110

Folland, C. K., Karl, T. R. and Vinnikovk K. Y. 1990. Observed climate variations and change. In *Climate Change: The IPCC Scientific Assessment* edited by J.T. Houghton, G.J. Jenkins and J.J. Ephraums. Cambridge University Press, Cambridge, UK

Foster, D. R., Schoonmaker, P. K. and Pickett, S. T. A. 1990. Insights from palaeoecology to community ecology. *Trends in Ecology and Evolution* 5:119-122

Frankel, O., Brown, A. H. D. and Burdon, J. J. 1995. *The Conservation of Plant Diversity.* Cambridge University Press, Cambridge, UK

Gamble, J.C. 1994. Long-term Planktonic Time Series as Monitors of Marine Environmental Change. In *Long-term Experiments in Agricultural and Ecological Sciences* edited by R. A. Leigh and A. E. Johnston. CAB International, Wallingford, UK

Garrels, R. M. and Lerman, A. 1981. Phanerozoic cycles of sedimentary carbon and sulphur. *Proceedings of the National Academy of Sciences U.S.A.* 78: 4652-4656

Gasser, M. 1989. Bedeutung der vegetativen Phase bei alpinen Pflanzen für die biologische Erosionsbek_mpfung in der alpinen Stufe. *Berichte des Geobotanischen Institutes ETH, Stiftung Rübel, Zürich* 55: 151-176

Gaston, K. J. 1994. *Rarity.* Chapman and Hall, London, UK

Gaston, K. J. and Blackburn, T. M. 1995. Birds, body size and the threat of extinction. *Philosophical Transactions of the Royal Society of London, B* 347:205-212

Gates, D. M. 1993. *Climate Change and its Biological Consequences.* Sinauer Associates, Sunderland, MA, USA

Gentry, A. H. 1982. Neotropical floristic diversity: phytogenetic connections between Central and South America, Pleistocene climatic fluctuations, or an accident of the Andean orogeny? *Annals of the Missouri Botanical Garden* 69: 557-593

GLOBEC. 1997. Global Ocean Ecosystem Dynamics (GLOBEC) Science Plan. IGBP Report 40, GLOBEC Report 9. International Geosphere – Biosphere Programme, Stockholm, Sweden

Goldsmith, B. (Editor). 1991. *Monitoring for Conservation and Ecology.* Chapman and Hall, New York, USA

Gomez-Pompa, A., Vasquez-Yanes, C and Guevara, S. 1972. The tropical rain forest: a non-renewable resource. *Science(Washington D.C.)* 177: 762-765

Good, R. 1931. A theory of plant geography. *New Phytologist* 30: 11-171

Graham, R. W. and Grimm, E. C. 1990. Effects of Global Climate Change on the Patterns of Terrestrial Biological Communities. *Trends in Ecology and Evolution* 5: 289-292

Graham, R. W., Lundelius, E. L., Jr., Graham, M. A., Schroeder, E. K., Toomey, R. S., III, Anderson, E., Barnosky, A. D., Burns, J. A., Churcher, C. S., Grayson, D. K., Guthrie, R. D., Harington, C. R., Jefferson, G. T., Martin, L. D., McDonald, H. G., Morlan, R. E., Semken, H. A., Jr., Webb, D., Werdelin, L. and Wilson, M. C. 1996. Spatial response of mammals to late Quaternary environmental fluctuations. *Science* 272:1601-1606

Granat, L., Rodhe, H. and Hallberg, R. O. 1976. The global sulphur cycle. *Ecological Bulletins (Stockholm)* 22: 89-134

Gregory, R. P. G. and Bradshaw, A. D. 1965. Heavy metal tolerance in populations of *Agrostis tenuis Sibth.* and other grasses. *New Phytologist* 64: 131-143

Grime, J. P. 1978. *Plant strategies and vegetation processes.* John Wiley and Sons, New York, USA

Grime, J. P. and Hunt, R. 1975. Relative growth-rate: its range and adaptive significance in a local flora. *Journal of Ecology* 63: 393-422

Guiot, J., de Beaulieu, J. L., Cheddadi, R., David, F., Ponel, P. and Reille, M. 1993. The Climate in Western Europe During the Last Glacial Interglacial Cycle Derived from Pollen and Insect Remains. *Palaeogeography Palaeoclimatology Palaeoecology* 103: 73-93

Gustafsson, M. and Lannér-Herrera, C. 1997. Overview of the *Brassica oleracaea* complex: their distribution and ecological specificities. In *Proceedings of the Workshops on the Conservation of the Wild Relatives of European Cultivated Plants. (Bocconea 7)* edited by B. Valdés, V. H. Heywood, F. M. Raimondo and D. Zohary. Azienda Foreste Demaniali della Regione Siciliana, Palermo, Italy

Hallbacken, L. and Tamm C. O. 1986. Changes in soil acidity from 1927 to 1982-1984 in a forest area of south-west Sweden. *Scandinavian Journal of Forest Research* 1: 219-232

Halley, J. M. 1996. Ecology, evolution, and 1/f-noise. *Trends in Ecology and Evolution* 11:33-37

Hannah, L., Lohse, D., Hutchinson, C., Car, J. L. and Lankerani, A. 1994. A preliminary inventory of human disturbance of world ecosystems. *Ambio* 23: 46–250

Hanski, I. 1994. Spatial scale, patchiness and population dynamics on land. *Philosophical Transactions of Royal Society London B* 343:19–25

Hanski, I., Moilanen, A. and Gyllenberg, M. 1996. Minimum viable metapopulation size. *American Naturalist* 147:527-541

Hanski, I., Pakkula, T., Kuussaari, M. and Guangchun, L. 1995. Metapopulation persistence of an endangered butterfly in a fragmented landscape. *Oikos* 72:21-28

Hardin, G. 1968. The tragedy of the commons. Science CLXII:713 - 727

Harper, J. L. 1987. The heuristic value of restoration ecology. In *Restoration Ecology - a Synthetic Approach to Ecological Research* edited by W. R. Jordan, M. E. Gilpin, and J. D. Aber. Cambridge University Press, Cambridge, UK

Harrison, S. 1994. Metapopulations and conservation. In *Large-scale Ecology and Conservation Biology* edited by P. J. Edwards, R. M. May and N. R. Webb. Blackwell Science, Oxford, UK

Hasler, A. R. 1992. Experimentelle Untersuchungen über klonal wachsende alpine Leguminosen. *Ver_ffentlichungen des Gebobotanischen Institutes der ETH Stiftung Rübel Zurich* No. 111

Heywood, V.H. (Editor) 1995. *The Global Biodiversity Assessment.* United Nations Environment Programme. Cambridge University Press, Cambridge, UK

Heywood, V. H., Mace, G. M., May, R. M. and Stuart, S. N. 1994. Uncertainties in extinction rates. *Nature* 368: 105

Heywood, V. H. and Stuart, N. 1992. Species extinctions in tropical forests. In *Tropical Deforestation and Species Extinction* edited by T. C Whitmore and J. A. Sayer. Chapman and Hall, London, UK

Hobbs, R. and Lleras, E. 1995. Chapter 13.4 Protecting and restoring ecosystems, species, populations and genetic diversity. In *Global Biodiversity Assessment* edited by V. H. Heywood. Cambridge University Press, Cambridge, UK

Holligan, P.M., Aarup, T. and Groom, S.B. 1989. The North Sea: Satellite Colour Atlas. *Continental Shelf Research* 9(8): 667-765.

Holling, C. S. 1973. Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics* 4:1-23

Hornby, A. S. 1974. *Oxford Advanced Learner's Dictionary of Current English.* Oxford University Press, Oxford, UK

Hornung, M. and Skeffington R. A. 1993. *Critical Loads: Concepts and Applications.* ITE Symposium 28, HMSO, London, UK

Houghton, J. T., Meira Filho, L. G., Callander, B. A., Harris, N., Kattenberg, A. and Maskell, K. (Editors) 1996. *Climate Change 1995: The Science of Climate Change.* Cambridge University Press, Cambridge, UK

House of Lords Select Committee on Science and Technology. 1996. *Fish stock conservation and management.* Session 95-96, HL Paper 25, 2nd Report. Stationary Office, UK

Hunt, E. G. and Bischoff, A. I. 1960. Inimical effects on wildlife of periodic DDD applications to Clear Lake. *California Fish and Game* 1:91-106

Hunter, M. L., Jacobson, G. L. and Webb, T., III 1988. Palaeoecology and the coarse-filter approach to maintaining biological diversity. *Conservation Biology* 2: 375-385

Huntley, B. 1988. Glacial and Holocene vegetation history: Europe. In *Vegetation History* edited by B. Huntley and T. Webb, III. Kluwer Academic Publishers, Dordrecht, The Netherlands

Huntley, B. 1990. Studying global change: The contribution of Quaternary palynology. *Palaeogeography, Palaeoclimatology, Palaeoecology (Global and Planetary Change Section)* 82: 53-61

Huntley, B. 1991. How plants respond to climate change: migration rates, individualism and the consequences for plant communities. *Annals of Botany* 67: 15-22

Huntley, B. 1995. Plant species' response to climate change: implications for the conservation of European birds. *Ibis* 137 (Supplement 1): 127-138

Huntley, B. 1996. Quaternary palaeoecology and ecology. *Quaternary Science Reviews* 15:591-606

Huntley, B. In press. The influence of a changing climate. pp. 3-11 in *Ecology and Landscape Development: History of the Mersey basin* edited by E. F. Greenwood. National Museums and Galleries of Merseyside, Liverpool, UK

Huntley, B., Berry, P. M., McDonald, A. P., Cramer, W. P. 1995. Modelling present and potential future ranges of some European higher plants using climate response surfaces. *Journal of Biogeography* 22:967-1001

Huntley, B. and Birks, H. J. B. 1983. *An atlas of past and present pollen maps for Europe: 0-13000 B.P.* Cambridge University Press, Cambridge, UK

Huntley, B. Collingham, Y. C., Hill, M. O. 1996. Tree species' responses to environmental changes during the last glacial: evaluating alternative hypotheses using a spatially-explicit migration model. *Italian Journal of Quaternary Science* 9:617-626

Huntley, B. and Webb, T., III 1989. Migration: species' response to climatic variations caused by changes in the earth's orbit. *Journal of Biogeography* 16: 5-19

IGBP. 1992. *Global Change Reducing Uncertainties*. International Geosphere-Biosphere Programme, The Royal Swedish Academy of Sciences, Stockholm.

Imbrie, J., Berger, A., Boyle, E. A., Clemens, S. C., Duffy, A., Howard, W. R., Kukla, G., Kutzbach, J., Martinson, D. G., McIntyre, A., Mix, A. C., Molfino, B., Morley, J. J., Peterson, L. C., Pisias, N. G., Prell, W. L., Raymo, M. E., Shackleton, N. J. and Toggweiler, J. R. 1993. On the structure and origin of major glaciation cycles. 2. the 100,000-year cycle. *Palaeoceanography* 8: 699-735

Imbrie, J., Boyle, E. A., Clemens, S. C., Duffy, A., Howard, W. R., Kukla, G., Kutzbach, J., Martinson, D. G., McIntyre, A., Mix, A. C., Molfino, B., Morley, J. J., Peterson, L. C., Pisias, N. G., Prell, W. L., Raymo, M. E., Shackleton, N. J. and Toggweiler, J. R. 1992. On the structure and origin of major glaciation cycles. 1. Linear responses to Milankovitch forcing. *Palaeoceanography* 7: 701-738

Interagency Ecosystem Management Task Force 1995. *The Ecosystem Approach: Healthy ecosystems and sustainable economies.* US Department Commerce, Springfield, VA, USA

Intergovernmental Oceanographic Commission (IOC) and Sir Alister Hardy Foundation for Ocean Science (SAHFOS). 1991. *Monitoring the health of the ocean: Defining the role of the Continuous Plankton Recorder in global ecosystem studies.* UNESCO, Paris, France

Intergovernmental Oceanographic Commission (IOC). 1993. Assessment and Monitoring of Large Marine Ecosystems. UNESCO: 1-51.

IPCC. 1995. *Climate Change 1995. IPCC Scientific Assessment.* Cambridge University Press, Cambridge, UK

IUCN. 1995. *IUCN Red List Categories.* IUCN, Gland, Switzerland

IUCN/UNEP/WWF 1993. *Caring for the Earth: A strategy for survival.* Mitchell Beazley, London, UK

Jalas, J. and Suonminen, J. (Editors) 1972. *Atlas Florae Europaeae*: Vol. 1. Pteridophyta (Psilotaceae to Azollaceae). Societas Biologica Fennica Vanamo, Helsinki, Finland

Jalas, J. and Suonminen, J. (Editors) 1976. *Atlas Florae Europaeae*: Vol. 3. Salicaceae to Balanophoraceae. Societas Biologica Fennica Vanamo, Helsinki, Finland

Jefferies, R. L., Gadallah, F. L., Srivastava, D. S, and Wilson, D. J. 1995. Desertification and trophic cascades in arctic coastal ecosystems: A potential climate change scenario? *European Commission Ecosystem Research Report 10: Global Change and Arctic Terrestrial Ecosystems.* Office for Official Publications of the European Union, Luxembourg

Jensen, M. E. and Bourgeron, P. S. 1994 (Editors). *Volume II: Ecosystem Management: principles and applications.* Gen. Tech. Rep. PNW-GTR-318, USDA, Forest Service, Portland, USA

Johnson, I. Elliott, C, Acreman, M. C. 1996. *Ecological acceptable flows Phase II. Report to Environment Agency.* Institute of Hydrology, Wallingford, UK

Jones, C. G. and Lawton, J. H. 1995 (Editors). *Linking Species and Ecosystems.* Chapman and Hall, New York

Jowett, D. 1964. Population studies on lead tolerant Agrostis tenuis. *Evolution* 18: 70- 80

Katenin, A. E. 1995. Plant communities of *Populus balsamifera* as a testimony of more favourable past climatic conditions. *European Commission Ecosystem Report 10: Global Change and Arctic Terrestrial Ecosystems*, pp. 249-252. Office for Official Publications of the European Union, Luxembourg.

Keigley, R. B. 1988. Developing methods of restoring vegetation communities while preserving genetic integrity. Proceedings of 8th High Altitude Revegetation Workshop, *Colorado State University Information Series* 61: 129-138

Knight, D. H. 1975. A phytosociological analysis of species-rich tropical forest on Barro Colorado Island, Panama. *Ecological Monographs* 45: 259-284

Koopowitz, H., Thornhill, A. D. and Andersen, M. 1994. A general stochastic model for the prediction of biodiversity losses based on habitat conversion. *Conservation Biology* 8: 425–438

Korner, C. 1996. The response of complex multispecies systems to elevated CO^2. In *Global Change and Terrestrial Ecosystems* edited by B. H. Walker, and W. L. Steffen. Cambridge University Press, Cambridge, UK

Kotanen, P. and Jefferies, R. L. 1987. The leaf and shoot demography of grazed and ungrazed plants of Carex subspathacea. *Journal of Ecology* 75:961-975

Kutzbach, J. E. and Guetter, P. J. 1986. The influence of changing orbital parameters and surface boundary conditions on climatic simulations for the past 18,000 years. *Journal of the Atmospheric Sciences* 43: 1726-59

Lande, R. and Schemske, D. W. 1985. The evolution of self-fertilisation and inbreeding depression in plants. I. Genetic models. *Evolution* 39: 24-40

Lawton, J. H. 1994. What do species do in ecosystems? *Oikos* 71:367-374

Lawton, J. H. 1995. Population dynamic principles. In *Extinction Rates* edited by J. H. Lawton and R. M. May. Oxford Univ. Press, Oxford, UK

Lawton, J. H. 1996. Patterns in ecology. *Oikos* 75: 145-147.

Lawton, J. H. 1996. Population abundances, geographic ranges and conservation: 1994 Witherby Memorial Lecture. *Bird Study* 43:3-19

Lawton, J. H. and Brown, V. K. 1993. Redundancy in ecosystems. In *Biodiversity and Ecosystem* Function edited by E. -D. Schulze, and H. A. Mooney. Springer-Verlag, Berlin

Lawton, J. H. and Jones, C. G. 1993. Linking species and ecosystem perspectives. *Trends in Ecology and Evolution* 8: 311–313

Lawton, J. H. and May, R. M. 1995 (Editors). *Extinction Rates.* Oxford University Press, Oxford, UK

Leopold, A 1949. *A Sand County Almanac.* Oxford University Press. New York, USA

Likens, G. E., Bormann, F. H., Pierce, R. S., Eaton, J. S, and Johnson, N. M. 1977. *Biogeochemistry of a Forested Ecosystem.* Springer-Verlag, New York, USA

Lister, A. M. 1993. 'Gradualistic' evolution: Its interpretation in Quaternary large mammal species. *Quaternary International* 19: 77-84

Llamas, M. R. 1988. Conflicts between wetland conservation and groundwater exploitation: Two case histories in Spain. *Environ. Geol. Water Science* 11: 241-251

Longhurst, A. 1985. The structure and evolution of plankton communities. Prog. Oceanog. 15, 1-35

Longhurst, A. 1995. Seasonal cycles of pelagic production and consumption. *Prog. Oceanog.* 36: 77-167.

Lovelock, J. E. 1979. Gaia: *A new look at life on Earth.* Oxford University Press, Oxford, UK

Lovelock, J. E. 1989. *The Ages of Gaia.* Oxford University Press, Oxford, UK

Lubchenco, J., Olson, A. M., Brubaker, L. B., Carpenter, S. R., Holland, M. M., Hubbell, S. P., Levin, S. A., MacMahon, J. A., Matson, P. A., Melillo, J. M., Mooney, H. A., Peterson, C. H., Pulliam, H. R., Real, L. A., Regal, P. J. and Risser, P. G. 1991. The sustainable biosphere initiative: an ecological research agenda. *Ecology* 72:371-412.

MacArthur, R. H. and Wilson, E. O. 1967 *The Theory of Island Biogeography*. Princeton University Press, Princeton, USA

Malassé, A. D. 1993. Continuity and discontinuity during hominization. *Quaternary International* 19: 85-98

Malone, C. R. 1996. The Federal ecosystem management initiative in the USA. In *Environmental Sustainability: case studies on the prospects of science and ethics* edited by J. Lemons, R. Goodland and L. Westra. Kluwer Academic Publishers, Dordrecht, Netherlands

Maltby, E. 1991. Wetlands - their status and role in the biosphere. In *Plant Life Under Oxygen Deprivation* edited by M. B. Jackson, D. D. Davies and H. Lambers. S. P. B. Academic Publishing, The Hague, The Netherlands

Maltby, E., Blackwell, M. S. A and Hogan, D. V. 1997. Wetlands in the landscape - linking policy to function. In *Wetlands: Landscape and institutional perspectives. proceedings of the 4th workshop of the Global Wetlands Economics Network (GWEN)* edited by Tore Soderqvist. Beijer International Institute of Ecological Economics, The Royal Swedish Academy of Sciences, Sweden

Maltby, E., Hogan, D. V., Immirzi, C.P., Tellam, J. H & van der Peiji, M. J. 1994. Building a new approach to the investigation and assessment of wetland ecosystem functioning. pp 637-658 in *Global Wetlands: Old World and New* edited by W. J. Mitsch. Elsevier, Amsterdam, The Netherlands

Maltby, E., Hogan, D. V. and McInnes, R. J. 1996. *Functional Analysis of European Wetland Ecosystems. Phase I Final Report. European Commission Ecosystems Research Report* 18. European Commission, Luxembourg

Mangel, M., Talbot, L. M., Meffe, G. K. et al 1996 Principles for the conservation of wild living resources. *Ecological Applications* 6(2): 338-362.

Margules, C. R. 1987. Single large or several small reserves? In *Nature Conservation: The Role of Remnants of Native Vegetation* edited by D. A. Saunders, G. W. Arnold, A. A. Burbidge and A. J. M. Hopkins. Surrey Beatty and Sons Pty in association with CSIRO and CALM

Masundire, H. M.; K. N. Eyeson and S. F. Mpuchane (Editors) 1995. *Proceedings of the conference on wetlands management in Botswana.* Botswana Wetlands Co-ordinating Committee, Kasana, Botswana

Matiza, T. and H. N. Chabwela. 1992. *Wetlands conservation conference for southern Africa: Proceedings of the SADCC conference held in Gaborone, Botswana.* IUCN, Gland, Switzerland

May, R. M. and Oster, G. F. 1976 Bifurcation and dynamic complexity in simple ecological models. *American Naturalist* 110: 573-599.

McGlone, M. S. 1988. Glacial and Holocene vegetation history: New Zealand. In *Vegetation History: Vol. 7* edited by B. Huntley and T. Webb, III. Kluwer Academic Publishers, Dordrecht, The Netherlands

McGlone, M. S., Salinger, M. J. and Moar, N. T. 1993. Palaeovegetation studies of New Zealand's climate since the last glacial maximum. In *Global Climates since the Last Glacial Maximum* edited by H. E. Wright, Jr., J.E. Kutzbach, T. Webb, III, W. F. Ruddiman, F. A. Street-Perrott and P. J. Bartlein. University of Minnesota Press, Minneapolis, USA

McNeely, J. A. 1995. Chapter. 11.2 The impact of human activity on biodiversity in *Global Biodiversity Assessment* edited by V. H. Heywood. Cambridge University Press, Cambridge, UK

Meisterhans, E. 1988. Vegetationsentwicklung auf Skipistenplanierungen in der alpinen Vegetationsstufe bei Davos. *Ver_ffentlichungen des Geobotanischen Institutes ETH Stiftung Rübel Zürich* No. 97

Miller, K. R. 1996. *Balancing the Scales: Guidelines for increasing biodiversity's chances through bioregional management.* World Resources Institute, Washington DC.

Mills, L. S., Soulé, M. E. and Doak, D. F. 1993. The keystone species concept in ecology and conservation. *Bio Science* 43: 219-224

Milner-Gulland, E. J. 1994 A population model for the management of saiga antelope. *Journal of Applied Ecology* 31: 25-39.

Milton, S. J. and Dean, W. R. J. 1995. How useful is the keystone species concept, and can it be applied to *Acacia erioloba* in the Kalahari Desert? *Zeitschrift für Oekologie und Naturschutz* 4: 147-156

Mitchell, J. F. B. 1983. The seasonal response of a general circulation model to changes in CO2 and sea temperatures. *Quarterly Journal of the Royal Meteorological Society* 109:113-152

Mitchell, J. F. B., Manabe, S., Meleshko, V. and Tokioka, T. 1990. Equilibrium climate change - and its implications for the future. In *Climate Change: The IPCC Scientific Assessment* edited by J. T. Houghton, G. J. Jenkins and J. J. Ephraums. Cambridge University Press, Cambridge, UK

Moloney, K. A. and Levin, S. A. 1996. The effects of disturbance architecture on landscape-level population dynamic. *Ecology* 77: 375-394

Mooney, H. A. and Koch, G. W. 1994. The impact of rising CO2 concentrations on the terrestrial biosphere. *Ambio* 23:74-76

Moran, G. F. and Hopper, S. D. 1987. Conservation of the genetic resources of rare and widespread eucalypts in remnant vegetation. In *Nature Conservation: The role of remnants of native vegetation* edited by D. A. Saunders, G. W. Arnold, A. A. Burbidge and A. J. M. Hopkins. Surrey Beatty and Sons Pty in association with CSIRO and CALM

Morris, D. W. 1995. Earth's peeling veneer of life. *Nature* 373:25

Moss, B., Balls, H. and Irvine, K. 1985. *Isolation of Broads as a Technique for Restoration. BARS 11.* Broads Authority, Norwich, UK

Moss, P. 1978. *Modern World History.* Granada Publishing, New York, USA

Murphree, M. W. 1991. Communities as institutions for resource management. *Paper Presented at the National Conference on Environment and Development,* Maputo, Mozambique

Naeem, S., Thompson, L. J., Lawler, S. P., Lawton, J. H. and Woodfin, R. M. 1994. Declining biodiversity can alter the performance of ecosystems. *Nature* 368:734-737

Naeem, S., Thompson, L. J., Lawler, S. P., Lawton, J. H. and Woodfin, R. M. 1995. Empirical evidence that declining biodiversity may alter the performance of terrestrial ecosystems. *Philosophical Transactions of the Royal Society of London, B* 347:249-262

Naiman, R. J. and H. Decamps (Editors). 1990. *The ecology and management of aquatic-terrestrial ecotones.* Parthenon Publishing Group, New York, USA

Neenan, M. 1960. The effects of soil acidity on the growth of cereals with particular reference to the differential reaction of varieties thereto. *Plant and Soil* 12: 324-338

Newmark, W. D. 1985. Legal and biotic boundaries of Western North American national parks: A problem of congruence. *Biological Conservation* 33:197-208.

Newsham, K. K., Frankland, J. C., Boddy, L. and Ineson, P. 1992. Effects of dry deposited sulphur dioxide on fungal decomposition of angiosperm litter. I. Changes in communities of fungal saprotrophs. *New Phytologist* 122: 97-110

Nilsson, Ch. and Grelsson, G. 1995. The fragility of ecosystems: a review. *Journal of Applied Ecology* 32: 677-692

North Sea Task Force. 1993. North Sea Quality Status Report. Oslo and Paris Commissions, London. Olsen and Olsen, Fredensborg, Denmark.

Noss, R.F. and A.Y. Cooperrider, 1994. *Saving Nature's Legacy: Protecting and Restoring Biodiversity.* Island Press, Washington DC.

Nozette, S. and Lewis, J. S. 1982. Venus: chemical weathering of igneous rocks and buffering of atmospheric composition. *Science* 216: 181-183

Odum, E. P. 1969. The strategy of Ecosystem Development. *Science* 164:262-270.

Onions, C. T. (Editor). *The Shorter Oxford English Dictionary.* Clarendon Press, Oxford, UK

Organisation for Economic Co-operation and Development (OECD). 1994. *Oceanography.* OECD, Paris, France

Owen, T. and Biemann, K. 1976. Composition of the atmosphere at the surface of Mars: Detection of argon-36 and preliminary analysis. *Science* 193: 801-803

Paice, J. P. 1974. *The Ecological History of Grizedale Forest, Cumbria, with Particular Reference to Tilia cordata Mill.* M.Sc. Thesis, University of Lancaster, UK

Paige, K. N. and Whitham, T. G. 1987. Overcompensation in response to mammalian herbivory: the advantage of being eaten. *American Naturalist* 129: 407-416

Paine, R. T. 1969. A note on trophic complexity and community stability. *American Naturalist* 103: 91-93

Parker, V. T. and Pickett, S. T. A. 1997. Restoration as an ecosystem process: implications of the modern ecological paradigm. pp 17-32 in *Restoration Ecology and Sustainable Development* edited by K. M. Urbanska, N. R. Webb and P. J. Edwards. Cambridge University Press, Cambridge, UK

Pauly, D. and Christensen, V. 1995. Primary production required to sustain global fisheries. *Nature* 374:255-257

Phillips, O. L. and Gentry, A. H. 1994. Increasing turnover through time in tropical forests. *Science* 263:954-958

Pickett, S. T. A. and Parker, V. T. 1994. Avoiding the old pitfalls - opportunities in a new discipline. *Restoration Ecology* 2: 180- 189

Pickett, S. T. A., Parker, V. T. and Fiedler, P. 1992. The new paradigm in ecology: implications for conservation biology above the species level. In *Conservation Biology: the theory and practice of nature conservation, preservation and management* edited by P. Fiedler and S. Jain. Chapman and Hall, New York. USA

Pigott, C. D. 1991. Biological Flora of the British Isles: *Tilia cordata Miller. Journal of Ecology* 79: 1147-1207

Pigott, C. D. 1992. Are the distributions of species determined by failure to set seed? In *Fruit and Seed Production* edited by C. Marshall and J. Grace. Cambridge University Press, Cambridge, UK

Pigott, C. D. and Huntley, J. P. 1981. Factors controlling the distribution of Tilia cordata at the northern limits of its geographical range. III. Nature and causes of seed sterility. *New Phytologist* 87: 817-839

Pimm, S. 1984. The complexity and stability of ecosystems. *Nature* 307:321-326

Pimm, S. 1991. *The Balance of Nature?* University of Chicago Press, Chicago, USA

Pimm, S. 1996. The lonely Earth. *World Conservation* 1/96: 8–9

Pimm, S. L. and Lawton, J. H. 1977. Number of trophic levels in ecological communities. *Nature* 268:329-331

Pimm, S. L. and Redfern, A. 1988. The variability of population densities. *Nature* 334:613-614

Pimm, S. L., Rusell, G. J., Gittleman, J. L. and Brooks, T. M. 1995. The future of biodiversity. *Science* 269:347-350

Pirot, J-Y. and Meynell, P. J. 1998. *Ecosystem Management: Lessons from around the World.* The World Bank, Washington D. C., USA

Planque, B. and Ibanez, F. 1996. Longterm-time series in Calanus finmarchicus abundance – a question of space? Oceanologica Acta 29, 159-164

PNCS (1992) *The Pakistan National Conservation Strategy.* IUCN, Karachi, Pakistan

Poffenberger, M. and McGean, B. (Editors) 1996. Village Voices, Village Choices. Joint forest management in India. Oxford University Press, Oxford, UK

Polunin, N. and Worthington, E. B. (1990) On the use and misuse of the term 'ecosystem'. *Environmental Conservation* 17(3):174

Prat, S. 1934. Der Erblichkeit der Resistenz gegen Kupfer. *Berichte der Deutschen Botanischen Gesellschaft* 52: 65-67

Primack, R. 1998. *Essentials of Conservation Biology. 2nd Edition.* Sinaur Associates, Sunderland, Massachusetts, USA

Quartley, C. and Reid, P.C. 1996. Long-term oceanographic datasets. *Sea Technology* 37(3):68-70.

Rahel, F. J. 1990. The hierarchical nature of community persistence: a problem of scale. *American Naturalist* 136: 328-344

Rainboth, W. J. 1996. *FAO Species Field Guide for Fisheries - Fishes of the Cambodian Mekong.* FAO, Rome, Italy

Ramanathan, V. 1988. The greenhouse theory of climate change: A test by an inadvertent global experiment. *Science* 240: 293-299

Ratcliffe, D. 1977. *A Nature Conservation Review.* Cambridge University Press, Cambridge, UK

Reid, P.C. 1994. *The North Sea Ministerial Process: Conclusions of the Intermediate Ministerial Meeting, December 1993. Marine Environmental Management. Review of Events in 1993 and Future Trends* 1(20):81-86.

Reid, P. C. . The North Sea Ecosystem: Status Report. In The Gulf of Mexico Large Marine Ecosystem. Edited by H. Kumpf, K. Steidinger and K. Shermann. Blackwell Science, Inc. Oxford, UK

Reid, P. C. and Hunt, H. G. 1998. Are observed changes in the plankton of the North Atalantic and North Sea linked to climate change? 2nd International Conference on Pelagic Biogeography. IOC Workshop Report No. 142, 310-315. IOC/UNESCO, Paris, France

Reid, W. V. 1992. How many species will there be? In *Tropical Deforestation and Species Extinction* edited by T. C. Whitmore and J. A. Sayer. Chapman and Hall, London, UK

Reynolds, B., Stevens, P. A., Adamson, J. K., Hughes, S. and Roberts, J. D. 1992. Effects of clearfelling on stream and soil water aluminium chemistry in three UK forests. *Environmental Pollution* 77: 157-165.

Ritchie, J. C. and MacDonald, G. M. 1986. The patterns of post-glacial spread of white spruce. *Journal of Biogeography* 13: 527-540

Roemmich, D and McGowan, J. 1995. Climatic warming and the decline of zooplankton in the California Current. Science 267, 1324-1326

Roozen, A. J. M. and Westhoff, V. 1985. A study on long term salt marsh succession using permanent plots. *Vegetatio* 61: 23-32

Rousseau, D.-D. 1997. The weight of internal and external constraints on *Pupiliia muscorum* L (Gastropoda: Stylommatophora) during the Quaternary in Europe. In *Past and Future Rapid Environmental Changes: The spatial and evolutionary responses of terrestrial biota* edited by B. Huntley, W. Cramer, A. V. Morgan, H. C. Prentice and J. R. M. Allen. Springer-Verlag, Berlin, Germany

Rukuni, M. 1994. The evolution of Agricultural Policy: 1890 -1990. In *Zimbabwe's Agricultural Revolution* edited by M. Rukuni and C. K. Eicher. University of Zimbabwe Publications, Harare, Zimbabwe

Safford, R. J. and Maltby, E. 1997. The overlooked values of *Melaleuca* wetlands. In *Towards Sustainable Management of Tram Chim National Reserve, Vietnam. Proceedings of a workshop on balancing economic development with environmental conservation* edited by R. J. Safford, D. V. Ni, E. Maltby and V.-T. Xuan. Royal Holloway Institute for Environmental Research, London, UK

Safford, R. J., Ni, D. V., Maltby, E. & Xuan, V.-T. (Editors.) 1997. *Towards Sustainable Management of Tram Chim National Reserve, Vietnam.* Royal Holloway Institute for Environmental Research, Virginia Water, UK

Safriel, U. N., Anikster, Y. and Valdman, M. 1997 Management of nature reserves for conservation of wild relatives and the significance of marginal populations. In *Proceedings of the Workshops on the Conservation of the Wild Relatives of European Cultivated Plants. (Bocconea 7)* edited by B. Valdés, V. H. Heywood, F. M. Raimondo and D. Zohary. Azienda Foreste Demaniali della Regione Siciliana, Palermo, Italy

Sathyendranath, S., Longhurst, A., Caverhill, C.M. and Platt, T. 1995. Regionally and seasonally differentiated primary production in the North Atlantic. Deep-Sea Research 42, 1773-1802.

Saunders, D. A., Hobbs, R. J. and Margules, C. R. 1991. Biological consequences of ecosystem fragmentation: a review. *Conservation Biology* 5: 18–32

Schemske, D.W., Husband, B. C., Ruckelhaus, M. H., Goodwillie, C., Parker, I. M. and Bishop, J. G. 1994. Evaluating approaches to the conservation of rare and endangered plants. *Ecology* 75:584-606.

Schimel, D., Alves, D., Enting, I., Heimann, M., Joos, F., Raynaud, D., Wigley, T., Prather, M., Derwent, R., Enhalt, D., Fraser, P., Sanhueza, E., Zhou, X., Jonas, P., Charlson, R., Rodhe, H., Sadasivan, S., Shine, K. P., Fouquart, Y., Ramaswamy, V., Solomon, S., Srinivasan, J., Albritton, D., Isaksen, I., Lal, M. and Wuebbles, D. 1996. Radiative Forcing of Climate Change. In *Climate Change 1995: the science of climate change* edited by J.T. Houghton, L.G. Meira Filho, B. A. Callander, N. Harris, A. Kattenberg and K. Maskell. Cambridge University Press, Cambridge, UK

Schlesinger, M. E. and Zhao, Z.-C. 1989. Seasonal climatic changes induced by doubled CO2 as simulated by the OSU atmospheric GCM/mixed-layer ocean model. *Journal of Climate* 2: 459-495

Schlesinger, W. H. 1991 *Biogeochemistry: An Analysis of Global Change.* Academic Press, New York, USA

Sherman, K. 1994. Sustainability, biomass yields, and health of coastal ecosystems: An ecological perspective. Mar. Ecol. Prog. Ser. 112, 277-301

Simberloff, D. 1992. Do species–area curves predict extinction in fragmented forest? In *Tropical Deforestation and Species Extinction* edited by T. C. Whitmore and J. A. Sayer. Chapman and Hall, London, UK

Simon, J. 1994. *Scarcity or Abundance?*: A debate on the environment. (with N. Myers) W. W. Norton, New York, USA

Smith, F. A., Betancourt, J. L. and Brown, J. H. 1995. Evolution of body size in the woodrat over the past 25,000 years of climate change. *Science* 270: 2012-2014

Smith, R. A. 1872. *Air and Rain.* Longmans Green and Co. Ltd., London, UK

Soulé, M. E. 1986. Conservation biology and the "real world". In *Conservation Biology: The science of scarcity and diversity* edited by M. Soulé. Sinauer Ass., Sunderland, Mass. USA

Soulé, M. E. (Editor) 1987. *Viable Populations for Conservation.* Cambridge University Press, Cambridge

Spence, D. H. N. 1957. Studies on the vegetation of Shetland. *Journal of Ecology* 45: 917-945

Steadman, D. W. 1995. Prehistoric extinctions of Pacific island birds: biodiversity meets zooarchaeology. *Science* 267:1123-1131

Steele, J. H. 1985. A comparison of terrestrial and marine ecological systems. *Nature* 313:355-358

Steele, J. H. 1991. Can ecological theory cross the land-sea boundary? Journal of Theoretical Biology 153, 426-436

Strahm, W. 1989. *Plant Red Data Book for Rodrigues.* Koeltz Scientific Books, Konigstein, for IUCN, The World Conservation Union

Strahm, W. 1996. Conservation of the flora of the Mascarene Islands. *Curtis's Botanical Magazine* 13(4): 228-238

Street-Perrott, F. A. and Perrott, R. A. 1993. Holocene vegetation, lake levels, and climate of Africa. In *Global Climates since the Last Glacial Maximum* edited by H. E. Wright, Jr., J. E. Kutzbach, T. Webb, III, W. F. Ruddiman, F. A. Street-Perrott and P. J. Bartlein, University of Minnesota Press, Minneapolis, USA

Stuart, A. J. 1993. The failure of evolution: Late Quaternary mammalian extinctions in the Holarctic. *Quaternary International* 19: 101-107

Sweden's Case Study. 1971. *Air Pollution Across National Boundaries.* The Impact on the Environment. Royal Ministry of Foreign Affairs and Royal Ministry of Agriculture, Stockholm, Sweden

Tansley, A. G. 1935. The use and abuse of vegetational concepts and terms. *Ecology* 16:284-307

Taylor, A. H. 1995. North-south shifts of the Gulf Stream and their climatic connection with the abundance of zooplankton in the UK and its surrounding seas. ICES Journal of Marine Science 52, 711-721

Thomas, C. D. and Jones, T. M. 1993. Partial recovery of a skipper butterfly (*Hesperia comma*) from population refuges: lessons for conservation in a fragmented landscape. *Journal of Animal Ecology* 62:472-481

Thomas, C. D., Thomas, J. A. and Warren, M. S. 1992. Distribution of occupied and vacant butterfly habitats in fragmented landscapes. *Oecologia* 92:563-567

Thompson, R. S. 1988. Glacial and Holocene vegetation history: Western North America. In *Vegetation History: Vol. 7* edited by B. Huntley and T. Webb, III. Kluwer Academic Publishers, Dordrecht, The Netherlands

Thompson, R. S., Whitlock, C., Bartlein, P. J., Harrison, S. P. and Spaulding, W. G. 1993. Climatic changes in the western United States since 18,000 yr B.P. In *Global Climates since the Last Glacial maximum* edited by H. E. Wright, Jr., J. E. Kutzbach, T. Webb, III, W. F. Ruddiman, F. A. Street-Perrott and P. J. Bartlein. University of Minnesota Press, Minneapolis, USA

Tilman, D. 1996. Biodiversity: population versus ecosystem stability. *Ecology* 77:350-363

Tilman, D. and Downing, J. A. 1994. Biodiversity and stability in grasslands. *Nature* 367:363-365

Tilman, D., May, R. M., Lehman, C. L. and Nowak, M. A. 1994. Habitat destruction and the extinction debt. *Nature* 371:65-66

Tilman, D., Wedin, D. and Knops, J. 1996. Productivity and sustainability influenced by biodiversity in grassland ecosystems. *Nature* 379:718-720

Tinker, P.B. 1994. Monitoring Environmental Change Through Networks. In Long-term Experiments in Agricultural and Ecological Sciences edited by R. A. Leigh and A. E. Johnston. CAB International, Wallingford, UK

Tischkov, A. A. 1996. Secondary succession of arctic ecosystems in relation to tundra restoration. In *Disturbance and recovery in Arctic Lands: An ecological perspective* edited by R. M. M. Crawford. Kluver Publishers, Dordrecht, The Netherlands

Tschurr, R. F. 1990. Single-ramet-cloning (SRC) and multi-ramet-cloning (MRC) : an example of basic and applied revegetation research. *Proceedings of 9th High Altitude Revegetation Workshop: Colorado Water Resources Research Information Series* 63: 234-237

Tschurr, R. F. 1992. Experimentelle Untersuchungen über das Regenerationsverhalten bei alpinen Pflanzen. *Ver_ffentlichungen des Institutes ETH Sitftung Rübel Zurich* No. 108.

Turner, I. M. 1996. Species loss in fragments of tropical rain forest: a review of the evidence. *Journal of Applied Ecology* 33: 200-219

Turner, I. M. and Corlett, R. T. 1996. The conservation value of small, isolated fragments of lowland tropical rain forest. *Trends in Ecology and Evolution* 11:330-333

Turner, I. M., Tan, H. T. W., Wee, Y. C., Ali Bin Ibrahim, Chew, P. T. and Corlett, R. T. 1994. A study of plant species extinction in Singapore: lessons from the conservation of tropical biodiversity. *Conservation Biology* 8: 705–712

Turner, R. K. 1991. Wetlands and economic management. *Ambio* 20(2):59-63

Ulrich, B. 1986 Natural and anthropogenic components of soil acidification. *Zeitschrift fr Pflanzenernährung und Bödenkunde* 149: 702-717

Ulrich, B., Mayer, R. and Khann, P. K. 1980. Chemical changes due to acid precipitation in a loess-derived soil in central Europe. *Soil Science* 130: 193-199

Urbanska, K. M. 1995a. Ecological restoration above the timberline and its demographic assessment. In *Restoration Ecology in Europe* edited by K. M. Urbanska and K. Grodzinska. Geobotanical Institute SFIT, Zurich, Switzerland

Urbanska, K. M. 1995b. Biodiversity assessment in ecological restoration above the timberline. *Biodiversity and Conservation* 4: 679-695

Urbanska, K. M. 1996. Reproductive behaviour of arctic/alpine plants and ecological restoration. pp. 481-501 in *Disturbance and recovery in Arctic Lands: An ecological perspective edited* by R. M. M. Crawford. Kluver Publishers, Dordrecht, The Netherlands

Urbanska, K. M. 1997a. Restoration ecology of alpine and arctic areas: are the classical concepts of niche and succession directly applicable? pp. 189-200 in *Restoration Ecology and Sustainable Development* edited by K. M. Urbanska, N. R. Webb and P. J. Edwards. Cambridge University Press, Cambridge, UK

Urbanska, K. M. 1997b. Safe sites - interface of plant population ecology and restoration ecology. pp. 81-110 in *Restoration Ecology and Sustainable Development* edited by K. M. Urbanska, N. R. Webb and P. J. Edwards. Cambridge University Press, Cambridge, UK

Urbanska, K. M. Hefti-Holenstenstein, B. and Elmer, G. 1987. Performance of some alpine grasses in single-tiller cloning experiments and the subsequent revegetation trials above the timberline. *Berichte des Geobotanischen Institutes ETH Stiftung Rübel Zurich* 53: 64-90

Valdés, B., Heywood, V. H., Raimondo, F. M. and Zohary, D. (Editors) 1997. *Proceedings of the Workshops on the Conservation of the Wild Relatives of European Cultivated Plants. (Bocconea 7).* Azienda Foreste Demaniali della Regione Siciliana, Palermo, Italy

van der Maarel, E. Boot, R. van Dorp, D. and Rijnties, J. 1985. Vegetation succession on the dunes near Oostvoorne, The Netherlands: A comparison of the vegetation in 1959 and 1980. *Vegetatio* 58: 137-187

Van Dyne, G. M. 1969. *The Ecosystem Concept in Natural Resource Management.* Academic Press, New York, USA

Vilchek, G. E. and Bykova O. Y. 1992. The origin of regional ecological problems within the northern Tyumen oblast, Russia. *Arctic and Alpine Research* 24: 99-107

Vitousek, P. M. 1992. Global environmental change: An introduction. *Annual Review of Ecology and Systematics* 23:1-14

Vitousek, P. M. 1994. Beyond global warming: ecology and global change. *Ecology* 75:1861-1876

Vitousek, P., Ehrlich, P., Ehrlich, A. and Matson, P. 1986. Human appropriation of the products of photosynthesis. *BioScience* 36:368-373

Vogt, K. A., Gordon, J. C., Wargo, J. P., and Vogt, D. J. .1997. *Ecosystems: Balancing science with management.* Springer-Verlag Inc., New York, USA

Walker, D. 1989. Diversity and stability. In *Ecological Concepts* edited by J. M. Cherrett. Blackwell Scientific Publishers, Oxford, UK

Walter, H. 1960. *Standortlehre (Analytische-okologische Geobotanik). Einführung in die Phytologie; Grundlagen der Pflanzenverbreitung.* Ulmer, Stuttgart, Germany

Watt, K. E. F. 1968. *Ecology and Resource Management: A quantitative approach.* McGraw-Hill, New York, USA

Watts, W. A. 1988. Late-Tertiary and Pleistocene vegetation history: Europe. In *Vegetation History: Vol.7* edited by B. Huntley and T. Webb, III. Kluwer Academic Publishers, Dordrecht, The Netherlands

Watts, W. A., Allen, J. R. M., Huntley, B. 1996. Vegetation history and climate of the Last Glacial at Laghi di Monticchio, southern Italy. *Quaternary Science Reviews*

WCMC 1992. *Global Biodiversity: Status of the Earth's living resources.* Chapman and Hall, London, UK

Webb, T., III 1988. Glacial and Holocene vegetation history: Eastern North America. In *Vegetation History* edited by B. Huntley and T. Webb, III. Kluwer Academic Publishers, Dordrecht, The Netherlands

West, R. G. 1964. Inter-relations of ecology and Quaternary palaeobotany. *Journal of Ecology* 52 (Supplement): 47-57

Westman, W. E. 1991. Ecological restoration projects: measuring their performance. Environmental *Professional* 13: 207-215

Westra, L. 1994. *An Environmental Proposal for Ethics: The principle of integrity.* Rowman, Lanham, USA

Whitmore, T. C. 1975. Tropical Rainforests of the Far East. Oxford University Press, London, UK

Wilhalm, T. 1996. *Clonal Growth and Fragmentation in the Alpine Tussock-forming Grass Poa alpina.* PhD. Thesis. Swiss Federal Institute of Technology, Zurich, Switzerland

Willis, K. G. and Garrod, G. D. 1992. Amenity value of forests in Great Britain and its impact on the internal rate of return from forestry. *Forestry* 65(3): 331-346

Wood, C. A. 1994 Ecosystem management: achieving the new land ethic. *Renewable Resources Journal* 12(Spring): 6-12

Woodward, F. I. 1987. *Climate and Plant Distribution.* Cambridge University Press, Cambridge, UK

Wookey, P. A. and Ineson, P. 1991. Chemical changes in decomposing forest litter in response to atmospheric sulphur dioxide. *Journal of Soil Science* 42: 615-628

Wookey, P. A., Robinson, C. H., Parsons, A. N., Welker, J. M., Callaghan, T. V. and Lee, J. A. 1995. Environmental constraints on the growth, photosynthesis and reproductive development of Dryas octopetala at a high Arctic polar semi-desert, *Svalbard. Oecologia* 102: 478-489

World Bank. 1997. Animals under threat. *Dissemination Notes* 57. World Bank Environment Department, Washington D. C., USA

World Resources Institute. 1994. *World Resources 1994-95. A guide to the global environment.* Oxford University Press, Oxford, UK

Wright, H. E., Jr., Kutzbach, J. E., Webb, T., III, Ruddiman, W. F., Street-Perrott, F. A. and Bartlein, P.J. (Editors) 1993. Global Climates since the last glacial maximum. University of Minnesota Press, Minneapolis, USA