Economics and Ecosystems

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Efficiency, Sustainability and Equity in Ecosystem Management

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Preface

This book shows how the concepts of economic efficiency, sustainability and equity (in other words: people–planet–profit) can be applied in ecosystem management. The book provides an overview of the three concepts, presents a framework for modelling the efficiency, sustainability and equity of ecosystem management, and contains three case studies that illustrate the framework. It also examines how complex ecosystem dynamics, such as thresholds and irreversible responses, influence options for ecosystem management.

The book is based on my PhD dissertation 'Optimising the management of complex dynamic ecosystems: an ecological–economic modelling approach', which I defended in January 2005. The dissertation has been rewritten and updated with the intention of producing a more broadly relevant text, building on the practical experiences with environmental management that I gained as environmental advisor for the FAO/World Bank Investment Centre (1997–2002) and in Shell International (2007–2010).

The book is targeted at students and practitioners with an interest in ecosystem management. The book has a quantitative approach, and provides general formulas for analysing ecosystem dynamics and ecosystem services. The presented modelling framework can be used to quantify the economic efficiency, sustainability and equity of potential ecosystem management options.

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1. Introduction

1.1 THE CONTEXT

Environmental and natural resources worldwide are under pressure to meet demands for food, fresh water, fibre and energy (e.g. Balmford et al., 2002; Millennium Ecosystem Assessment, 2005). These pressures can be expected to further increase in the coming decades. Global population levels will increase from the current 6.5 billion to some 9 billion in 2050 (medium population scenario, UN, 2003), with a large majority of the increase occurring in developing countries. Growing production and consumption levels, in particular in China and India, will further increase the demand for natural resources. These trends are also reflected in the state of the world's ecosystems. Increasingly, the degradation of ecosystems affects human welfare (Millennium Ecosystem Assessment, 2005).

Hence, there is an urgent need for enhanced management of the remaining ecosystems. Identifying appropriate ecosystem management options requires understanding, among others, the ecological dynamics of ecosystems, the cultural, social and institutional context, and the economic costs and benefits of ecosystem management options. Three criteria are prominent in the evaluation of environmental management options: equity, sustainability and profitability – or, coined slightly differently: people, planet and profit. These three criteria for evaluating management options have now been endorsed by a broad range of actors, including governments, the private sector and NGOs.

Whereas the three criteria have become commonplace in environmental and resource management, their application is often constrained by difficulties in defining and measuring the profitability, sustainability and equity impacts of a policy or project. Assessing environmental management options requires an integrated approach combining insights from, among others, ecology, geography, economics and sociology. Since these disciplines tend to have different conceptual and methodological approaches, their integration is often not straightforward.

In recognition of the need to develop interdisciplinary research and assessment tools in support of environmental management, a number of integrated approaches have been developed, such as integrated (environmental) assessment, human ecology and ecological economics. Among these various approaches, ecological economics may most explicitly aim to integrate ecological and economic approaches in support of ecosystem management (Costanza and Daly, 1987). A key paradigm underlying ecological economics is that, ultimately, the world's natural resource base is finite, and that there is a need to better account for the increasing scarcity of natural resources in decision making (e.g. Boulding, 1966).

1.2 THE PURPOSE OF THIS BOOK

This book is targeted at students and professionals in the field of environmental management. It provides a framework for analysing the economic efficiency, sustainability and equity implications of ecosystem management options. Specifically, the book presents (1) an overview of how the concepts of efficiency, sustainability and equity can be used in relation to ecosystem management; (2) a general framework for the quantitative analysis and modelling of ecosystem management options; and (3) three case studies in which the framework is applied to assess management options for a specific ecosystem.

Specific attention is paid to complex, non-linear responses of ecosystems. Complex dynamics include, for instance, irreversible responses and/or thresholds in ecosystem responses to stress. They have been found to occur in a wide range of ecosystems including lakes, coastal estuaries, forests and rangelands. This book contains a general description of different types of complex ecosystem dynamics, indicates how these dynamics can be included in ecological–economic models, and examines the implications of different types of complex ecosystem dynamics for environmental management.

Around half of the book deals with the modelling of ecosystem management options in three case study sites. The case studies show how the described dynamic systems modelling approach can be applied to analyse the efficiency, sustainability and equity implications of ecosystem management options in a practical setting. The case study sites are, respectively, a hypothetical forest ecosystem, a Dutch wetland (De Wieden) and a semi-arid rangeland in Senegal (the Ferlo). The case studies also illustrate the mathematics that can be used to model ecosystem dynamics and ecosystem services supply.

1.3 GENERAL APPROACH OF THIS BOOK

This book presents a dynamic systems approach for analysing ecosystem management options. It combines insights from ecology, economics and, to some extent, policy studies. Particular topics covered in the book are ecosystem dynamics, ecosystem services analysis and valuation, stakeholder involvement and resource-use optimisation. The approach developed in this book can either be used as an analytical framework, or as a basis for modelling ecology–ecosystem interactions.

The book should be seen as being written in the context of ecological economics rather than environmental economics, even though the valuation approaches applied in the book are grounded in neoclassical welfare economics. Basic valuation approaches, as well as the key pitfalls and limitations of ecosystem valuation, are also briefly discussed. The main aim of the book is to provide guidance on the integrated analysis of efficiency, sustainability and equity aspects in ecosystem management. All three of these aspects provide information required for deciding on ecosystem management options and it is not implied that one of these criteria is, or should be, predominant in ecosystem management.

The book focuses on environmental management at the scale of the ecosystem. An ecosystem can be defined as 'the individuals, species and populations in a spatially defined area, the interactions among them, and those between the organisms and the abiotic environment' (Likens, 1992). Following the interpretation of the Millennium Ecosystem Assessment (2003), ecosystems comprise natural as well as strongly human-influenced systems, including croplands. Ecosystems have also been defined as a 'functional unit' with specific components, hierarchy and processes that distinguish it from other ecosystems. Modelling ecosystem dynamics requires capturing these key components and their interactions (Holling et al., 2002). Following the Millennium Ecosystem Assessment (2003), this book assumes that ecosystems can be identified across a range of spatial and temporal scales, ranging in size from a local fish pond up to the North Atlantic Ocean.

Ecosystem services are a central concept in this book, providing a link between the ecosystem and the economic system. In recent years, a rapidly increasing number of publications has provided frameworks and approaches for analysing and interpreting ecosystem services. In this book, the Millennium Ecosystem Assessment (2003, 2005) provides the main conceptual basis for analysing ecosystem services, with a number of minor deviations according to Hein et al. (2006).

Ecosystem management requires consideration of the impacts of management options on the dynamics and state of the ecosystem and, subsequently, the provision of ecosystem services by the ecosystem. This can only be meaningfully done based on an adequate consideration of the dynamics of the ecosystem – which are only very seldom linear and gradual, and much more often 'complex'. Complex dynamics include irreversible, non-linear and/or stochastic responses of the ecosystem to human and/or ecological drivers (e.g. Holling and Gunderson, 2002). Complex dynamics have been found to be crucial for explaining changes in, among others, freshwater lakes (Larsen et al., 1981; Timms and Moss, 1984; Scheffer, 1998), marine fish stocks (Steele and Henderson, 1984; Steele, 1998), woodlands (Dublin et al., 1990), rangelands (Friedel, 1991), coral reefs (Knowlton, 1992; Nyström et al., 2000) and coastal estuaries (Murray and Parslow, 1999).

The different chapters of this book provide different levels of detail on the concepts of efficiency, sustainability and equity in relation to ecosystem management. A basic description of these three concepts, as well as of ecosystem services and economic valuation of ecosystems, is provided in Chapter 2. Chapter 3 presents a dynamic systems modelling approach that can be used for the quantitative analysis of the economic efficiency, sustainability or equity aspects of ecosystem management options. The approach involves the construction of differential equations to capture ecosystem dynamics and ecosystem services supply in combination with ecosystem service valuation techniques.

Chapter 4 provides a first application of the framework and approach, for a hypothetical forest ecosystem. This chapter also further elaborates on the implications of pursuing efficiency versus sustainability in ecosystem management, as well as related topics such as the Safe Minimum Standard for ecosystem management. Chapter 5 presents a case study that involves pollution control in a specific wetland (De Wieden, the Netherlands), and Chapter 6 analyses efficient stocking rates in a semi-arid rangeland in the Sahel (the Ferlo, Senegal). For both ecosystems, economic efficient management strategies are identified, and sustainability and stakeholder implications of the various management options are discussed. Finally, Chapter 7 provides a general overview of how ecosystem services assessment and the proposed dynamic systems modelling approach can be applied to support environmental management.

2. Ecological–economic concepts

2.1 EFFICIENCY–SUSTAINABILITY–EQUITY

2.1.1 Introduction

In the last decades, a broad consensus has emerged that ecosystem management needs to consider and balance social, economic and environmental criteria (also expressed as people–profit–planet). In general terms, economic efficiency expresses the generation of welfare, based on an optimal use of natural resources and other production factors. Social criteria deal with such aspects as the distribution of welfare among people, and their involvement and representation in decision making. Environmental sustainability expresses, in general terms, whether the use of a natural resource does not exceed the regenerative capacity of that resource and if the resource is maintained at an adequate level to permit future uses.

This section describes these basic concepts of economic efficiency, equity and sustainability in more detail. Clearly, they are not the only criteria for decision making on ecosystem management. For instance, legal and technical criteria will often also determine the design of a project or management strategy. However, the three aforementioned criteria are among the most important ones for ecosystem management. In addition, ecosystem management often involves trade-offs between these criteria, which means that they need to be considered in an integrated manner.

The three concepts are, at times, difficult to apply, and a whole literature is devoted to each of them. This section provides a brief overview, focussing on their general principles and their implications for ecosystem management. In addition, Section 2.1.5 briefly discusses discounting in ecosystem management. Discounting involves the comparison of present and future costs and benefits, and is therefore a crucial element in examining the potential gaps between economic efficient and sustainable ecosystem management. Section 2.1.6 explores market failures and their implications for ecosystem management.

2.1.2 Efficiency in Ecosystem Management

In economics, an allocation of resources is said to satisfy the efficiency criterion if the net benefits from the use of those resources are maximised by that allocation (Tietenberg, 2000). For instance, in the case of reducing pollution in a lake, an efficient reduction of pollution loading involves analysing the economic costs of the pollution as a function of the degree of pollution (e.g. fish mortality), identifying the costs of waste-water treatment, and establishing the amount of pollution loading where the pollution and abatement costs are minimised (and the net benefits of the lake and its uses are maximised). In other words, efficient ecosystem management involves maximising the net economic benefits supplied by the ecosystem, considering both the benefits provided by the ecosystem and the costs of managing the ecosystem.

The ethical basis for assessing efficiency is derived from the Pareto criterion. Following this criterion, *static* economic efficiency implies the following. For some particular initial distribution of property rights, an allocation of resources is efficient if there is no feasible reallocation that can increase any person's utility without decreasing someone else's utility (e.g. Freeman, 1993). Utility indicates the relative satisfaction that a person gains from the consumption of a good or service. Utility can not be empirically observed or measured, and is applied as a relative measure, for instance, to compare the satisfaction levels a person gains from the consumption of different combinations of goods. A central construct of utility is that the utility gained by one additional unit of consumption of a certain good or service (e.g. a piece of chocolate) decreases when the total consumption level of that good or service increases (i.e. decreasing marginal utility). For reasons of simplicity, instead of utility, this book will generally refer to the net benefits of ecosystem management, even though utility is the theoretically more correct measure for analysing the efficiency of ecosystem management options.

There are usually many allocations that satisfy the Pareto criterion. Both Kaldor and Hicks further developed the Pareto approach to identify efficient allocations. According to the criterion proposed by Kaldor, a reallocation is efficient if it is possible for the winners to fully compensate the losers of the reallocation and still leave everyone better off. The Hicksian test asks whether it is possible for the losers to bribe the gainers to obtain their consent to forego the proposed reallocation. If the expected value of the reallocation of the resources for the gainers would be so high that it exceeds the maximum bribe that would be offered by the losers, the reallocation passes the Hicks efficiency criterion (Hicks, 1939). Hence, following the Kaldor–Hicks efficiency criterion, suboptimal allocations can always be rearranged so that some people are better off and no one is hurt by the rearrangement. Following the interpretation of Kaldor–Hicks, the efficient allocation is also optimal. However, additional provisions are needed to define optimal resource management in the case of intertemporal or intergenerational resource allocation, and to deal with social inequity, e.g. in case one stakeholder is poor and is not able to compensate a richer stakeholder for foregoing a loss resulting from the rearrangement of an allocation.

In the case of ecosystem management, the manager is often confronted with *intertemporal* allocation questions, for instance, in the case where it should be decided if a particular resource should be harvested now or at some moment in the future. The formulation of an intertemporal efficiency criterion requires the assumption that it is possible to define the aggregate utility of all living people over time. Given this, an allocation of resources over time is intertemporally efficient if, for some given level of utility at the present time, future utilities are at their maximum feasible levels. In this case, future utility can only be increased at the expense of the current utility. Howarth and Norgaard (1990) showed that effects of initial allocations on equity and efficiency readily translate from a static to an intergenerational context. Following standard neo-classical approaches, future and present costs and benefits can be compared through discounting. By discounting future costs and benefits, the efficient ecosystem management option can be determined, given a certain discount rate. Discounting is further discussed in Section 2.1.5. Note that another important factor in the analysis of intertemporal efficiency is technological progress. Technological progress may lead to a more efficient use of resources in the future, allowing, under a number of conditions, the maintenance of utility levels even at a decreasing capital stock. The topic of technological progress is outside the scope of this book and not further discussed, but see, for instance, Dasgupta (1993) for more information.

Taking income inequalities into account in the identification of optimal resource allocations requires the specification of a social welfare function. A social welfare function allows the analysis of the welfare implications of changes in income for different groups/income levels in a society. Social welfare functions reflect that, in general, an increase in income of 1 euro generates more utility for a poor person than the same increase for a richer person. A range of social welfare functions have been developed; see, for instance, Arrow (1963) and Sen (1970). When both intertemporal aspects and equity are to be considered in the identification of socially optimal allocations, an intergenerational social welfare function is required.

In the case of environmental and resource management, the mathematical basis for analysing the efficiency of resource use is provided by

Hotelling (1931). Hotelling examined how the social welfare from the exploitation of a *non-renewable* resource can be maximised over time. He argued that current extraction involves an opportunity cost, which equals the value that might have been obtained by extraction of the resource at a later date. The difference between the value of extraction in the future and the value of extraction at present is usually referred to as the scarcity rent of the resource. The 'Hotelling rule' states that resource extraction is intertemporally efficient if the increase in rent of the resource equals the social discount rate (Berck, 1995). In the analyses of the efficiency of *renewable* resources use, the growth of the resource needs to be accounted for. In a simple model, this growth depends upon the size of the stock in relation to the environment's carrying capacity for the species involved. For instance, Gordon (1954) and Schaefer (1957) prepared economic models for analysing the efficiency of a fishery, using simple logistic growth curves to describe the growth of the fish stock. Efficient ecosystem management needs to consider the costs of maintaining and managing ecosystems, as well as the benefits derived from ecosystems in the form of various ecosystem services (Odum and Odum, 1972; Bouma and Van der Ploeg, 1975; Hueting, 1980). In assessing the efficiency of ecosystem management, the full set of goods and services supplied by the ecosystem, including non-market benefits, should be considered.

2.1.3 Sustainability in Ecosystem Management

The Hotelling rule compares the intertemporal aspects of resource use on the basis of the social discount rate. However, even at low discount rates, the importance of the welfare of future generations rapidly diminishes. Because of the large weight discounting attaches to the welfare of current generations as compared to the welfare of future generations, this approach has been criticised as ethically questionable. In response to this shortcoming, the concept of *sustainability* was introduced. Sustainable development was first endorsed in the World Conservation Strategy proposed by UNEP and two environmental NGOs (IUCN/UNEP/WWF, 1980). The primarily ecological focus of the sustainable development concept used in the initial report was broadened in the widely known report 'Our Common Future' published by the World Commission on Environment and Development (the 'Brundtland report') in 1987 (WCED, 1987). The Brundtland commission defined sustainable development as: 'development that meets the needs of the present without compromising the ability of future generations to meet their own needs' (WCED, 1987). Even though the concept is now widely used, the interpretation of sustainable development and, hence, sustainability is not straightforward. This relates, for instance, to the interpretation of the concept 'need': Which consumption level can be regarded as sufficient to meet these needs? And which combination of production factors is required to ensure these needs?

Hence, subsequent to the Brundtland report, many studies have further examined the sustainability concept. A main issue in the interpretation of sustainable development is the assumed degree of substitutability between natural and man-made capital. This has been the subject of much research in environmental and ecological economics. For instance, Pearce et al. (1989), Barbier and Markandya (1990) and Daly (1990) assume a low degree of substitutability between natural and man-made capital. Pearce et al. (1989) and Barbier and Markandya (1990) state that sustainable development invokes maximisation of the benefits of economic development subject to maintaining the services and quality of natural resources over time. Along this line of reasoning, Daly (1990) argues that sustainability requires that: (1) harvest rates of renewable resources (e.g. fish, trees) not exceed regeneration rates; (2) use rates of non-renewable resources (e.g. coal, gas, oil) not exceed rates of development of renewable substitutes; and (3) rates of pollution not exceed the assimilative capacities of the environment.

Others have criticised this strong interpretation of sustainability. For instance, Beckerman (1994) assumed unlimited capital–resource substitutability, from which he derives that 'strong sustainability, overriding all other considerations, is morally unacceptable as well as totally impractical'. Dasgupta (1993) also argued that the substitution possibilities are high, driven by innovation and technological progress. Innovations continuously expand the possibilities to extract resource deposits, use resources in an efficient manner and recycle wastes.

If substitutability is assumed to be high, the well-known Hartwick rule offers some guidance on the maintenance of consumption levels under resource depletion: under many circumstances in a closed economy with non-renewable resources, the rent derived from resource depletion is exactly the level of capital investment that is needed to achieve constant consumption over time (Hartwick, 1977; Asheim, 1986). Hartwick's rule has been widely adopted in environmental policy – many governments have stated the importance of investing rents from natural resource depletion in building up capital in the rest of the economy (Pezzey and Toman, 2002).

An intermediate position on the interpretation of sustainability is that natural and man-made capital can be either substitutes or complements depending upon the characteristics of the economic system and the specific natural and man-made capital involved (e.g. Georgescu-Roegen, 1979; Cleveland and Ruth, 1997). In this view, the rate of substitutability depends, among others, upon the type of ecosystem service involved. For instance, the regulation of climate and biochemical cycles, as well as several cultural services, can only to a very limited extent be replaced by man-made capital (Costanza and Daly, 1992; Victor, 1994). Solow (1993) also follows a more intermediate position. He argues that it is not possible to preserve the full stock of natural capital and suggests a weaker definition of sustainability where partial substitution of man-made and natural capital is allowed.

The issue of substitutability in relation to renewable natural resources can be illustrated with the development of global fish stocks. The ongoing trend of 'fishing down the foodchain' (Pauly et al., 1998; Myers and Worm, 2003) indicates that the availability of fish, in particular top predators such as tuna, is likely to strongly decline in the coming decades. Different groups of people have different possibilities to substitute for declining fish resources (by switching to other fish, aquaculture fish, or other sources of protein). Besides the technical possibility of substituting natural for man-made capital, issues are the degree of substitution possible (taste of tuna versus, for example, cultivated salmon), and, in particular, the cost of substitution. For instance, many coastal populations in developing countries are not able to access alternative protein sources following the decline in fish stocks they traditionally depended upon (Alder and Sumaila, 2004). Hence, at the level of the ecosystem, substitution possibilities are likely to differ among stakeholders, with those groups that are natural resource dependent and with little capital to invest in adaptation being most vulnerable.

Based upon the assumed rates of substitutability, Carter (2001) classifies the different definitions of sustainability into four main categories: (1) very weak; (2) weak; (3) strong; and (4) very strong sustainability. Very weak sustainability allows for infinite substitution between natural and other capital (human and economic). In weak sustainability, it is recognised that certain life-supporting ecosystem services can not be replaced, but otherwise it allows for the substitution between different types of capital. Strong sustainability states that the total natural capital stock should not be further reduced, but that limited replacement of one type of natural capital with other types of natural capital is possible (e.g. reforestation may offset clear-cutting of forest in other locations, or even the destruction of a certain amount of coral reefs). Finally, very strong sustainability implies that no reduction of the stock and composition of natural capital is allowed (Carter, 2001). Other authors have linked sustainability to the maintenance of the integrity of the world's ecosystems. In this approach, particular attention is given to the dynamic relations between and among ecosystems, and the importance of the life-support services of ecosystems.

From this perspective, sustainable management is interpreted as management that maintains the resilience of ecosystems (Common and Perrings, 1992; Levin et al., 1998).

In this book, following the Brundtland definition and in line with Pearce et al. (1989), the following definition of sustainable ecosystem management is used: 'management that maintains the capacity of the ecosystem to provide future generations with the amount and type of ecosystem services at a level at least equal to the current capacity.' Among others, this definition implies that biodiversity in the ecosystem is maintained, for two reasons: (1) maintenance of (functional) biodiversity is required to support the functioning and resilience of the ecosystem (see Section 3.3.3); and (2) maintenance of biodiversity is required to sustain the biodiversity conservation service of ecosystems (see Section 2.2.2).

This definition of sustainable ecosystem management corresponds to a strong sustainability criterion (Carter, 2001). The selection of the strong sustainability criterion allows for an explicit comparison of the implications of pursuing different ecosystem management options, and avoids the risk of overestimating substitutability rates between natural and man-made capital (see above). Implicit in applying the sustainability concept is a view on dealing with environmental change in the time span of several generations. Hence, assessing sustainability will normally involve modelling the impact of ecosystem management on the state and the stability of the ecosystem, and its capacity to supply ecosystem services, over a prolonged time period.

2.1.4 Equity in Ecosystem Management

In social sciences, the concept of equity is often linked to, or used interchangeably with, fairness and justice (Konow, 2001). With regards to ecosystem management, equity aspects are particularly relevant with regards to: (1) sharing the benefits supplied by ecosystems; as well as (2) the representation of different stakeholders in the design and implementation of ecosystem management strategies. This is briefly discussed below.

Benefit sharing

Stakeholder groups often benefit from different ecosystem services, and changes in ecosystems tend to affect stakeholders in different manners (Hein et al., 2006). In addition, within groups of stakeholders, individuals may be affected in a different ways. For example, the poorest among local people residing in tropical forests tend to have few alternative income sources and therefore be most dependent on income from the sale of locally collected non-timber forest products (e.g. Campbell, 1996). Hence, the way in which ecosystem management options affect different stakeholders and individuals is a key concern for decision making. However, unlike economic efficiency or, to some extent, sustainability, it is difficult to quantify equity impacts in a simple metric.

A range of approaches towards defining equity have been proposed. On the one hand, it has been sought to define normative theories of justice, i.e. definition of justice on theoretical grounds (e.g. Rawls, 1971; Nozick, 1974; Baumol, 1986). An example of this interpretation is given by Nozick (1974): 'the principle of distributive justice is that a distribution is just if everyone is entitled to the holdings they posses under that distribution.' Just distribution may be related to income levels, to equal access to opportunities for generating income (Roemer, 1996), or to access to 'primary goods' (which include income and wealth, but also, for example, free speech and freedom of religion; Rawls, 1971).

On the other side of the spectrum, there are a range of authors stressing that equity is strongly shaped by context, including cultural values, precedent and the types of goods and services being distributed, and that each case requires a distinct interpretation of equity (e.g. Walzer, 1983; Young, 1994). Recent studies have combined the two approaches, seeking to describe justice and equity in general terms (e.g. Kahnemn et al., 1986; Konow, 2001). Based on the extensive literature on this topic, some general concepts underlying equity that are relevant to ecosystem management can be distilled:

- **The Basic Needs Principle**. This concept focuses on the poor in a society and states that their income should not fall below a certain minimum level (Streeten, 1980)
- **The Difference Principle**. This principle is based on the ethics proposed by Rawls (1971), and states that the preferred distribution is the one that maximises the welfare of the worst off.
- **The Accountability Principle**. This principle states that inequality is acceptable provided that everyone had equal opportunity at the initial allocation and that differences in income are a consequence of differences in effort (Konow, 1996).
- **The 'Just Desserts' Concept**. This concept states that remedies for injustice should be proportionate to the weight of the injustice, and not cause secondary inequity (Konow, 2001).

Note that these principles may, to some extent, be incompatible, e.g. the Difference and the Accountability Principles. Analysing equity according to the principles above requires that utility can be compared in an ordinal framework. In macro-economics, equity is often measured based on income distribution, and expressed in for instance the Gini Index (see Section 3.4.3).

When the concept of equity is applied to ecosystem management, it is, however, not only the relative income levels that determine the fairness of different ecosystem management options. In addition, stakeholders may also have different (formal or informal) access rights and/or a different degree of dependency on the ecosystem to sustain their livelihood. They also may have a different history with regards to using a resource, as in the case of traditional forest dwellers versus outside logging companies. Hence, there is large variation in the management history, and the cultural and institutional setting among ecosystems. Consequently, while considering the general principles for equity described above, criteria to judge the equity of ecosystem management options need to be specified for each ecosystem.

Representation

Effective ecosystem management is fully dependent on the collaboration between stakeholders managing the resource. Changes in the ecosystem are a function of the aggregated impact of management decisions of different stakeholders, and the management strategy of one stakeholder or group of stakeholders often has a direct impact on the supply of ecosystem services to other stakeholders. Top-down approaches enforcing ecosystem management approaches to local stakeholders have often had limited effectiveness, in particular, if local stakeholders have few alternative options to generate income, if it required changes to traditional use patterns, and where enforcement was difficult (e.g. due to remoteness, lack of monitoring, etc.).

Hence, designing and implementing ecosystem management strategies generally requires stakeholder consultation and participation – while recognising that stakeholder participation also has a number of limitations. The use of stakeholder meetings to shape environmental policies bears the risk that environmental policies primarily reflect the interests of those groups represented in the stakeholder meetings, and not the interests of the general public and future generations (e.g. Soma and Vatn, 2009). Hence, basing ecosystem management strategies on stakeholder participation is no guarantee that economic efficient, sustainable or equitable ecosystem management will be achieved.

Nevertheless, clearly, without support from local stakeholders, ecosystem management strategies may be difficult to enforce, and stakeholder participation is a crucial element in formulating objectives and approaches for ecosystem management. A large body of literature covers the topic of stakeholder representation in environmental management (e.g. Renn et al., 1993; Creighton et al., 1998; Van den Hove, 2000), and this topic is not further covered in this book. Note that, in principle, stakeholder participation and analytical approaches aiming to assess the efficiency, sustainability and equity implications of ecosystem management options are complementary. Information on the implications of ecosystem management options for different stakeholders including the general public and future generations can support the participation process, and stakeholder participation is required to identify ecosystem management options that are practically feasible.

2.1.5 Discounting

The application of efficiency and strong sustainability criteria often leads to diverging views on the ecosystem management approach to be followed (e.g. Opschoor and Van der Ploeg, 1990; Atkinson and Pearce, 1993). For instance, it may be efficient to immediately harvest all stands of timber in a forest, even if this would be unsustainable and cause irreversible loss of ecosystem services for future generations. Often, degradation of the environment involves short-term benefits (e.g. clear-cut of the timber stands), whereas sustainable management leads to a more long-term flow of benefits (e.g. through a sustainable harvesting regime). Hence, an important aspect in analysing the economic efficiency and sustainability of ecosystem management is the discount rate used to compare present and future flows of benefits derived from the ecosystem. The social discount rate represents the time preference for society as a whole (as opposed to the private discount rate). The social discount rate can be derived on the basis of the consumption discount rate (Pearce and Turner, 1990).

The consumption discount rate (r) depends upon three factors, the elasticity of marginal consumption (η), the growth rate of per capita consumption (c), and the utility discount rate (ρ), according to the following equation (Lind, 1982):

$$r = \eta \cdot c + \rho \tag{2.1}$$

The first part of the equation indicates that one unit of benefit may provide less utility in the future because society is likely to experience a growth in overall income and consumption levels ($c > 0$), and because of a decreasing marginal utility of consumption ($\eta > 0$), i.e. when society becomes richer in the future, an additional unit of consumption provides less utility. The growth in income and consumption levels can be derived from statistics (e.g. Cline, 1992), although these may be difficult to obtain or (partly) lacking where they concern the consumption of non-market benefits. The decreasing marginal utility of consumption (η) has been examined by, among others, Arrow et al. (1996), who state that a plausible value for η is in the order of 1 to 2. The utility discount rate ρ expresses that society has a positive time preference for consumption; there is a preference for immediate rather than future consumption.

The discount rate to be used in environmental cost–benefit analysis is still subject to debate (e.g. Howarth and Norgaard, 1993; Khanna and Chapman, 1996; Hanley, 1999). For instance, Freeman (1993) indicates that the social discount rate, based upon the after-tax, real interest rate, should be in the order of 2 to 3% provided that the streams of benefits and costs accrue to the same generation. Weitzman (2007) suggests that plausible values for each of the variables 'elasticity of marginal consumption' (η), 'the growth rate of per capita consumption' (c), and the 'utility discount rate' (ρ) are around 2, yielding a social discount rate of 6%. The Stern review of the economics of climate change made a case for using a much lower utility discount rate (ρ) of only 0.1, arguing that there are no moral grounds for preferring consumption by the current generation over consumption by future generations (see Stern, 2008). The Stern review consequently used a social discount rate of 1.3%. Dasgupta (2006) and Nordhaus (2007) disputed the assumptions underlying the Stern review and proposed that the social discount rate should be in the order of 4.5%. Hence, there is no consensus on the social discount rate to be used, with plausible values ranging from 1.3 to 6%. Critical elements in assuming a value for the social discount rates are the utility discount rate (in other words, the weight attributed to consumption by future versus present generations) and the assumed increase in consumption rates over time.

A point that requires further analysis is the extent to which different countries will experience a growth in overall income and consumption levels in the future (i.e. if 'c' remains positive). Climate change, natural resource depletion and potential other factors may impact the amount of natural capital available to society. For instance, Talberth et al. (2007) studied the Genuine Progress Indicator (GPI) in the US. The GPI uses the same personal consumption data as GDP but makes deductions to account for income inequality, costs of crime, environmental degradation and loss of leisure. It makes additions to account for the services from consumer durables, public infrastructure and volunteering and housework. Compared to GDP, the GPI therefore presents a potentially more accurate representation of changes in a society's overall consumption level, accounting for environmental change and a range of other factors. For the US, the GPI was found to be generally positive for the period prior to 1980, and to fluctuate without a clear long-term trend in a range of around +2% and −2% per year for the period 1980–2002. For the various methodological issues and pitfalls related to correcting GDP figures for environmental degradation and other aspects, see, for instance, Hamilton (2000) and Boyd (2007).

Note also that rates of 1.3 to 6% are relatively low compared to the rates often used in cost–benefit analysis of public and private sector investment projects (Tietenberg, 2000). Still, they lead to rapid depreciation of future costs and benefits; at a discount rate of 2%, the value of 1 euro in 100 years amounts to not more than 14 cents. Hence, through discounting, even with a low discount rate, a much larger weight is attached to the net benefits accruing to current generations as compared to the benefits for future generations. Often, the use of a high discount rate will favour ecosystem management options that lead to relatively fast depletion of resources, whereas a low discount rate will stress the economic benefits of more sustainable management options (Pearce and Turner, 1990; Tietenberg, 2000). Besides using a (very) low discount rate, a potential option to increase the importance of long-term impacts in discounting is the use of discount rates that decrease over time (Cropper and Laibson, 1999). However, in this case a problem arises when policy makers decide to design a new policy several years onwards, which would require a new discount rate to be selected at that point in time (Solow, 1999).

The question arises of whether discounting in an environmental cost–benefit analysis setting is appropriate for long-term impacts involving the maintenance of the life-support function of the planet at all. For instance, Nordhaus (1999) suggests that for long-term issues involving environmental damage such as climate change, discounting and CBA is a poor substitute for policies that focus directly on long-term objectives (such as, for instance, the stabilisation of greenhouse gas concentrations). Further information on potential long-term objectives for environmental management, at the scale of the planet, can be found in, for instance, Rockström et al. (2009).

2.1.6 Market Failures

Under a range of conditions, markets realise efficiency in the allocation of goods and services, including the allocation of goods as input in the production process and as final goods or services to consumers, given a certain initial distribution of property rights. These conditions include: (1) markets exist for all goods and services; (2) all goods are private goods; (3) all markets are perfectly competitive; (4) all buyers and sellers have perfect information; and (5) property rights are fully assigned. According to neoclassical economics, perfect markets would also be able to generate intertemporal efficiency, provided that these conditions are satisfied at all times now and in the future. The latter requires, for instance, the presence of future markets for all goods and services with full information on the characteristics of the goods and services in relation to other goods and services for both buyers and sellers. In principle, such a perfect market would ensure that in a world with decreasing natural resources, the marginal value and hence the price of the natural resources would increase up to the point where an equilibrium would be found where the value of natural resources would be high enough to provide an incentive to manage them sustainably.

Unfortunately, markets for natural resources and ecosystem services are far from perfect. Because of market failures, markets are generally not able to evoke an efficient management of ecosystems. Table 2.1 lists a selection of the market failures that are most relevant for ecosystem management. A key issue that often occurs in the case of ecosystems is that economic benefits of ecosystems are public goods, they do not accrue to the owner or manager of the ecosystem. For instance, carbon sequestration, the conservation of biodiversity and the regulation of watersheds are ecosystem services that can be of high economic importance, but they often do not lead to any income to the local ecosystem manager – unless appropriate payment vehicles such as Payment for Ecosystem Services schemes are put in place (see Section 2.4.4).

2.2 ECOSYSTEM SERVICES

2.2.1 Ecosystems, Functions and Services

The UN Convention on Biological Diversity has provided the following definition of an ecosystem: 'A dynamic complex of plant, animal and micro-organism communities and non-living environment interacting as a functional unit' (UN, 1992). Following this concept, ecosystems may lack clearly defined boundaries. However, analysis of ecosystem services, as well as ecosystem modelling, requires that the object of the analysis is clearly defined. Therefore, this book applies a spatially explicit definition of ecosystems: 'the individuals, species and populations in a spatially defined area, the interactions among them, and those between the organisms and the abiotic environment' (Likens, 1992). This implies that ecosystems may contain different sub-ecosystems within the spatially defined system to be studied. The interpretation of ecosystems in this book, as in the Millennium Ecosystem Assessment (2003), also entails agricultural and semi-natural systems such as cropland, heathlands, etc.

In the early 1970s, the concept of ecosystem *function* was proposed to

Table 2.1 Market failures of particular relevance to ecosystem management

Market inefficiency	**General description**
The public goods character of many ecosystem services	The provision of public ecosystem services, in a pure market economy, is constrained by the free-rider effect; individuals are unwilling to pay for a public service as they will also receive the service when it is fully paid for by others. Consequently, the supply of the service is below its social optimal level of provision.
A lack of property rights	Property rights include the rights, privileges and limitations to the use of a resource; a lack of property rights reduces the incentives for sustainable resource use as there is no guarantee to whom the long-term benefits of the ecosystem accrue.
Externalities	Environmental externalities occur when the use of environmental resources by one agent affects the utility or production possibilities of another agent in an unintended way. Externalities can be either positive or negative depending upon the impacts on other agents.
Discrepancies between private and social discount rates	Efficient resource allocation requires that individuals and firms use the same discount rate as appropriate for society at large. Because individuals and firms may be uncertain regarding future government policies, the private discount rate often exceeds the social discount rate.
Imperfect information	The attainment of efficient outcomes through unregulated market behaviour supposes that all actors have full information on the direct and external impacts of their transactions. In the case of ecosystem management, such perfect information is not always available. A lack of information may be related to insufficient understanding of the (complex) dynamics of the ecosystem, or the economic value generated by the ecosystem.

Source: Mäler (1985); Tietenberg (2000).

facilitate the analysis of the benefits that ecosystems provide to society (Bouma and Van der Ploeg, 1975; Hueting, 1980). An ecosystem function is defined as 'the capacity of the ecosystem to provide goods and services that satisfy human needs, directly or indirectly' (De Groot 1992). Ecosystem functions depend upon the state and the functioning of the ecosystem.

For instance, the function 'production of firewood' is based on a range of ecological processes involving the growth of plants and trees that use solar energy to convert water, plant nutrients and CO_2 to biomass.

A function may result in the supply of *ecosystem services*, depending on the demand for the good or service involved. Ecosystem services have been defined as 'the benefits provided by ecosystems' (Millennium Ecosystem Assessment, 2003) and include both the economic goods and services provided by the ecosystem to society (Costanza et al., 1997). For example, the amount of firewood extracted from an ecosystem depends on the demand from the local community and the costs at which firewood can be obtained. The supply of ecosystem services will often be variable over time, and both actual and potential future supplies of services should be included in the valuation (Drepper and Månsson, 1993; Barbier, 2000; Mäler, 2000).

Functions and services do not necessarily correspond one to one, i.e. a function may contribute to the supply of different services or a service may depend on different functions (e.g. Ansink et al., 2008). For instance, the function 'capacity to supply fish' may provide two services: 'recreation' and 'supply of fish as food product', involving two different sets of stakeholders. In principle, the user has the choice of valuing services or functions; both express the benefits supplied by the natural environment to society, and both valuation approaches should in the end lead to a consistent value indication. The main difference is that valuation of services is based on valuation of the flow of benefits, and valuation of functions is based on the environment's capacity to supply benefits. The first approach, i.e. using ecosystem services, expresses clearly the current benefits received, but additional analyses are required if the flow of ecosystem services is likely to change in the short or medium term (e.g. if current extraction rates are above the regenerative capacity of the ecosystem).

Functions better indicate the value that can be extracted in the long term, and their value is not biased by temporary overexploitation. However, it is often much more difficult to assess the capacity to supply a service than to assess the supply of the service itself. For instance, for the function 'supply of fish', this requires analysis of the sustainable harvest levels of the fish stocks involved which needs to be based on a population model including such aspects as reproduction, feed availability and predation levels. Hence, most valuation studies are based on the valuation of services rather than functions.

2.2.2 Types of Ecosystem Services

In the approach taken in this book, three different categories of ecosystem services are distinguished: (1) provisioning services; (2) regulating

Table 2.2 List of ecosystem services

Category	Examples of goods and services provided
Provisioning services	– Food – Fodder (including grass from pastures) – Fuel (including wood and dung) – Timber, fibres and other raw materials – Biochemical and medicinal resources – Genetic resources – Ornamentals
Regulating services	– Carbon sequestration – Climate regulation through control of albedo, temperature and rainfall patterns – Hydrological service: regulation of the timing and volume of river flows – Protection against floods by coastal or riparian systems – Control of erosion and sedimentation – Nursery service: regulation of species reproduction – Breakdown of excess nutrients and pollution – Pollination – Regulation of pests and pathogens – Protection against storms – Protection against noise and dust – Biological nitrogen fixation (BNF)
Cultural services	– Biodiversity conservation service (habitat service): provision of a habitat for wild plant and animal species – Provision of cultural, historical and religious heritage (e.g. a historical landscape or a sacred forests) – Scientific and educational information – Opportunities for recreation and tourism – Amenity service: provision of attractive housing and living conditions

Source: Van der Maarel and Dauvellier (1978); Ehrlich and Ehrlich (1981); Costanza et al. (1997); De Groot et al. (2002); Millenium Ecosystem Assessment (2003).

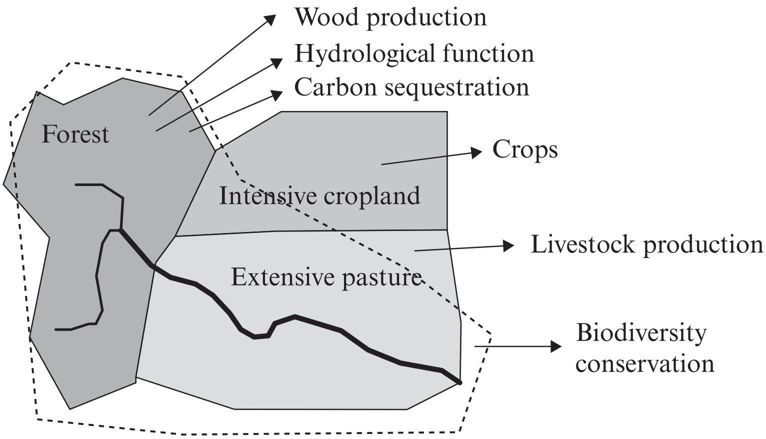
services; and (3) cultural services, based upon the Millennium Ecosystem Assessment (2003). These categories are described below, and Table 2.2 presents an overview of the ecosystem services in each category.

1. *Provisioning services* reflect goods and services *produced* by or in the ecosystem, for example, a piece of fruit or a plant with pharmaceutical

properties. The goods and services may be provided by natural, semi-natural or agricultural systems and, in the calculation of the value of the service, the relevant production and harvest costs have to be considered.

2. *Regulating services* result from the capacity of ecosystems to regulate climate, hydrological and biochemical cycles, earth surface processes and a variety of biological processes. These services often have an important spatial aspect. For instance, the flood control service of an upper watershed forest is only relevant in the flood zone downstream of the forest. The nursery service is classified as a regulation service. It reflects that some ecosystems provide a particularly suitable location for reproduction and involves a regulating impact of an ecosystem on the populations of other ecosystems.
3. *Cultural services* relate to the non-material benefits people obtain from ecosystems through recreation, cognitive development, relaxation and spiritual reflection. This may involve actual visits to the area, indirectly enjoying the ecosystem (e.g. through nature movies) or gaining satisfaction from the knowledge that an ecosystem containing important biodiversity or cultural monuments will be preserved. The latter may occur without having the intention of ever visiting the area (Aldred, 1994). The category cultural services also includes the biodiversity conservation, or habitat service, that represents the benefits that people obtain from the existence of biodiversity and nature (not because biodiversity provides a number of services, but because people believe its conservation is important in itself). In this way, the list deviates from the Millennium Ecosystem Assessment (2003) where biodiversity is assumed to support the supply of other services by enhancing ecosystem functioning and resilience, but where the value of biodiversity in itself is not explicitly recognised. However, the Millennium Ecosystem Assessment classification does no justice to the importance of protecting biodiversity in natural parks without any view on using biodiversity, as in strict nature reserves (IUCN category 1a). Because the importance attached to biodiversity is strongly dependent on the cultural background of the observer, the service is classified as a cultural service (Hein et al., 2006).

The Millennium Ecosystem Assessment (2003) also distinguishes the category ‘supporting services’. Supporting services represent the ecological processes that underlie the functioning of the ecosystem. However, their inclusion in valuation may lead to double counting as their value is reflected in the other three types of services. In addition, there are a very large number of ecological processes that underlie the functioning of



Notes: The arrows indicate the different ecosystem services provided by land cover units. Biodiversity partly depends on the combination of forest, pasture and cropland habitats, as indicated by the dotted line (see also text above).

Figure 2.1 Land cover and ecosystem services

ecosystems, and it is unclear on which basis supporting services should be included in, or excluded from, a valuation study. Therefore, considering the focus of the book on applying ecosystem services analysis and valuation for environmental management, this category of services is not further considered here (cf. Hein et al., 2006).

Clearly, there is a strong relation between land cover and ecosystem services supply. Land cover units will typically supply a specific mix of ecosystem services, see for instance Figure 2.1. Some services are confined to a specific land use unit, as in the case of wood production being confined to forest lands. In other cases, ecosystem services depend on the combination and spatial pattern of land cover units. For instance, the borders between forests and grasslands are typically rich in biodiversity, and an open landscape with small-scale agricultural activities and patches of forest may be attractive for tourism. In other cases, a land cover unit may exercise a negative influence over the supply of ecosystem services in another unit, as in the case of a highway reducing the potential for tourism in nearby areas (see e.g. Willemen et al., 2008).

2.2.3 Quantifying Ecosystem Services

Before ecosystem services can be valued, they need to be quantified in biophysical terms. The techniques required to analyse services in biophysical terms depend entirely on the services that have been selected for the assessment. It should be noted that, particularly for regulating services, the quantification of the service is often at least as time and data consuming as the subsequent economic analysis. In addition, every service, in every economic, environmental and social context will require a specific approach with respect to the data and required approach for analysis. In the sections below, guidance is provided on approaches that can be taken for each service category.

Provisioning services

For provisioning services, surveys can reveal the flows of products harvested from the ecosystem, for instance, expressed as kilograms of fruits or tons of timber harvested per time unit. A provisioning service may be supplied in terms of an annual or seasonal flow, or in terms of a one-off harvest. In order to analyse the value of the provisioning service, information is also needed on the efforts required to extract the products from the ecosystem. In the case of harvesting in natural forests, this relates to labour and possibly the tools or equipment required for harvesting. In the case where the product is obtained from cultivated agricultural land, valuation should consider the various inputs required in the agricultural production process. Besides labour and equipment, this also includes, for example, land, fertilisers, pesticides, seeds, etc.

Regulating services

In the case of regulating services, it is important to consider the precise nature of the service supplied as well as its spatial and temporal dimensions. Table 2.3 provides a list of potential indicators that can be used to measure regulating services. The precise indicators will depend on the objective and scale of the assessment as well as the availability of data. For instance, the hydrological service can be expressed as both a reduction in peak flows, and an increase in low season flow depending on the area under consideration (flood risk versus risks of seasonal water shortages).

The supply of a regulation service may be variable in space. For example, the hydrological service has a distinct spatial component, because flood risks will decrease with increasing distances from the water course, as a function of the topography of the valley. Spatially explicit analysis of ecosystem service supply normally requires GIS (for examples see Geoghegan et al., 1997, Voinov et al., 1999; Willemen et al., 2008). In a GIS, initial conditions, processes and implications of decision variables need to be specified for each spatial unit distinguished and data requirements are generally high.

In addition, temporal scales need to be considered. Ecosystem service

Table 2.3 Biophysical assessment methods for regulating services

Regulating services	Assessment method
Carbon sequestration	Modelling of carbon flows in the ecosystem
Climate regulation through regulation of albedo, temperature and rainfall patterns	Regional climate models
Regulation of the timing and volume of river and groundwater flows	GIS models including run-off and river flow as a function of, among others, plant cover, soil properties and land management
Protection against floods by coastal or riparian ecosystems	Modelling of flood risks with different vegetation cover; alternatively, comparison of impacts of past floods in protected and non-protected areas.
Regulation of erosion and sedimentation	Erosion model following USLE or other models to determine erosion rates. Analysis of sedimentation rates requires catchment models of run-off and erosion, transport and deposition of sediment particles.
Regulation of species reproduction (nursery service)	Model of species reproduction, based on juveniles per successful breeding or spawning effort and the factors determining the success of reproduction (e.g. water quality, vegetation cover, etc.)
Breakdown of excess nutrients and pollution	Dependent on denitrifcation rates and phosphate absorbtion rates, which vary as a function of retention time, oxygen level, iron concentrations, temperature, etc.
Pollination (for most plants)	Pollination rates for agricultural crops can be found in the literature (e.g. Klein et al., 2007), for non-cultivated species data is much scarcer.
Regulation of pests and pathogens	Information availability strongly dependent on the pests or pathogen involved, for some pests literature is available indicating the factors determining the chance and severity of outbreaks.
Protection against storms	Simple models can be used to calculate the reduction in wind speed as a function of e.g. tree cover and surface roughness. These can be translated into the wind's capacity to detach and transport particles.
Protection against noise and dust	Literature is available in order to make rough estimates of the impacts of vegetation belts on dust and air quality.

Table 2.3 (continued)

Regulating services	**Assessment method**
Control of run-off	Infiltration rates under different types of plant cover and land management need to be spatially modelled to retrieve impacts of vegetation on run-off (e.g. Luijten et al., 2000, see also Bosch and Hewitt, 1982, for an overview).

supply will, in many cases, vary over time depending on fluctuations in the ecosystem (e.g. as a function of rainfall) as well as human management. For instance, the service 'carbon sequestration' depends on the building up of carbon in either above-ground biomass or as soil organic matter. The actual uptake depends on the growth of the plants minus the decomposition of organic material in the soils. This uptake tends to decrease as newly planted forests or plantations develop into mature forest stands. Total carbon sequestration and the time to reach maturity depend on the type of forest and climatic conditions involved.

Cultural services

The perceived benefits from cultural services strongly depend on the cultural backgrounds of the people that receive the service. It is a function of religious, moral, ethical and aesthetical motives, and these motives vary substantially between different societies. Ranging from indigenous to industrial societies, there are striking differences in the way cultural and amenity services are perceived, experienced and valued by different cultures. In order to quantify the service, it is both the type of interaction and the numbers of people involved that are relevant indicators. The type of interaction ranges from frequent or occasional visits to more passive types of benefiting from the presence of a certain ecosystem, e.g. from simply knowing that the ecosystem is maintained and preserved. Prior to valuation of the service, both the type of interactions and the amount of people involved need to be analysed.

Biodiversity conservation service

In the last decades, a large number of methods to quantify biodiversity and other ecological values have been developed. Wathern et al. (1986) mentioned that over 100 of these techniques have been described in the literature. The most widely used criteria for ecological value relate to the species richness of the ecosystem and the rarity of the species it contains. A brief summary of several potential indicators for the biodiversity conservation service is provided below, for the two categories of species level and ecosystem level indicators.

Species level indicators

- **Number of species in specific classes.** Given the large number of species, indicators presenting the species richness of an area need to focus on (a combination of) specific taxonomic groups, such as mammals, meadow birds or vascular plants. Although the number of species in specific groups is an indicator of the species diversity of an area, drawbacks are that it does not indicate the population numbers per species (which may be below viable population numbers) and that it gives equal weighting to each species.
- **Biodiversity indices.** The most well-known of these indicators are the Simpson and Shannon Indices. They express the species diversity in an ecosystem, taking into account both species richness and the relative abundance of each species. However, the indicators are difficult to interpret, and they also provide equal weighting to each species. For more information on these two indices and how they can be applied, see, for example, Duelli and Obrist (1998).
- **Numbers of Red List and/or endemic species.** The IUCN Red List has a global cover and provides taxonomic, conservation status and distribution information on plants and animals. The number of species evaluated for the list is currently (2009) over 45 000. Certain taxonomic groups have been completely, or almost completely, assessed (mammals, birds, amphibians, freshwater crabs, warm-water reef building corals, conifers and cycads). The cover is not complete for all taxonomic groups, with remaining data deficiencies for freshwater, marine and semi-arid ecosystems. The list provides a good starting point for identifying the number of species of particular concern for nature conservation that are present in an ecosystem.
- **Populations of keystone species.** The keystone species concept stipulates the existence of a limited number of species that regulate essential ecosystem processes such as nutrient recycling, see Pain et al. (2003) for an example. Whereas keystone species may exist for some ecosystems, it is as yet unclear if keystone species can be defined for all ecosystems. The keystone species concept in relation to other theories related to the resilience of ecosystems is further discussed in Section 3.3.3. Where they can be identified, monitoring the abundance of keystone species provides an indication of the functioning of the ecosystem. In these cases, the loss of keystone species would lead to the loss of a range of other species in an ecosystem.

Ecosystem level indicators

- **Presence of species that are indicative for environmental quality.** Maintaining environmental quality is one of the preconditions for conserving biodiversity. The occurrence of a species in an ecosystem is determined by a host of factors including dispersal factors (barriers, history, etc.), disturbance factors (extreme events, human pressures) and resource factors (nutrients, food, etc.) (Guisan and Thullier, 2005). Disturbance may affect biodiversity, with those species that require specific ecological niches particularly vulnerable to environmental change. Environmental quality indicators provide information on the degree of disturbance and, hence, the sustained potential of an ecosystem as habitat for (rare and threatened) species. A well-known example is the use of aquatic macroinvertebrates that are sensitive to water pollution as an indicator for stream water quality (e.g. Heino et al., 2003).
- **Ecosystem disturbance in terms of land area affected.** Ecosystem disturbance, or its inverse: the area of preserved ecosystem remaining is a key indicator for biodiversity conservation. A physical loss of ecosystems, for instance through land use conversion, has clear impacts on its biodiversity value. However, it is often difficult to define and qualify the degree of disturbance to which ecosystems have been exposed, for example to relate deforestation to ecosystem disturbance. A number of methods have been developed, for example the Habitat Index (Hannah et al., 1995) or the Natural Capital Index, which is the product of the size of a natural area and its nature quality. The nature quality of an ecosystem is then defined as the ratio between the current state and a particular baseline state, based on a range of indicators such as the abundance of characteristic species, expressed as a percentage (Ten Brink and Tekelenburg, 2002).
- **Extent and effectiveness of protected areas.** There are currently over 100 000 protected areas worldwide, covering over 12% of the Earth's land surface (Chape et al., 2005). However, there is large variation in the effectiveness of the protected areas. For example, according to a recent report, illegal logging and/or land use conversion has taken place in no less than 37 out of 41 national parks in Indonesia (Ministry of Forestry, 2006), with satellite imagery indicating that, in the worst cases, up to half the protected area has been exposed to heavy logging and/or land use conversion (Curran et al., 2004). Hence, both the extent and effectiveness of the protected areas need to be analysed in case this indicator is used to report on biodiversity trends.

Double counting of ecosystem services

An important issue in the valuation of ecosystem services is the double counting of services (Millennium Ecosystem Assessment, 2003; Turner et al., 2003). Specifically, there is a risk of double counting in relation to the regulating services that support the supply of other services from an ecosystem. For example, consider a natural ecosystem that harbours various populations of pollinating insects. These insects pollinate both the plants inside the natural ecosystem and the fruit trees of adjacent orchards. In an analysis of the economic value of the natural area, only the pollination of the adjacent fruit trees should be included as a regulation service. As for the various trees inside the natural area, the produce from these trees (e.g. wood, rattan and fruits) should be included in the valuation (as provisioning services), but the pollination of these natural trees should not, as this would lead to double counting

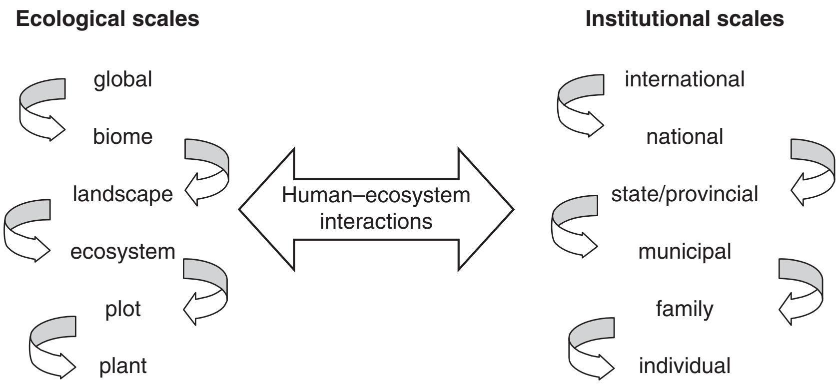
In general, regulating services should only be included in the valuation if (1) they have an impact outside the ecosystem to be valued; and/or (2) if they provide a *direct* benefit to people living in the area (i.e. not through sustaining or improving another service). The first case is illustrated by the example of the fruit trees above. An example of a service that may provide a direct benefit inside an area that is not included in other ecological services, is the service 'protection against noise and dust' provided by a green belt besides a highway. If this affects the living conditions of people living inside the study area, it needs to be included in the valuation. A prerequisite for applying this approach to the valuation of regulating services is that the ecosystem is defined in terms of its spatial boundaries – otherwise the external impacts of the regulating services can not be precisely defined.

2.3 SCALES AND STAKEHOLDERS

2.3.1 Ecological and Institutional Scales

Scales refer to the physical dimension, in space or time, of phenomena or observations (O'Neill and King, 1998). According to the original definition, ecosystems can be defined at a wide range of spatial scales (Tansley, 1935). These range from the level of a small lake up to the boreal forest ecosystem spanning several thousands of kilometres. As the scale of a particular analysis usually needs to be defined, it has become common practice to distinguish a range of spatially defined ecological scales (Holling, 1992; Levin, 1992). They vary from the level of the individual plant, via ecosystems and landscapes, to the global system, see Figure 2.2.

Ecosystem services are generated at all *ecological scales*. For instance,



Source: Adapted from Leemans (2000).

Figure 2.2 Selected ecological and institutional scales

fish may be supplied by a small pond, or may be harvested in the Pacific Ocean. Biological nitrogen fixation enhances soil fertility at the ecological scale of the plant, whereas carbon sequestration influences the climate at the global scale. The supply of a regulation service such as the hydrological service depends on a range of ecological processes that operate at the scale of the watershed.

In the socio-economic system, a hierarchy of *institutions* can be distinguished (Becker and Ostrom, 1995; O'Riordan et al., 1998). They reflect the different levels at which decisions on the utilisation of capital, labour and natural resources are taken. At the lowest institutional level, this includes individuals and households. At the higher institutional level scales can be distinguished: the communal or municipal, state or provincial, national and international level (see Figure 2.2). Many economic processes, such as income creation, trade and changes in market conditions can be more readily observed at one or more of these institutional scales.

The next section examines how ecological and institutional scales influence the supply of the three different types of ecosystem services.

2.3.2 Scales of Ecosystem Services

Provisioning services

The possibility of harvesting products (e.g. fish) from natural or semi-natural ecosystems depends upon the stocks involved. The development of these stocks is driven by harvest rates, endogenous ecological processes and, potentially, exogenous drivers resulting from environmental change.

For instance, fish stocks may develop as a function of (1) harvest rates; (2) natural, annual variations or trends in ecological processes such as predation or reproductive success; and (3) external factors such as changes in seawater temperature or currents resulting from climate change. Analysing the supply of provisioning services, and the efficiency, sustainability and equity aspects of resource management options, requires understanding responses to management at the scale of the ecosystem (e.g. the lake or the North Atlantic ocean).

However, the benefits of the supply of a provisioning service may accumulate to stakeholders at a range of institutional scales. Local residents are often an important actor in the harvest of the resources involved, unless they do not have an interest in, or access to, the resource (e.g. due to a lack of technology, or because the ownership or user-right of the resource resides with other stakeholders). In addition, there may be stakeholders' interests at larger scales if the goods involved are harvested, processed or consumed at larger scales. For example, a marine ecosystem may be fished both by local fishermen and an international fleet, and fish from the same ecosystem may be consumed in markets in different continents.

Regulating services

A regulation service is interpreted as an ecological process that supports the supply of one or more provisioning or cultural services, or provides a direct benefit to people (see Section 2.2.3). Because the ecological processes involved take place at specific ecological scales, it is often possible to define the ecological scale at which the regulation service is generated (see Table 2.4). For many regulating services, not only the scale but also the position in the landscape plays a role – for example, the impact of the water buffering capacity of forests will be noticed only downstream in the same catchment (Bosch and Hewitt, 1982). Stakeholders in a regulation service are all people residing in or otherwise depending upon the area affected by the service, and the scale at which stakeholders can be identified varies depending on the specific service involved.

Cultural services

Cultural services may also be supplied by ecosystems at different ecological scales, such as a monumental tree or a natural park. Stakeholders in cultural services can vary from the individual to the global scale. For local residents, an important cultural service is commonly the enhancement of the aesthetic, cultural, natural and recreational quality of their living environment. In addition, particularly for indigenous people, ecosystems may also be a place of rituals and a point of reference in cultural narratives (Posey, 1999; Infield, 2001). Nature tourism has become a major cultural service in

Table 2.4 Most relevant ecological scales for analysing regulating services

Ecological scale	**Dimensions**	**Regulating services**
Global	> 1 000 000 km^2	Carbon sequestration Climate regulation through regulation of albedo, temperature and rainfall patterns
Biome – landscape	10 000–1 000 000 km^2	Regulation of the timing and volume of river and ground water flows Protection against floods by coastal or riparian ecosystems Control of high sediment loads in rivers and sedimentation Regulation of species reproduction (nursery service)
Ecosystem	1–10 000 km^2	Breakdown of excess nutrients and pollution Insect pollination (for most plants) Regulation of pests and pathogens Protection against storms Protection against noise and dust
Plot – plant	< 1 km^2	Control of run-off and associated loss of soil nutrients Biological nitrogen fixation (BNF)

Notes: Some services may be relevant at more than one scale and/or depend on processes operating across a range of scales.

Western countries, and it is progressively gaining importance in developing countries as well. Because the value attached to the cultural services depends on the cultural background of the stakeholders involved, there may be very different perceptions of the value of cultural services among stakeholders at different scales. Local stakeholders may attach particular value to local heritage, cultural or amenity services, whereas national and/or global stakeholders may have a particular interest in the conservation of nature and biodiversity (e.g. Swanson, 1997; Terborgh, 1999).

The scales at which ecosystem services are generated and supplied determine the interests of the various stakeholders in the ecosystem. Services generated at a particular ecological level can be provided to stakeholders at a range of institutional scales, and stakeholders at an institutional scale can receive ecosystem services generated at a range of ecological scales. When the value of a particular ecosystem service is assessed, different indications of its value may be found depending upon the institutional level at which the analysis is performed. For example, local stakeholders may particularly value a provisioning service that may be irrelevant at the national or international level. Hence, if a valuation study is implemented with the aim of supporting decision making on ecosystems, it is crucial to consider the scales at which the ecosystem services are supplied.

2.3.3 Stakeholders

A stakeholder can be defined as any entity with a declared or conceivable interest or stake in a policy concern (Schmeer, 1999). Stakeholders can be of different form, size and capacity including individuals, organisations, or unorganised groups. In most cases, stakeholders fall into one or more of the following categories: international actors (e.g. donors), national or political actors (e.g. legislators, governors), public sector agencies, interest groups (e.g. unions, medical associations), commercial/private for-profit organisations, non-profit organisations (NGOs, foundations), civil society members and users/consumers. Government institutions are stakeholders for resources in their jurisdiction and citizens of other countries may be stakeholders when they derive welfare from the long-term indirect benefits from ecosystem services such as carbon sequestration, tourism and nature conservation.

Stakeholders have four main attributes with respect to their interests in ecosystem services: the type of resource use practiced by the stakeholders, the level of influence (power) they hold, their degree of dependency on the ecosystem services (availability of alternatives) and the group/coalition to which they belong. These attributes can be identified through various data collection methods, including interviews with country experts knowledgeable about stakeholders or with the actual stakeholders directly; see, for example, Creighton et al. (1998). It is clear that the stakeholders deriving benefits from an ecosystem may be just as diverse as the ecosystem services themselves. Nevertheless, it is crucial to consider the differences in stakeholders when analysing ecosystem services, as stakeholder interests and access rights will determine the interests and motivations of stakeholders in managing the resource and management plans need to be fine-tuned with these interests in order to obtain stakeholder collaboration at different levels.

2.4 VALUES AND VALUATION OF ECOSYSTEMS

2.4.1 Basic Introduction to the Economic Valuation of Ecosystem Services

Various schools of economic theory have provided different interpretations and definitions of the concept of value. In neo-classical economics, value is related to the price of the good or service in an open and competitive market, as a function of demand and supply. Accordingly, for traded ecosystem services, under perfect market conditions, price reflects the marginal economic value of the service. Analysing the overall economic value of the supply of an ecosystem service (or any other good) requires establishing the consumer and producer surplus, as briefly described below. For ecosystem services not traded in a market, consumer and producer surpluses may be difficult to analyse and various alternative valuation approaches have been developed, as elaborated in Section 2.4.3.

The consumer surplus

The concept of consumer surplus was first described by Dupuit and introduced to the English speaking world by Marshall (in 1920): 'The excess of price which a consumer would be willing to pay rather than go without the thing, over that what he actually pays is the economic measure of this surplus of satisfaction' (Johansson, 1999). In other words, the individual consumer surplus equals the maximum willingness-to-pay of a consumer for a good minus the price the consumer faces for that good. Estimation of the consumer surplus generally requires the construction of a demand curve, either reflecting the demand of an individual (for the individual consumer surplus) or the demand of society at large (for the aggregated consumer surplus).

Hicks (1941) found an inconsistency in the ordinary, or Marshallian, consumer surplus: an individual may change the total basket of goods and services obtained following changes in the price of a specific good or service. Consequently, Hicks developed several alternative concepts to estimate consumer surplus that account for such changes, the most well-known being the compensating variation (CV) and the equivalent variation (EV) (see e.g. Freeman, 1993, for details). Willig (1976) has shown that under two conditions the difference between EV, CV and the Marshallian consumer surplus is small: (1) if the income elasticity of demand for the good in question is low; and (2) if the consumer surplus is low in terms of percentage of income. These conditions imply that it is only correct to use the ordinary demand curve in the case of marginal changes in the supply of a good. Construction of Hicks-compensated demand functions requires analysis of the overall consumption patterns (Johanson, 1999).

The producer surplus

The producer surplus indicates the amount of net benefits a producer gains, given his production costs and the (market) price he receives for his products (Varian, 1993). In the valuation of ecosystem services, the producer surplus needs to be considered if there are costs related to 'producing' the ecosystem good or service, such as, for example, the costs related to collecting or harvesting forest products (Freeman, 1993; Hueting et al., 1998). In the case where an ecosystem services approach is used to analyse activities such as agriculture or fisheries, clearly, the full production costs of the fisherman (boat, equipment, labour, etc.) or farmer (land, machinery, inputs, labour, etc.) need to be accounted for. The estimation of the producer surplus generally requires the construction of a supply curve indicating production costs for all producers in a market. For public ecosystem services, supply curves can be seen as reflecting the costs of measures to restore and conserve the supply of services. For these services, a supply curve is often difficult to construct and the producer surplus is difficult to establish (Hueting et al., 1998).

The concepts of consumer and producer surplus are illustrated with the example of the pollination service. Insect pollination is required for a range of crops including apples, oranges, almonds, etc. (see e.g. Klein et al., 2007, for a full overview). Insect pollination can be achieved by bringing in beehives, or can be performed by naturally occurring bees or, for some crops, other animals. In the latter case, pollination is an ecosystem service, in particular, a regulating service required for agricultural production. In the valuation of pollination, it is necessary to consider the scale at which pollination is studied. For instance, in the case where the value of pollination in one particular farm is studied, there will probably be no price effects since the production of this farmer is likely to be small compared to the overall market supply. In this case, changes in the producer surplus can be estimated on the basis of multiplying physical changes in ecosystem services supply with net revenues generated per unit of ecosystem service. For example, Rickets et al. (2004) relate the value of the pollination service supplied by forest patches on a Costa Rican coffee farm (which serve as habitat for pollinating bees) to the impact of pollination on the coffee yields, the total area of coffee plants pollinated and the net benefits obtained from the sale of coffee (off-farm price minus variable production costs).

However, where pollination declines at the national scale, price effects for pollinated crops become increasingly likely, because the supply of the affected crops is reduced while demand, presumably, is not affected (Hein, 2009). Valuation of pollination services at the national scale, therefore, needs to consider that prices may not be constant. In this case, demand and supply curves have to be constructed to analyse changes in the producer and consumer surplus as a function of changes in the supply of the pollination service. This is illustrated in Figure 2.3. Figure 2.3 shows that a decline in the pollination service may reduce agricultural production, and shift the supply curve of the affected crops to the left, from S to S'.

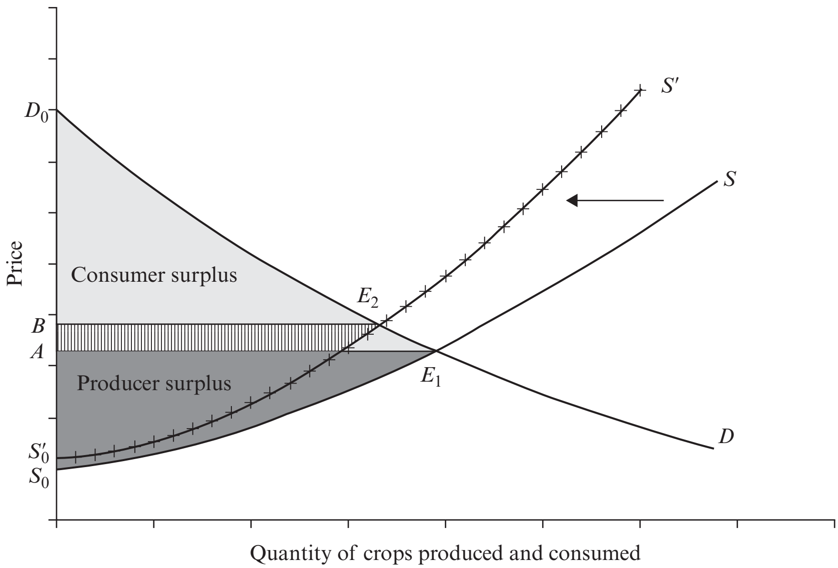


Figure 2.3 Changes in the consumer and producer surplus in a case where pollination losses affect agricultural production

This shift reflects that farmers will obtain a lower harvest at relatively higher production costs. Consequently, a new market equilibrium (E_2) is reached, at a higher food price and with a lower quantity of crops traded in the market.

Consequently, the producer surplus changes from S_0AE_1 to $S_0'BE_2$ and the consumer surplus from D_0AE_1 to D_0BE_2. From Figure 2.3, it is clear that all consumers will be affected by the decline in pollination. With regards to the producer surplus, there may also be producers that are not affected by a decline in the pollination service and that may benefit by obtaining a higher price for their crops, for example, producers in the part of the country not affected by a reduction in pollination services (if any) or producers growing substitute crops that are less dependent on pollination. Hence, the producer surplus may increase or decrease when pollination services are affected, depending on the shape of the demand and supply curves.

2.4.2 Types of Economic Value

There are several types of economic value and different authors have provided different classifications for these value types (e.g. Pearce and

Turner, 1990; Hanley and Spash, 1993; Munasinghe and Schwab, 1993; Millennium Ecosystem Assessment, 2003). In general, the following four types of value can be distinguished: (1) direct use value; (2) indirect use value; (3) option value; and (4) non-use value.

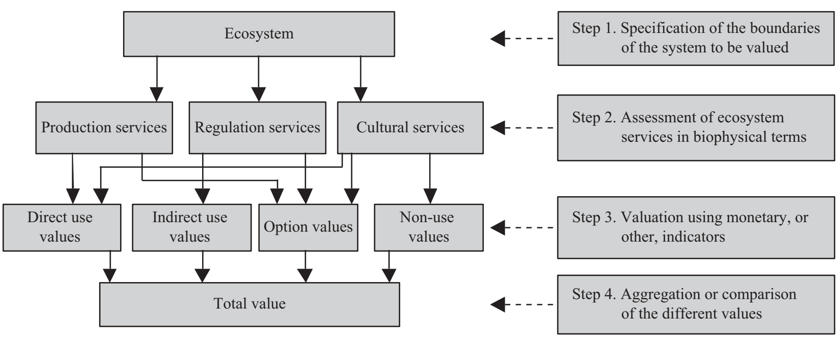
1. **Direct use value** arises from the direct utilisation of ecosystems (Pearce and Turner, 1990), for example, through the sale or consumption of a piece of fruit. All provisioning services and some cultural services (such as recreation) have direct use value.
2. **Indirect use value** stems from the indirect utilisation of ecosystems, in particular through the positive externalities that ecosystems provide (Munasinghe and Schwab, 1993). This reflects the type of benefits that regulating services provide to society.
3. **Option value** relates to risk. Because people are unsure about their future demand for a service, they are willing to pay to keep the option of using a resource in the future – insofar as they are, to some extent, risk averse (Weisbrod, 1964; Cichetti and Freeman, 1971). Option values may be attributed to all services supplied by an ecosystem. Various authors also distinguish quasi-option value (e.g. Hanley and Spash, 1993), which represents the value of avoiding irreversible decisions until new information reveals whether certain ecosystems have values we are not currently aware of (Weikard, 2003). Although theoretically well established, the quasi-option value is in practice very difficult to assess (Turner et al., 2000).
4. **Non-use value** is derived from attributes inherent to the ecosystem itself (Cummings and Harrison, 1995; Van Koppen, 2000). Hargrove (1989) has pointed out that non-use values can be anthropocentric, as in the case of natural beauty, as well as ecocentric, based upon the notion that animal and plant species have a certain 'right to exist'. Kolstad (2000) distinguishes three types of non-use value: existence value (based on utility derived from knowing that something exists); altruistic value (based on utility derived from knowing that somebody else benefits); and bequest value (based on utility gained from future improvements in the well-being of one's descendants). The different categories of non-use value are often difficult to separate, both conceptually (Weikard, 2002) and empirically (Kolstad, 2000). Nevertheless, it is important to recognise that there are different motives to attach non-use value to an ecosystem service, and that these motives depend upon the moral, aesthetic and other cultural perspectives of the stakeholders involved.

In principle, the four value types: direct use, indirect use, option and non-use value are exclusive and may be added. The sum of the direct use, indirect use and option values equals the total use value of the system; the sum of the use value and the non-use value has been labelled the 'total economic value' of the ecosystem (Pearce and Turner, 1990). If all values are expressed as a monetary value, and if the values are expressed through commensurable indicators (e.g. consumer and/or producer surplus), the values can be summed.

2.4.3 Valuation Approaches and Techniques

Figure 2.4 presents a basic framework for analysing the economic value of ecosystem services. The framework involves four subsequent steps: (1) definition of the spatial and temporal boundaries of the (eco)system and identification of the services to be studied; (2) quantification of ecosystem services in biophysical terms; (3) valuation of ecosystem services; and (4) aggregation or comparison of values of different services. The services to be in or excluded from the assessment are determined by the objectives and system boundaries of the assessment. The framework is static, i.e. it allows an analysis of the services and values supplied by an ecosystem under current land use and management; a dynamic framework is presented in Chapter 3.

In the last three decades, a range of economic valuation methods for ecosystem services has been developed. They differ for private and public goods, as described below.



Source: Hein et al. (2006).

Figure 2.4 Schematic overview of an ecosystem services valuation approach

Valuation of private goods

In the case of private goods or services traded in the market, price is the measure of marginal willingness to pay and it can be used to derive an estimate of the economic value of an ecosystem service (Hufschmidt et al., 1983; Freeman, 1993). The appropriate demand curve for the service can – in principle – always be constructed. However, in practice this is often difficult, as (1) it is not always known how people will respond to large increases or decreases in the price of the good, and (2) it may be difficult to assess when consumers will start looking for substitute goods or services. In the case of price distortions, for example because of subsidies, taxes, etc., an economic (shadow) price of the good or service in question needs to be constructed. In some cases, this can be done on the basis of the world market prices (Little and Mirrlees, 1974; Little and Scott, 1976). In the case where the private good is not traded in the market, for example, because it is used for self-consumption, shadow prices need to be constructed on the basis of: (1) the costs of substitutes; or (2) the derived benefit of the good (Munasinghe and Schwab, 1993).

Valuation of public goods

For public goods or services, the marginal willingness to pay can not be estimated from the direct observation of transactions and the demand curves are usually difficult to construct (Hueting, 1980). Two types of approaches have been developed to obtain information about the value of public ecosystem services: the revealed and stated preference approach (Pearce and Howarth, 2000). Pearce and Turner (1990) called them indirect and direct preference methods, respectively.

The *revealed preference* approaches use a link with a marketed good or service to indicate the willingness-to-pay for the service. There are two main types of revealed preference approaches:

- **Physical linkages.** Estimates of the values of ecosystem services are obtained by determining a physical relationship between the service and something that can be measured in the market. The main approach in this category is the damage–function (or dose–response) approach, in which the damages resulting from the reduced availability of an ecosystem service are used as an indication of the value of the service (Johanson, 1999). This method can be applied to value, for instance, the hydrological service of an ecosystem.
- **Behavioural linkages.** In this case, the value of an ecosystem service is derived from linking the service to human behaviour – in particular, people's expenditures to offset the lack of a service, or to obtain a service. An example of a behavioural method is the Averting

Behaviour Method (ABM). There are various kinds of averting behaviour: (1) defensive expenditure (a water filter); (2) the purchase of environmental surrogates (bottled water); and (3) relocation (OECD, 1995). The travel cost method is another example of an indirect approach using behavioural linkages (Van Kooten and Bulte, 2000).

With *stated preference* approaches, various types of questionnaires are used to reveal the willingness-to-pay of consumers for a certain ecosystem service. The most important approaches are the Contingent Valuation Method (CVM) and related methods. In the last decades, CVM studies have been widely applied (see Nunes and van den Bergh, 2001 for an overview). It is the only valuation method that can be used to quantify the non-use values of an ecosystem in monetary terms. Information collected with well-designed CVMs has been found suitable for use in legal cases in the US – as in the case of the determination of the amount of compensation to be paid after the Exxon Valdez oil spills (Arrow et al., 1993). Nevertheless, various authors question their validity and reliability – both on theoretical and empirical grounds. There are two main points of criticism against CVM. First, CV estimates are sensitive to the order in which goods are valued; the sum of the values obtained for the individual components of an ecosystem is often much higher than the stated willingness-to-pay for the ecosystem as a whole. Second, CV often appears to overestimate economic values because respondents do not actually have to pay the amount they express they are willing to pay for a service (see Diamond and Hausman, 1994; Cummings and Harrison, 1995; Hanemann, 1995; Carson, 1998).

In response to the difficulties encountered in quantifying the non-use values of ecosystems in monetary terms, some authors have proposed quantifying this value in *ecological* terms only. For ecological quantification, a range of indicators is available, such as the species richness of the ecosystem, and the rarity of the species it contains (see Section 2.2.3). Accordingly, other indicators may be used for health impacts and cultural services of ecosystems.

If non-monetary indicators are used for the non-use values, the values can be presented side-by-side, which means it is left to the reader to compare the two value types (see Strijker et al., 2000, for an example). Alternatively, they can be compared using Multi-Criteria Assessment (MCA). In MCA, stakeholders are asked to assign relative weights to different sets of indicators (non-monetary as well as monetary), enabling comparison of the indicators (e.g. Nijkamp and Spronk, 1979). A whole set of different MCA techniques have been developed in recent decades (see e.g. Hajkowicz and Collins, 2007 for an overview).

Different stakeholder groups can be expected to have different perspectives on the importance of the different types of value (Vermeulen and Koziell, 2002). Through group valuation, citizens' juries or the use of deliberative processes, stakeholders can be encouraged to converge on a representative assessment of the values of different ecosystem services. For more information on these alternative approaches to examining the benefits provided by ecosystem see, for instance, O'Neill (2001).

An overview of the different valuation methods, and the value types they can be used for, is presented in Table 2.5. Further details on the various ecosystem valuation techniques are provided in Dixon and Hufschmidt (1986), Pearce and Turner (1990), Hanley and Spash (1993) and Pearce and Moran (1994). Costanza et al. (1997) and Pearce and Pearce (2001) provide an overview of the values of a range of ecosystem services in selected ecosystems. If few data are available for an ecosystem, crude estimates of the values of ecosystem services may be obtained through 'benefit transfer' – the transfer of ecosystem values to settings other than those originally studied (Green et al., 1994; Willis and Garrod, 1995; Brouwer et al., 1997). However, the values provided by ecosystems are often strongly dependent on the biophysical, economic and institutional context, and benefit transfer is prone to a high degrees of uncertainty.

A number of general recommendations can be provided with regards to ecosystem services valuation. First, typically, a significant part of the effort required for the valuation of ecosystem services is related to quantifying services in biophysical terms. This holds, in particular, for regulating services.

Second, while it is common practice to base a valuation study on ecosystem services rather than functions, care needs to be taken that assumptions underlying future flows of services are realistic, i.e. potential overharvesting leading to decreasing supply in the future needs to be accounted for.

Third, double counting of services needs to be avoided, by defining the system boundaries of the assessment and identifying provisioning and regulating services in such a way that overlap is avoided (see also Section 2.2.3).

Fourth, it remains particularly cumbersome to value the biodiversity conservation service, for which no meaningful monetary indicators may be applicable, and even biophysical indicators may be difficult to define. Pending new breakthroughs in terms of defining and quantifying biodiversity, a small set of indicators for the biodiversity conservation service may need to be fine-tuned to each ecosystem to be assessed.

Table 2.5 Valuation methods and the value types for which they are typically applied

Valuation method	**Suitable for**	**Typically used to analyse**			
		direct use values	indirect use values	option values[1]	non-use values
Stated preference methods					
(a) Market valuation	Ecosystem goods and services traded on the market	x	x		
(b) CVM	The use of CVM is limited to goods and services that are easily to comprehend for respondents – excluding most regulating services	x		x	x
Revealed preference methods					
(a) Hedonic pricing	Applicable where environmental amenities are reflected in the prices of specific goods, in particular property.	x			
(b) Travel cost method	Can be used to value the recreation service.	x			
(c) Averting Behaviour Method	Mostly applicable to regulating services, for instance, the water purification service.		x		
(d) Damage function approach	Applicable where loss of ecosystem services will cause economic damage, e.g. through an increased flood risk.		x		
Ecological valuation methods					
a) Ecological valuation	Only for the non-use value of the biodiversity conservation service				x

Notes: [1] Analysing the option value requires the specification of the risk averseness of the involved stakeholders (e.g. Wik et al., 2004).

Source: Pearce and Turner (1990); Hanley and Spash (1993); Munasinghe (1993); Cummings and Harrison (1995).

Fifth, valuation studies should specifically consider the spatial and temporal scales relevant to the study. The spatial scale of the assessment determines which stakeholders are included or excluded from the assessment. For example, the decline of herring stocks in the Dutch North Sea fishing grounds negatively affected Dutch fishermen, but had a positive impact on the profits of Danish herring fishermen (Lindebo, 2004; Simmonds, 2007). Hence, the calculated costs of ecosystem degradation depends on the impacts being analysed for Dutch fishermen, for Danish fishermen or at the European scale. With regards to the temporal scale, the value of ecosystem services can be expected to vary over time, for instance because markets will change or because technologies capable of producing substitute goods may improve.

2.4.4 Developing Markets for Ecosystem Services

In recent years, Payment for Ecosystem Services (PES) schemes have emerged as an innovative option to provide incentives for sustainable ecosystem management. PES involves payments from a beneficiary to a provider of an ecosystem service, and the basic rationale for PES is that the benefits for the beneficiary may exceed the (opportunity) costs for providing the service. A typical example is a PES scheme for maintaining the hydrological service, where downstream water users pay upstream land owners for maintaining the forest cover and, consequently, the regulation of downstream water supply.

PES schemes require the identification of providers and beneficiaries of the services, the valuation of the ecosystem service(s) to be included, and the set-up of a payment scheme that regulates the transfer of payments from beneficiaries to providers in return for maintaining the supply of the ecosystem service. PES approaches include schemes involving payments from governments to private stakeholders, and between private stakeholders. They can involve local and national as well as globally relevant ecosystem services and stakeholders.

A critical element in any PES scheme is the transaction costs, which are related to monitoring the supply of the service and the efforts undertaken by the providers of the service, and to the management and disbursement of funds, in particular, in the case where the service is provided by a range of different stakeholders. In the case where the transaction costs exceed the difference between the (opportunity) costs and the benefits, the economic rationale for the scheme ceases to exist. PES schemes are also unlikely to be successful if local beneficiaries are poor and have no funds available to pay for the ecosystem services they receive.

PES schemes are already having a major impact on promoting sustainable ecosystem management at the local and national scales. For instance, the US Government spends over US$ 1.7 billion per year to induce farmers to protect land (UNEP, 2005). In several Latin American countries, including Costa Rica, Mexico and Colombia, irrigation water-user groups, municipal water supply agencies and other governmental bodies have initiated and executed PES schemes aimed at maintaining downstream water supply (Pagiola et al., 2005). In Kenya, the Wildlife Foundation is securing migration corridors on private land through conservation leases at US$ 4 per acre per year (UNEP, 2005). In Costa Rica, a formal, countrywide PES programme has been established, extending to the hydrological service, biodiversity conservation and carbon sequestration (Pagiola, 2008).

A major benefit of PES schemes is that they are able to generate additional funding for sustainable ecosystem management, based on the supply of ecosystem services that were not previously marketed. Furthermore, PES schemes can generate a long-term flow of funds necessary to protect certain ecosystem services, A key challenge is to continue the development of PES schemes at the national, but, in particular, also at the global scale. Given that a large part of the world's biodiversity can be found in tropical zones, and that tropical forests and peatlands are major stocks of carbon, there is ample rationale for setting up global PES schemes for the biodiversity conservation and carbon sequestration services. This is an ongoing, and rapidly evolving process involving a wide range of institutions; see, for example, Gibbs (2007), Wunder (2007) and Bishop et al. (2008).

2.4.5 Constraints to and Criticism on Ecosystem Valuation

Various authors have criticised the economic valuation of ecosystems and ecosystem services, for example Vatn (2005) and Spash (2008), see also Pearce et al. (2005) for criticism raised with regards to environmental cost–benefit analysis. Vatn (2005) describes the following four principal points of concern regarding the valuation of ecosystem services: (1) a lack of full information on ecosystem services; (2) value incommensurability; (3) the problem of composition; and (4) the income-dependency of willingness to pay estimates. A brief overview of these points is provided below.

A lack of full information on ecosystem services

A lack of information is a frequent constraint to ecosystem services valuation. For instance, there may be only an approximate indication of the full array of benefits provided by an ecosystem, the marginal value of ecosystem services in the case of strong changes in supply, ecosystem dynamics and how they influence future supply of the service, etc. These constraints progressively increase at coarser scales and with increasing complexity of the ecosystem.

Value incommensurability

Value incommensurability means that different types of values, for instance the values related to biodiversity, cultural functions of ecosystems and values derived from products harvested in an ecosystem, cannot be measured on one and the same scale. This argument is based on the observation that individuals have different motives for managing ecosystems, and that they therefore have difficulty in interpreting services and values along one dimension (see e.g. Gregory et al., 1993).

The problem of composition

The problem of composition indicates that the supply of an ecosystem service is always dependent on the functioning of the ecosystem supplying the service, and that demarcating parts of the environment for the purpose of valuation may lead to underestimation of the value of the ecosystem at large.

The income-dependency of willingness-to-pay estimates

The income-dependency of willingness-to-pay (WTP) estimates is a concern where there are large income discrepancies between different stakeholders. The WTP estimate is bound by the income of the respondent and restricts the articulation of unrealistically high WTP statements in a contingent valuation study (Arrow et al., 1993). However, this also implies that the preferences of the rich will count for more than the preferences of the poor in a valuation study incorporating WTP estimates for ecosystem services.

These four points are each significant limitations to the applicability of ecosystem services valuation approaches. The problem of value incommensurability is fundamental. As analysed in, for example, Spash (2008), economic valuation implicitly takes a narrow-minded view of the various motivations that people have to relate to ecosystems and the environment in which they live in general. Consequently, capturing the multiple motivations people have to value ecosystems in the single metric of money will be prone to a significant degree of uncertainty. The issue of the income dependency of willingness-to-pay estimates can be partly overcome by using Willingness-To-Accept (WTA) indicators for the value of an ecosystem service. Also, poor people can demand high compensation for environmental damage. In principle, a loss of the ecosystem for a stakeholder would need to be valued on the basis of WTA in the case where the stakeholder holds a (formal or traditional) right to use the ecosystem (Vatn, 2005).

The problem of decomposition, and a lack of full information on ecosystem dynamics and the benefits provided by ecosystem services, are significant issues with regards to ecosystem management in general. It could be argued that ecosystem management is always faced with these constraints and that the approach of quantifying ecosystem services, in combination with modelling ecosystem dynamics (Chapter 3), provides some of the information required for informed decision making. Nevertheless, it is clear that value estimates are subject to uncertainty, which needs to be explicitly considered in any valuation study.

The degree of uncertainty varies between different ecosystem services. In general, it is relatively easy to value provisioning services, particularly if constant prices can be assumed. Regulating services require a certain degree of understanding of the processes taking place in ecosystems, and how these processes depend on ecosystem structure and functioning. In addition, when ecosystem services are considered at coarser scales, there are increasing constraints on how economic valuation can be applied – the value of essential life support services at the global scale (regulation of climatic processes, ultimately even oxygen production) is essentially infinite. Hence, a lack of full information is a particular issue with regards to valuing regulating services. For cultural services, value incommensurability is a key constraint. There may be a whole range of motives to value biodiversity conservation and the different cultural aspects related to ecosystems. For instance, it is questionable whether economic valuation of the preservation of a species, a specimen of a threatened species, or of human health, can meaningfully be done.

Hence, the degree to which the four constraints described in the bullet points above apply differs between the three types of ecosystem services. It is nevertheless clear that economic valuation of ecosystem service is no panacea and that there are various significant methodological constraints to ecosystem service valuation. At best, economic valuation of ecosystem services can meaningfully express a substantial part of the societal benefits provided by an ecosystem. Therefore, the main added value of ecosystem services analysis and valuation may be to allow (1) presentation of a comprehensive, qualitative overview of the various benefits provided by ecosystems; and (2) indication of the *minimum* economic value generated by an ecosystem, accounting only for those ecosystem services and ecosystem values that can meaningfully be quantified (see also Pearce, 2007; Daily et al., 2009).

Given these constraints, ecosystem services valuation can contribute to ecosystem management in a number of ways. First, in a whole range of countries and environments, there is as yet insufficient understanding of how ecosystems contribute to human well-being. In this case, there is a risk that the default value of ecosystems in decision making is set at zero. In the case of a lack of information on the value of ecosystems, identification and valuation of ecosystem services can provide an incentive for setting up more efficient and more sustainable management regimes.

Second, ecosystem services valuation allows incorporation of part of the societal costs of environmental degradation (and the benefits of rehabilitation) in cost–benefit analysis (CBA) – see also Section 7.4.2. Compared to a situation where environmental degradation is not included in cost–benefit analysis, this allows, in principle, enhanced decision making from both an economic efficiency and sustainability perspective. However, clearly, in the case where not all benefits provided by ecosystems can be meaningfully translated into a monetary value, care needs to be taken in the interpretation of cost–benefit analyses.

Third, analysing and, where feasible, valuing ecosystem services can assist in better understanding stakeholder motivations for ecosystem management. Commonly, different stakeholders depend on different services provided by ecosystems (see, for instance, the case study presented in Chapter 5), and analysing ecosystem services reveals how stakeholders may be differently affected by changes in ecosystems or ecosystem management.

Finally, quantifying and valuing ecosystem services may assist in establishing Payment Schemes for ecosystem services, as discussed previously in Section 2.4.5.

The argument in favour of ecosystem service valuation may be most pronounced in developing countries with a lack of financial resources to maintain natural areas and obvious reasons to increase local income levels. In these countries, the incentive to preserve natural ecosystems is likely to increase if local and/or national income can be generated with payments for ecosystem services, in particular carbon sequestration and biodiversity conservation. This is made explicit, for instance, in a statement by the Office of the President, Republic of Guyana (2008), which indicates that, given the global importance of carbon sequestration and biodiversity conservation in Guyanan forests, international payments to contribute to the (opportunity) costs of managing and preserving these forests are a prerequisite for their long-term conservation.

In summary, the added value of ecosystem services valuation is connected to economic efficiency, being one of the criteria commonly applied in decision making on ecosystems, and the general lack of understanding of benefits supplied by ecosystems. Nevertheless, ecosystem services valuation, and analysis of the efficiency of ecosystem management options is subject to a number of constraints, which means that in many cases only a minimum economic value can be meaningfully estimated. As with any assessment technique, in the economic valuation of ecosystem services, uncertainty ranges should be indicated, and it should be made explicit which services and value aspects were, and were not, included in the valuation.

3. A quantitative ecological–economic assessment approach

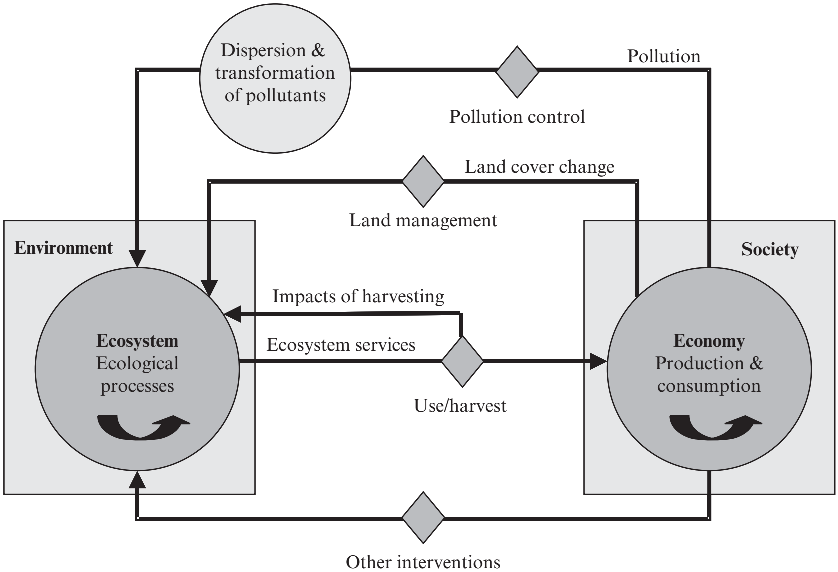
3.1 A FRAMEWORK FOR ANALYSING ECOSYSTEM MANAGEMENT OPTIONS

This chapter presents a quantitative, dynamic framework for ecological–economic assessment and modelling of ecosystem management options. The framework can be used to assess the efficiency, sustainability and equity aspects of ecosystem management options, while accounting for the complex dynamics of ecosystems, and is visualised in Figure 3.1. The interactions presented in the framework need to be expressed in monetary terms when their impact on the economy is studied, and in biological or physical units when their interactions with the ecosystem are studied.

The framework includes four types of management measures: (1) the use or harvest of ecosystem services; (2) pollution control, e.g. by applying waste-water treatment; (3) managing land cover; and (4) other, direct interventions in the ecosystem. Each measure corresponds to a decision variable. These measures, in combination with ecological processes, determine the dynamics of the ecosystem – which can proceed according to different ecological models. Subsequent changes in the state of the ecosystem influence its capacity to supply ecosystem services. The total welfare supplied by the ecosystem is a function of the net benefits provided by ecosystem services, and the costs of pollution control, ecosystem interventions and managing land cover change.

When the figure is applied, the various management options, the dynamics of the ecosystem, and ecosystem services supply need to be quantified for the specific ecosystem involved. The different elements of the framework are elaborated below.

In Figure 3.1, the ecosystem changes over time as a function of (1) internal, ecological processes; (2) changes in the overall environment in which the ecosystem is embedded; and (3) ecosystem management. Internal ecological processes, for example succession or predation, take place over a range of temporal and spatial scales (e.g. Holling, 1992; Holling et al., 2002). In addition, the ecosystem, and the processes driving ecological change, are influenced by changes taking place in the overall environment



Notes: Diamonds indicate the management options.

Figure 3.1 Framework for assessment and modelling of ecosystem management options

in which the ecosystem is embedded, for instance climate change that may affect temperature and rainfall patterns.

The economy functions within the context shaped by society. It's functioning is regulated by market dynamics and influenced by institutions, such as organisational and ownership arrangements prevailing in a society, etc. (e.g. North, 1990). Two central interrelated processes in the economy are production and consumption. Ecosystem services can be one of the inputs required for production (e.g. by providing a raw material). Ecosystem services can also support consumption (e.g. by providing an opportunity for recreation).

Ecosystem services represent the benefits provided by ecosystems, including provisioning, regulating and cultural services. The use of ecosystem goods and services is driven by developments in the economy (the demand for ecosystem services) and the ecosystem itself (the capacity of the ecosystem to supply services). The use of ecosystem services has an impact on the ecosystem. Provisioning services are extractive, and the impact depends upon the harvest level in relation to the carrying capacity of the ecosystem, as well as the harvesting techniques applied.

For instance, timber harvesting will affect the forest's remaining standing stock and may cause soil compaction from the use of machinery to transport the felled trees out of the forest. Regulating services relate to the external impacts of the ecosystem, and their supply therefore tends to have little impact on the ecosystem. Cultural services may involve visits to the ecosystem, in which case the impact depends upon the number of visitors and their activities in relation to the carrying capacity of the ecosystem for these activities.

Pollution comprises a second type of impact of the economy on ecosystems. Consumption and production processes may lead to pollution including discharges to water, emissions to atmosphere, and the generation of solid waste. The possibility of disposing of unwanted substances provides a benefit for the economic system, but may have an adverse impact on the functioning of the ecosystem and the supply of ecosystem services. In general, pollution control can take place by modifying the production process or consumption pattern, changing the location where the pollutant enters the environment, or applying treatment processes (end-of-pipe technologies). The residual impacts of pollution on an ecosystem depends upon the total emissions and amounts of waste released, the type of pollutant, the application of waste treatment technologies, the breakdown of the polluting compounds in the environment and dispersion processes that determine how much of the pollutant ends up in the ecosystem (RIVM/UNEP, 1997).

Changes in land cover are a third principal mechanism through which economic activities influence the ecosystem. Land cover change involves, for example, the conversion of a forest to agricultural land, or of agricultural land into urban land. Land cover change may proceed in an abrupt manner as well as more gradually, and generally involves a marked change in the type of ecosystem services supplied. Ecosystem services may be connected in different ways to land use. Most provisioning services are strongly linked to the ecosystem productivity of a particular plot, e.g. the production of wood (mean annual increment) depends on the vegetation and biophysical characteristics of a particular site (soils, temperature, water, etc.). However, many regulating and cultural services also depend on the spatial configuration of the landscape. For instance, the hydrological service depends on the vegetation cover across the watershed, and the recreational service of an extensive agricultural landscape may be a function of the mix of agricultural and natural elements in the landscape (e.g. Willemen et al., 2008). For these services, land cover changes need to be analysed, in a spatially explicit manner, at the scale of the watershed or landscape.

Ecosystem interventions comprises the fourth, broad category of human impacts on the ecosystem. It includes both measures with a positive impact on the ecosystem, such as reforestation in a degraded forest or biomanipulation of a lake (Chapter 5), as well as human disturbance of the ecosystem, for instance, through the introduction of alien invasive species. Ecosystem maintenance and rehabilitation measures will come at a cost to the economic system, but may result in the increased supply of ecosystem services. Human disturbance may also take place through the generation of negative external impacts, for example in the case of the construction of a highway nearby a national park that may affect recreational or habitat services through noise pollution.

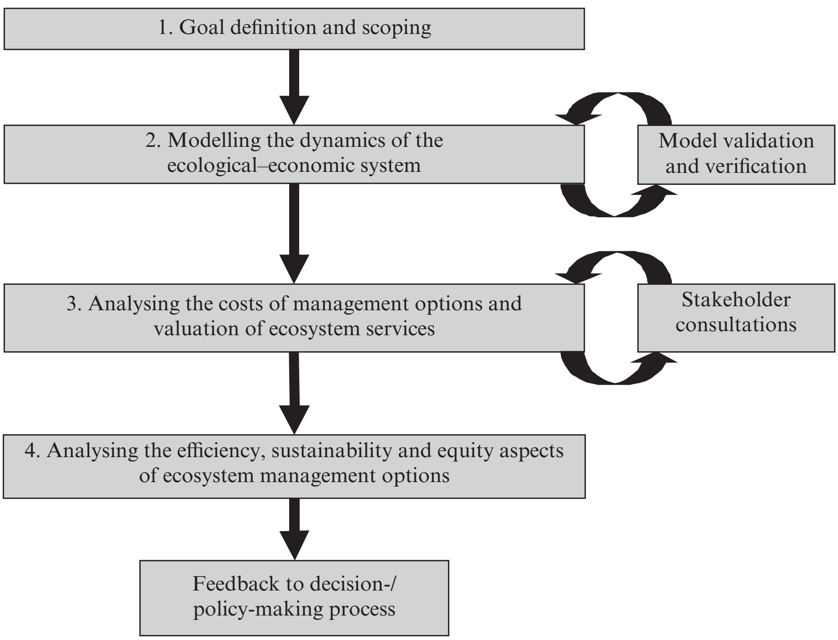
Ecological and economic processes, and interactions between the two systems, take place across a range of spatial and temporal scales. For instance, pollutants may be dispersed at all scales, ranging from water pollution affecting a local pond to greenhouse gas emissions affecting the global atmosphere. Figure 3.1 depicts management options at the scale of the ecosystem, but application of the framework requires that the various scales at which drivers for ecosystem change operate and the different scales at which ecosystem services may be supplied are accounted for.

Application of the framework is generally data-intensive, requiring ecological data series in order to model the dynamics of the ecosystem, and economic valuation data for the ecosystem services supplied by the ecosystem. Quantifying economy–ecosystem interactions requires the application of basic mathematics. This chapter presents some of the basic equations that can be used to analyse the efficiency, sustainability and equity implications of ecosystem management options. The different modelling approaches are further illustrated in the case studies presented in Chapters 4 to 6.

Note that Figure 3.1 can be easily related to the DPSIR framework (Driving forces, Pressures, State, Impacts and Responses) commonly used in environmental sciences (OECD, 1979). Consumption and production processes constitute the main *drivers* for the interactions between the economic system and the ecosystem. The production and consumption processes entailed in the economic system can result in four basic types of *pressure* on the ecosystem, for example, in the case of extraction of ecosystem services at a level exceeding the carrying capacity of the ecosystem. The pressures may lead to a change in the *state* of the ecosystem. The final order *impact* of these changes is expressed as a change in the capacity of the ecosystem to supply ecosystem services. In *response*, several measures can be taken, aimed at modification of the driving forces, reducing the pressures, or rehabilitating the state of the ecosystem through the implementation of ecosystem interventions.

3.2 APPLICATION OF THE ASSESSMENT FRAMEWORK

Application of the framework in order to model ecosystem–economy interactions at the scale of the ecosystem involves four main steps: (1) definition of the objective of the assessment; (2) modelling the dynamics of the ecological–economic system; (3) analysing the costs of management options and valuation of ecosystem services flows; and (4) analysing the efficiency, sustainability and equity aspects of the ecosystem management options, as shown in Figure 3.2. These four steps are described below. Part of Step 2 is to validate and verify model outcomes, based on ecological data such as time series of specific indicators of the ecosystem state. Step 3 normally requires consultation of stakeholders in order to identify and assess ecosystem management options that also meet legal and technical criteria, and to identify and verify value estimates for



Notes: Selected stakeholders will often be consulted in all steps of the assessment. Step 3, in addition, requires systematic and comprehensive stakeholder consultation, in order to reveal perspectives and values of all relevant stakeholders involved in the management of the ecosystem under consideration.

Figure 3.2 Basic steps in ecological–economic assessment and modelling ecosystem services. Clearly, stakeholder interactions are not limited to Step 3, stakeholder inputs drive the process of initiating and scoping the ecological–economic assessment, and documenting, reporting and presenting the outcomes to stakeholders will normally be an integral part of the analysis.

1. Goal definition and scoping

At the scale of the ecosystem, ecological–economic analysis can be used for a number of purposes. First, to analyse the present and future flows of benefits generated by an ecosystem, under current management. This type of analysis should include modelling ecosystem change, as a function of internal drivers (e.g. resource harvesting) and external drivers (e.g. climate change) in order to assess if present flows of benefits can be sustained. Benefits can be linked to stakeholders in order to reveal stakeholders' interests in ecosystem management.

Second, ecological–economic modelling can support the analysis of the economic benefits (including non-market benefits), and impacts on different stakeholders of specific ecosystem management options. Feeding the ecosystem management options into the model can show the implications of each option and assist in selecting the preferred ecosystem management option.

Third, ecological–economic modelling can assist in identifying the most efficient management option (i.e. generating the highest discounted net economic benefits) or the optimal management option (using a social welfare function to account for different impacts on stakeholders). Identification of the most economic efficient, or 'efficient' ecosystem management option requires an additional optimisation step. The social welfare function can include aspects such as income differences between stakeholders; for example giving higher weights to benefits generated for poor stakeholders. These various aspects are elaborated in the remainder of this chapter.

2. Modelling the dynamics of the ecological–economic systems

This step involves the modelling, in physical terms, of the dynamics of the ecosystem, and the impact of changes in ecosystem state on the system's capacity to supply ecosystem services. First, this requires definition of the ecological–economic system to be studied, in terms of its spatial and temporal boundaries. Relevant spatial boundaries are the spatial delineation of the ecosystem to be analysed, as well as identification of the scales that will be included with regards to analysing the benefits provided by the ecosystem. The latter may range from local stakeholders benefiting from provisioning services up to global benefits such as carbon sequestration.

Depending on the objective of the analysis, local to global benefits need to be included in the assessment.

Next, this second step involves the identification of the interactions between the ecosystem and the economic system, including the internal and external drivers for the ecosystem, the key ecological processes that guide the dynamics of the ecosystem, and the various services supplied by the ecosystem and how these services depend on the state of the ecosystem. For systems subject to complex dynamics, it is important that these dynamics are reflected in the model. This requires the modelling of the main ecosystem components and the feedback mechanisms between them, including relevant non-linear and/or stochastic processes. In spite of the large number of ecological processes regulating the functioning of ecosystems, recent insights suggest that the main ecological structures are often primarily regulated by a small set of processes (Harris, 1999; Holling et al., 2002). This indicates that inclusion of a relatively small set of key components and processes in the model may be sufficient to represent the (complex) dynamics of the system with a level of accuracy that is adequate for the purpose of the assessment. Chapters 4, 5 and 6 illustrate how ecosystem dynamics and service supply can be captured in a (small) set of differential equations, for three different ecosystem types.

3. Analysing the costs of management options and valuation of the ecosystem services

In this third step, the interactions between the economy and the ecosystem need to be expressed in a monetary measure. This involves both quantifying the costs of the ecosystem management options, for example, through the establishment of a pollution abatement cost curve, and the valuation of changes in the supply of ecosystem services. Generally, valuation of the costs of management options is straightforward. These require specific investments, for which cost estimates can be obtained from, for example, vendors of equipment. Valuation of ecosystem services flows, or changes in ecosystem services is generally more complex, particularly if changes in ecosystem services supply affect prices and demand and supply curves for ecosystem services need to be established. Valuation can be particularly challenging for non-market benefits where there tends to be a shortage of information on (shadow) price changes as a function of changes in supply and demand. Comprehensive stakeholder consultation is required in this step in order to identify ecosystem management options that are potentially acceptable to stakeholders, and to analyse the value attributed by stakeholders to ecosystem services, in particular for those ecosystem services that can not be valued based on market prices.

4. Analysing the efficiency, sustainability and equity aspects of management options

Once the ecological–economic model has been constructed, it can be used to assess the efficiency and sustainability of different ecosystem management options. The efficiency of ecosystem management can be revealed through comparison of the net welfare generated by the ecosystem and the costs involved in maintaining and managing the ecosystem (e.g. Pearce and Turner, 1990). Through a simulation or algebraic optimisation approach, efficient management options, i.e. management options that provide maximum benefits given a certain objective function, can be identified. The sustainability of management options can be examined by analysing their long-term consequences for the state of the ecosystem including its capacity to supply ecosystem services (Pearce et al., 1989; Barbier and Markandya, 1990). Equity aspects require a further interpretation of the outcomes of the model. The model allows the specification of how ecosystem change will affect different stakeholders, who tend to benefit from different ecosystem services. Weighting and comparison, in a quantitative manner, of impacts on different stakeholders requires a further interpretation, for example with a social welfare function. Where the model includes different drivers, efficient combinations of ecosystem management can be identified, for example a specific reduction in pollution loading in combination with a change in resource harvest levels.

Identifying the efficiency, sustainability and equity aspects of ecosystem management options is only the first step in developing a comprehensive resource management strategy. Once the preferred ecosystem management strategy has been defined, the second step is to set up the institutional arrangement required to manage the ecosystem in the preferred way, including monitoring and enforcement approaches. Stakeholder involvement is a critical step with regards to both identifying the preferred ecosystem management regime and to adjusting or setting up these institutional arrangements. This book focuses on the first step, i.e. identifying efficient, sustainable and/or equitable ecosystem management options. For further information on the institutional aspects of ecosystem management, see, for example, Gunderson et al. (1995), Leach et al., (1999), Costanza et al. (2001) and Vatn (2005).

The next sections of this chapter present the framework for analysing and modelling the efficiency, sustainability and equity of ecosystem management, based on the flowchart presented in Figure 3.1. In particular, they present a number of general models for ecosystem change, provide further guidance on analysing the efficiency, sustainability and equity aspects of ecosystem management and indicate potential sources of uncertainty in applying the framework.

3.3 MODELLING ECOSYSTEM CHANGE

3.3.1 General Models of Ecosystem Change

Early models of ecosystem dynamics were based upon the Clementsian theory of ecological succession (Clements, 1916; Weaver and Clements, 1938). In these models, succession comprises the subsequent dominance of a range of relatively stable communities, for instance from grass to shrub to forest, called seres. It was assumed that any particular ecosystem has a single, persistent state, called the climax, which represents the end stage of a successional series. Succession to the climax is a steady process that can be reversed by disturbances such as fires or sustained drought.

Subsequently, many advances in the understanding of ecosystem dynamics have been made. Tansley (1935) described the 'polyclimax' theory that stated that the climax vegetation of a region may consist of a mosaic of vegetation climaxes controlled by soil moisture, nutrients, slope exposure, fire and animal activity. Watt (1947) recognised that succession often represents phases in a cycle of vegetation development. Cyclic replacement results from the destruction of existing vegetation by disturbance or some characteristic of the dominant organisms, and is followed by re-establishment of the vegetation. The occurrence of fluctuations, i.e. non-successional or short-term reversible changes, was described by Rabotnou in 1974. They may be the result of temporary environmental stresses, such as fire, moisture fluctuations, wind, etc. (Rabotnou, 1974 in Smith, 1990).

A more recent general model describing complex ecosystem dynamics is the 'adaptive cycle' model (Holling et al., 2002). This model contains four phases in the development of the ecosystem: (1) exploitation (colonisation of disturbed areas); (2) conservation (slow accumulation and storage of material and energy); (3) release (creative destruction); and (4) reorganisation (preparing for the next phase of exploitation by making space and nutrients available). A well-known illustration of this is provided by the spruce-fir forest–budworm cycle that occurs in the eastern part of North America. Mature forest accumulates a high volume of foliage that dilutes the effectiveness of the search for budworms by insectivorous birds. This triggers an insect outbreak resulting in the death of a substantial proportion of the trees. This is followed by the death of the budworms and the regrowth of the trees, and the cycle repeats itself (Ludwig et al., 1978; Clark et al., 1979).

3.3.2 Complexities in Ecosystem Dynamics

Characteristic of the more recent theories on ecosystem change is the distinction of various types of complex dynamics in ecosystems. Complex dynamics are irreversible and/or non-linear changes in the ecosystem as a response to ecological or human drivers. Below, the following types of complex dynamics are briefly examined: irreversibilities; multiple states and thresholds; and stochasticity and lag-effects. These complex dynamics occur in a wide range of ecosystems, and have a major impact on the efficiency, sustainability and equity impacts of ecosystem management options.

Irreversible dynamics

Irreversible changes in ecosystems occur when the ecosystem is not, by itself, able to recover to its original state following a certain disturbance. Irreversible changes may be permanent, as in the global loss of a species, or they may only be reversed through substantial interventions in the ecosystem, for example, in the case of reforestation on sites where natural processes would not lead to recovery of the tree cover. Irreversibility comprises different mechanisms, and can take place at different scales. For instance, it can relate to the extinction of a particular species, or the conversion of an ecosystem (e.g. Barbault and Sastrapradja, 1995). It may also refer to irreversible changes in the state of an ecosystem, as in the case of a transition from a rangeland dominated by palatable grasses to one dominated by unpalatable shrubs (Laycock, 1991). Recuperation of the ecosystem may be prohibited by certain processes, such as rapid erosion in a badland, or can be constrained by the amount of time needed to regain the system, as may be the case with regrowth of tropical forests. Irreversibility can also be related to the building-up of a stock. At the level of a lake, pollution loading may be irreversible if the pollutant is not decomposed and if the lake does not drain elsewhere (e.g. Larsen et al., 1981). At the global scale, the increased loading of the atmosphere with carbon dioxide is an example of a process that can be considered as partly irreversible at human time scales (IPCC, 2007). Irreversible change may either be rapid, involving a threshold, or more gradual. Often, it is subject to considerable uncertainty, for instance, with reference to the point where the change has become irreversible (e.g. Scheffer and Carpenter, 2003).

Multiple states and thresholds

Multiple states are relatively stable configurations of the ecosystem, caused by the existence of feedback mechanisms that reinforce the system to be in a particular state (Carpenter et al., 1999; Scheffer et al., 2001). The state of the ecosystem is determined by its historical development, such as a different sequence of recruitment (e.g. Drake, 1990) or may be a consequence of physical or biological perturbation, such as changes in nutrient loading or species depletion or invasion (e.g. Barkai and McQuaid, 1988). The probability that a disturbance leads to a shift from one state to the next depends upon the magnitude of the disturbance and on the resilience of the current state. Often, the shift between multiple states occurs suddenly and comprises the existence of threshold effects (Wissel, 1984; Muradian, 2001). Multiple states and thresholds have been observed in a range of ecosystems, including freshwater lakes (Larsen et al., 1981; Timms and Moss, 1984), marine fish stocks (Steele and Henderson, 1984; Steele, 1998), woodlands (Dublin et al., 1990), rangelands (Friedel, 1991), coral reefs (Knowlton, 1992) and coastal estuaries (Murray and Parslow, 1999).

A particular type of dynamics that occurs in some ecosystems, in conjunction with multiple states and thresholds, is hysteresis. Hysteresis occurs when the ecosystem's response to an increasing pressure follows a different trajectory from a response to a release in pressure (Scheffer et al., 2000). An example is provided by the response of an estuary to nutrient loading. At low nutrient loads, seagrass may dominate the flora, but with increased nutrient loading the phytoplankton concentrations gradually increase. At a critical load, the phytoplankton concentration is so high that seagrass does not have enough light to grow. The seagrass population collapses, which further increases the turbidity of the water because of the resuspension of sediments no longer trapped by the seagrass. To re-establish the seagrass beds, nutrient loads have to be reduced considerably below the critical load (Borum and Sand-Jensen, 1996; Murray and Parslow, 1999). Other ecosystems in which hysteresis has been detected include shallow lakes (Timms and Moss, 1984), rangelands (Walker, 1993), hemlock–hardwood forests (Augustine et al., 1998) and deep lakes (Carpenter et al., 1999).

Stochasticity and lag-effects

The ecosystem may also develop as a consequence of stochastic natural conditions, for instance when ecosystem change is driven by fires or high rainfall events. In the marine environment, major changes in the dominant fish species occupying a particular niche may be triggered by relatively minor, stochastic fluctuations in the fish community (Steele and Henderson, 1984). Lag effects appear when impacts of specific drivers occur with a certain delay, for example, because changes need to be triggered by a specific event.

These main types of complex dynamics are of major importance for the understanding of ecosystem dynamics (Scheffer et al., 2001). They also determine the response of the ecosystem to management, including changes in the capacity of ecosystems to provide goods and services following the implementation of management measures. Hence, consideration of these dynamics, where they occur, is required in order to ensure the ecological realism of ecological–economic models.

3.3.3 The Resilience of Ecosystems

The concept of resilience has been widely used in the analysis of ecosystem dynamics (Carpenter et al., 2001). There are two main definitions of resilience. According to the first definition, resilience measures the ability of a system to resist disturbance as well as the rate at which it returns to equilibrium following disturbance (Pimm, 1984; Tilman and Downing, 1994). This definition has been used, in particular, to analyse ecosystem stability near an equilibrium steady state (Holling and Gunderson, 2002). In the second definition, resilience expresses the capacity of a system to undergo disturbance and maintain its structure, functions and controls (Holling, 1986; Holling and Gunderson, 2002). This definition emphasises conditions far from an equilibrium steady state, where instabilities can flip a system into another regime of behaviour or steady state (Holling and Gunderson, 2002). The applicability of these two definitions depends upon the dynamics of the particular ecosystem involved and the magnitude and type of disturbance studied. Whereas the first definition may be most applicable to systems subject to gradual changes, the second definition is likely to be more applicable for ecosystems subject to multiple states and thresholds. The resilience of an ecosystem varies for different types of disturbances (Carpenter et al., 2001). For instance, a particular type of forest may show different degrees of resilience to perturbations caused by logging or fire. Resilience may also vary between different ecosystem states, for example a lake ecosystem may have a different resilience for a particular type of stress in a clear water state compared to its resilience in a turbid water state (Carpenter et al., 2001).

A much-debated issue is the relation between the loss of biodiversity and a decrease of resilience of ecosystems. At two extremes in the discussion are the rivet and the functional redundancy hypothesis. The rivet hypothesis (Ehrlich and Ehrlich, 1981) states that all species contribute to the maintenance of the functioning of the ecosystem. The functional redundancy hypothesis states that a limited number of species (the so-called keystone species) are responsible for the maintenance of the functioning of the ecosystem and that these species can take over each other's role if some of them disappear (Walker, 1992). In an intermediate and more widely accepted hypothesis, Walker (1995) argues that both species diversity and functional diversity are important, but that the diversity of species within each functional guild is the most important factor in maintaining ecosystem resilience. Ecosystem functioning can be largely maintained despite loss of species diversity until the final species representing functional guilds begin to disappear (Mageau et al., 1998). In this sense, biodiversity can be seen as providing insurance capital for securing the functioning of the ecosystem (Barbier et al., 1994).

3.3.4 Modelling Ecological–Economic Dynamics

Model types

Environmental and ecological–economic models can be used to forecast changes in an ecosystem as a function of input variables, as well as to maximise (or minimise) the output of the ecosystem as a function of decision variables and ecological and economic parameters and variables. Ecological–economic modelling can be pursued through either the construction of an integrated model covering both ecological and economic processes, or by employing a system of heuristically connected sub-models (Turner et al., 2000). In this book, only integrated models are considered. There are a number of integrated approaches to environmental economic modelling, including: generalised input–output models; neo-classical growth models; dynamic system models; and spatially explicit models (Van den Bergh, 1996; Turner et al., 2000). For an overview of other models used in environmental economics (e.g. neo-Keynesian and game-theoretic models), that are less relevant for the ecological–economic modelling of ecosystems, the reader is referred to, for instance, Van den Bergh et al. (1988), Faucheux et al. (1996) and Folmer et al. (1998). The four selected ecological–economic model types are briefly described below.

The input–output approach The input–output approach allows analysis of the interactions between components in the ecological and economic system, and has been frequently applied in environmental and ecological economics (Van den Bergh, 1996). Input–output models are based upon a transactions table that records the flows of goods between and within the different economic sectors and the ecosystem. This type of models is, however, subject to severe limitations, for example, they are not able to account for profit-maximising behaviour. In response to these limitations, general equilibrium models have been developed (Greenaway et al., 1993). These models allow for substitution and an optimal choice of the input mix in production and consumption functions, based on, for example, cost minimisation. In addition, prices are endogenised by linking them to the volumes and allocation decisions in the models (Van den Bergh, 1996). Examples of an input–output modelling approach implemented at the level of the ecosystem include, for instance, Midmore and Harrison-Mayfield (1996). Characteristic for both the input–output and the general equilibrium approach is that the models tend to be highly aggregated, in comparison to the level of aggregation encountered in most ecological models (Deacon et al., 1998; Perrings, 1998). A major limitation of these models is that there is limited scope to model complex economy–ecosystem interactions and ecological processes at the appropriate ecological scale.

Neo-classical growth models Neo-classical growth models deal with the development of (a sector of) the economy over time. In this type of modelling, economic growth is linked to the growth rate of labour supply (linked to population increases), capital accumulation (through savings) and technological progress (Solow, 1956). Hence, for a fixed population level, and with a given savings rate, technological progress is the main motor for economic development. In environmental economics, neo-classical growth models can be used, for instance, to examine the potential development of emissions and the demand for resource inputs over time (see Barro and Sala-i-Martin, 1995).

Dynamic systems models A systems approach is based upon the modelling of a set of state (level) and flow (rate) variables in order to capture the state of the system, including relevant inputs, throughputs and outputs, over time. Dynamic systems models use a set of differential equations to capture the dynamics of the ecosystem. A dynamic systems model contains state and flow indicators and variables that capture, for instance, the amount of standing biomass (state), the harvest of wood (flow) and the price of wood (time dependent variable). The models runs on the basis of predefined time-increments and requires fully defined initial conditions. For a theoretical treatment of dynamic systems models, the reader is referred to, for example, Huggett (1993). The systems model may comprise a range of theoretical, statistical or methodological constructs, dependent upon the requirements of the model. The systems approach can contain non-linear dynamic processes, feedback mechanisms and control strategies, and can therefore deal in an integrated manner with economic–ecological realities (Costanza et al., 1993; Van den Bergh, 1996). Therefore, it is more suitable to examine the implications of ecosystem complexities for the efficient management of ecosystems compared to the two previous model types.

Spatially explicit models Contrary to the other types of models, spatially explicit models allow for dealing with the spatial variations of ecosystems and economic systems. Spatial models are generally developed in a GIS environment. Examples of this approach are provided in Geoghegan et al. (1997) and Voinov et al. (1999). The approach offers a number of new possibilities, such as the optimisation of spatial planning processes (Bockstael et al., 1995). The spatial variation of ecological processes has been elaborately studied, for instance in the fields of eco-hydrological models (e.g. Pieterse et al., 2002), and erosion and soil transport models (e.g. Schoorl et al., 2002). In economics, spatial, urban and transport economists have paid special attention to studying the spatial allocation of resources (see Fujita et al., 1999). Groeneveld (2004) offers an example of how the cost-effectiveness of biodiversity conservation in different spatial patterns of agricultural and natural land use can be calculated. Data requirements are generally high for a spatially explicit approach. Initial conditions, processes and implications of decision variables need to be specified for each spatial unit.

Given the high aggregation levels at which input–output and neo-classical growth models can most effectively be applied, and their limited scope for inclusion of ecological feedbacks and non-linear responses, dynamic systems modelling is best targeted for the modelling of complex dynamic ecosystems. A dynamic systems modelling approach to reveal the ecological and/or economic impacts of ecosystem management options has been applied in numerous case studies in the past decades, see, for example, Costanza and Ruth (1998), Grasso, (1998) and Eriksson and Hammer (2006). As further elaborated below, various types of dynamics and interactions can be modelled on the basis of specifying relations between state and flow variables. Dynamic systems models can be integrated in a GIS by defining initial conditions, ecosystem dynamics and spatial interactions for each spatial unit in the GIS. A dynamic systems modelling approach is also used in the case studies presented in Chapters 4 to 6.

Applying dynamic systems models

Over the last four decades, dynamic systems models have widely been used as the basis for bio-economic and ecological–economic modelling. The initial bio-economic models dealt with, for instance, identifying efficient harvesting strategies for a renewable resource (e.g. Clark, 1976). They were based on a simplified set of assumptions regarding the ecological dynamics of that resource, with a key assumption being that changes in population sizes are gradual and a function of harvest rates, and in some cases one or a few environmental conditions. Harvesting rates and costs of the resources is typically a function of the population size of the resource being exploited (Perrings, 2000). These models generally followed a dynamic systems modelling approach, with resource stocks depicted by one or more state variables, and harvesting and growth as the flow variables. Typically, the ecosystem dynamics are represented by a system of differential equations, that have the following general form, with S the size of the stock and s the harvest levels, at time t:

$$dS/dt = f(s_t, S_t) \tag{3.1}$$

This functionality expresses that the growth of the stock (dS/dt) is both determined by (1) the size of the stock (S_t), usually in relation to the ecosystem's carrying capacity for that stock; and (2) the annual harvest of the stock (s_t). The growth function of the stock may take the form of, for instance, a logistic growth curve (i.e. an 's-curve'), or a set of Lotka-Volterra equations (which indicate predator–prey relations). The reproduction rate of a species generally depends, among others, upon the size of the stock, expressed as for example the numbers of individuals of a species present, in relation to the carrying capacity of its environment (e.g. Pielou, 1969). A well-known example of this type of model is the fisheries model first constructed by Gordon (1954) and Schaefer (1957). The growth function may also include a number of variables that allow for the modelling of the impact of various environmental or management factors on the growth. These factors may express, for example, how pollution leads to a reduction of the carrying capacity for a particular species, or affects the reproduction rate.

In recent years, building on the original bio-economic models that linked ecological stocks to economic outputs, a suite of more refined models has been developed. Whereas the original bio-economic models tended to focus on maximising an economic objective function, and generally included the dynamics of only one or a limited number of species, more recent ecological–economic models tend to have more detailed ecological relations, and also allow for a broader range of applications. Many ecological–economic models explicitly consider complex ecosystem dynamics such as irreversibility, multiple states and thresholds (e.g. Grasso, 1998).

Modelling complex dynamics

Strictly speaking, even the simple logistic growth model contains a basic type of irreversibility: once the resource is totally depleted, it will no longer recover. A slightly more advanced interpretation of the logistic growth curve assumes that a minimum stock is required to allow for recovery. In more complex models, irreversibility can also be introduced by limiting or eliminating the reproduction or growth rates of particular species when other components are affected, for instance by pollution (e.g. Wolff, 2000). The economic implications of discontinuous and irreversible ecosystem change have, for instance, been examined by Arrow et al. (1995), who provide a general economic model for dealing with irreversibility and derive that the maintenance of the resilience of ecosystems is an important factor for economic efficient management.

Stochastic events, such as weather extremes, fires or pest outbreaks, may affect the ecosystem state directly, but can also cause fluctuations in the conditioning factors of the ecosystem (Scheffer et al., 2001). In a growth function based upon a logistic growth curve, stochasticity may be expressed through fluctuations in either the stock, the growth factor or in the environmental parameters that determine the ecosystem's carrying capacity for the species involved. For instance, the regrowth of a fish stock following a period of intensive fisheries may depend upon the characteristics of the remaining fish population (age, size, etc.) as well as environmental parameters such as water temperature, fluctuations in feed availability, etc. There are ample examples of ecological–economic models that have accounted for stochasticity. For instance, Reed (1988), Perrings (1997) and Bulte and Van Kooten (1999) examine the implications of stochasticity for the efficient management of fish populations, rangelands and metapopulations, respectively.

In general, modelling complex ecosystem dynamics requires the employment of sets of (differential) equations to capture drivers, changes in ecosystem state, and resulting implications for ecosystem services supply. Common challenges in ecosystem modelling are (1) to simplify the complexity of the ecosystem in a small set of key relations; and (2) to find the data to calibrate and operate the model, in particular, if time series are required to calibrate the ecosystem's response to different levels of stress or different types of management. The model needs to be developed based on sound understanding of the type of dynamics and of the major control variables occurring in the studied ecosystem. This is illustrated in the three case studies presented in Chapters 4 to 6, where different models are developed for the forest, shallow lake and rangeland ecosystem.

3.4 ANALYSING THE EFFICIENCY, SUSTAINABILITY AND EQUITY OF ECOSYSTEM MANAGEMENT

3.4.1 Efficient Ecosystem Management

In the context of dynamic systems models, two approaches can be followed to determine the value of the decision variables that provides maximum net economic benefits: (1) a simulation (programming) approach; and (2) an algebraic, static or dynamic optimisation approach. In both cases an ecological–economic model is developed, but the optimal solution is found in different manners. In the simulation approach, a model is developed to represent modifications in the ecosystem and the economic system, and the key interactions as a function of the decision variable(s). By simulating the development of the ecosystem for a range of values of the decision variables, optimal solutions can be revealed – within the tested range and under the tested conditions. For simulation approaches, specific programs are available such as Stella or SIMULI, although basic models can also be constructed in a spreadsheet program.

In the algebraic optimisation approach, optimal solutions are found in a numerical or algebraic manner, through the preparation of the Hamiltonian function and solving the first and second order conditions (Chiang, 1992). This approach has the advantage that efficient solutions can be identified with greater accuracy, and that efficient solutions can be proven rather than demonstrated. It also avoids the risk that optimal solutions that are not in line with the time increments of a simulation model are overlooked. However, a disadvantage is that, with an increasing complexity of ecological dynamics and ecosystem–economy interactions, it is becoming progressively more difficult to construct and solve the algebraic equations depicting efficient solutions. In these cases, a simulation approach may be the only feasible option.

Following the framework presented in Figure 2.3, there are four principal types of ecosystem management: (1) changing the use level of ecosystem services; (2) the control of pollution influxes; (3) land cover change; and (4) direct interventions in the ecosystem. Below, it is analysed how the efficiency of these types of measures can be assessed, and the conditions that need to be met to achieve efficient management are considered. Subsequently, the way the management of ecosystems subject to more than one type of interaction can be optimised is briefly discussed.

Efficient extraction of renewable resources

Efficient resource extraction has been studied for over a century. Early contributions focussed on forestry (Faustmann, 1849), whereas studies on fisheries management (Gordon, 1954; Schaefer, 1957) and grazing systems (Dillon and Burley, 1961; Hildreth and Riewe, 1963) are more recent. The standard models assume a logistic growth curve, with low resource growth at low population sizes and at population sizes close to the carrying capacity (Pielou, 1969). In addition, these models may consider quality and price changes, cost for inputs and harvesting costs. Forest management models have dealt with, in particular, the choice of the optimal rotation period, while in fisheries and grazing systems, the key decision variable is the harvest rate. For forest stands, Faustmann models have been widely used to optimise rotation periods (Tahvonen, 1991). The basic principle of these types of model is that, for the economic efficient rotation period, the marginal value of the growth of the timber stock equals the marginal opportunity costs of not harvesting. The opportunity costs depend upon the costs of capital and the interest foregone on the site value of the land (Faustmann, 1849; Brazee, 2001). Assuming perfect markets and perfect foresight, the model leads to a constant rotation period that maximises the present value of forest land over an infinitely long time horizon (Samuelson, 1976). The original models mentioned above assumed relatively simple ecosystem dynamics, and these dynamics have been much refined in a wide range of more recent studies. For example, Tahvonen (2004) extended the original Faustmann model to account for different age classes, and Janssen et al. (2004) developed a rangeland model including different drivers for ecosystem change including rain, fire and grazing. A range of other studies are available to support analysis of the welfare implications of specific types of complex dynamics. For instance, Perrings and Pearce (1994) analyse the implications of ecological thresholds for biodiversity conservation strategies. Below, a very basic model for the algebraic optimisation of resource extraction is presented.

In a deterministic, dynamic, single-species model, the efficient stock and harvest level depend upon the marginal growth rate of the stock, and the discount rate used. The stock's marginal growth rate determines the rents that can be obtained from the natural capital stock, whereas the discount rate indicates the rents that can be obtained from depletion of the natural capital stock and investing the benefits in man-made capital. Clark (1976) assumed fixed harvest costs (i.e. harvest costs independent from the stock size) and showed that if the reproduction rate of the resource is lower than the discount rate, it may be efficient, from a utilitarian point of view, to harvest the full stock. This situation does not generally apply, as normally the harvest costs will increase with decreasing stock levels. Moreover, there may be a range of hidden costs related to overharvesting of particular species through the disturbance of the ecosystem, which may affect the whole range of ecosystem services supplied by the ecosystem (Jackson et al., 2001).

A mathematical approach can be used to calculate efficient harvest levels as a function of the costs and benefits of harvesting the resource. This requires, as a first step, formulation of an objective function that indicates the net benefits generated by ecosystem management options as a function of ecosystem dynamics, management, and values and prices of ecosystem services. For resource extraction, a general objective function can be expressed as:

$$J = \int_0^T e^{-rt}\{B(s_t) - C(s_t, S_t)\}\,dt \tag{3.2}$$

where $B(s_t)$ are the benefits and $C(s_t, S_t)$ are the harvesting costs related to the use of an ecosystem service, aggregated over a period T and using a discount factor e^{-rt} (and a discount rate r). The benefits depend upon the harvest level s_t only, whereas the harvest costs also depend upon the size of the stock S_t, which reflects that an increasing scarcity of the resource may increase the costs of harvesting, for instance in case of a fish stock.

In addition to specification of the objective function, the development of the stock needs to be modelled. For example, let $\theta(S)$ be the natural growth of the population, β the constant growth factor, and S_{max} the carrying capacity. In the case of a simple logistic growth model, the objective function is subject to:

$$dS/dt = \theta(S) - s_t\text{; and} \tag{3.3}$$

$$\theta(S) = \beta \cdot S_t \cdot \left(1 - \frac{S_t}{S_{\max}}\right) \tag{3.4}$$

The first of the two equations above indicates that the stock (S) changes over time as a function of natural reproduction θ(S) and the harvest of the resource (s_t). The second equation presents natural reproduction θ(S) as a function of the size of the stock (S_t) in relation to the carrying capacity of the environment for that stock (S_{max}), according to a logistic growth model.

As explained above, in order to find the management option (the resource harvest regime) that provides maximum benefits, both a simulation and an algebraic optimisation procedure can be followed. Algebraic optimisation is grounded in optimal control theory. In this approach, the objective function can be maximised using the Hamiltonian function (see Perman et al. (1999) for a general introduction and Chiang (1992) for a more in-depth theoretical background). The current value Hamiltonian, noted as H, can, for the case described above, be expressed as:

$$H_t = B(s_t) - C(s_t, S_t) + \lambda_t \cdot [\theta(S) - s_t], \tag{3.5}$$

with λ_t being the time-dependent co-state variable, representing the shadow price of one unit of stock of the resource at time t. The necessary conditions for a maximum are:

$$\partial H_t/\partial s_t = 0 = \partial B/\partial s_t - \partial C/\partial s_t - \lambda_t \tag{3.6}$$

$$\partial \lambda/\partial t = r \cdot \lambda_t - \partial H_t/\partial S_t \tag{3.7}$$

$$S(t_0) = S_0 \ (\textit{initial condition}) \tag{3.8}$$

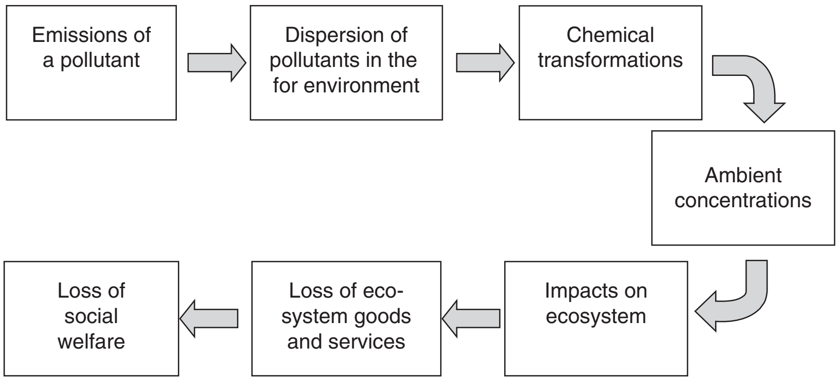
$$H(t_T) = 0 \ (\textit{transversality condition}) \tag{3.9}$$

Solving these conditions yields the efficient harvest rates at each point in time. The first condition essentially shows that at the point providing maximum net benefits the net (shadow) price will equal the gross price minus the marginal harvest costs. The second equation is the Hotelling efficient harvesting condition for a renewable resource in which harvesting costs depend upon stock level. When the Hotelling equation is satisfied, the rate of return the resource owner obtains from the harvest equals r, the rate of return that could be obtained by investment elsewhere in the economy.

The third equation indicates the initial conditions of the system to be optimised. And the fourth equation indicates the terminal conditions that apply at time T. The aim of the Hamiltonian is to identify optimal harvesting rates considering the size of the stock from time $t = 0$ to T, and this implies that the value of the resource at time T should be zero, either because the resource is depleted or because harvesting does not yield any further net benefits (if there is residual value, the harvesting regime can not be optimal). It is, of course, also possible to define alternative terminal conditions and pose alternative constraints to the optimisation process, for example the condition that a certain amount of stock needs to be preserved at time T.

Efficient levels of pollution control

The optimal level of pollution is usually discussed in terms of the intersection of the marginal damage function and the marginal control cost function (see Tietenberg, 2000). The marginal damage function shows the damage resulting from pollution as a function of emissions of a particular pollutant. The marginal control cost function shows the cost of reducing emissions of the pollutant below the level that would occur in an unregulated market economy. The marginal damage function is composed of a chain of functional relationships, as depicted in Figure 3.3. Dispersion processes and chemical transformations may reduce local pollution loads. In some types of ecosystems, time lags may play a role, for example if there are buffers in the ecosystem that absorb pollutants and release them once the input of pollutants has decreased (Carpenter et al., 1999).



Source: Adapted from Farmer et al. (2001).

Figure 3.3 Schematic overview of a marginal damage function

Ecological–economic modelling of pollution control requires analysis of four main elements: (1) the costs of pollution control; (2) the relation between dispersal of pollution and the build-up of pollution loads in the ecosystem; (3) the impact of pollution loads on the capacity of the ecosystem to provide goods and services; and (4) the benefits foregone as a result of a loss of ecosystem services. A general objective function is:

$$J = \int_0^\infty e^{-rt}\{B(P_t) - C(p_t)\}\,dt \tag{3.10}$$

Where the net benefits are a function of the total benefits (B) of the services provided by the ecosystem, which depend upon the pollution level (P) of the ecosystem, and the costs (C) of reducing the pollution inflow in the ecosystem (p). e^{-rt} is the discount factor. Assessing the efficient pollution level requires modelling of the relation between the pollution concentrations (P) and the pollution inflow (p). In its simplest form, this relation may be described by the following equation (see Carpenter et al., 1999):

$$dP/dt = p_t - \varphi P_t + f(P_t) \tag{3.11}$$

The factor p_t is the inflow of pollutants in the ecosystem under consideration. The segment φP_t expresses the removal of pollutants from the system by means of outflow, which depends on the concentration P_t and, in the case of a lake, the amount of water φ flowing out of the system. The function $f(P_t)$ indicates the behaviour of the pollutant in the ecosystem. It reflects breakdown or accumulation of the pollutant by ecological processes, as well as the buffering of the pollutant in different compartments in the ecosystem. Examples of relevant ecological processes are denitrification (for nitrogen pollution), immobilisation and uptake in the food chain. Decomposition rates are small for persistent pollutants and are generally higher for organic nitrogen, and phosphorous compared to inorganic pollutants. Functions reflecting buffering may be rather complex, for instance in the case of a pollutant in a lake buffered in the lake sediments, which may be influenced by pH, temperature, etc.

To analyse efficient pollution control levels in an algebraic manner rather than with a simulation approach, the objective function needs to be maximised using the current value Hamiltonian (see Chiang, 1992). The Hamiltonian, noted as H, can be expressed as:

$$H_t = B(P_t) - C(p_t) + \lambda_t\, [p_t - \varphi P_t + f(P_t)], \tag{3.12}$$

with λ_t being the time dependent co-state variable that can be interpreted as the shadow price of one unit of pollution. In line with the Hamiltonian defined above for efficient resource harvesting, the necessary conditions for a maximum are:

$$\partial H_t / \partial p_t = 0 = B \cdot \partial P_t / \partial p_t - \partial C / \partial p_t + \lambda_t \tag{3.13}$$

$$d\lambda / dt = r \cdot \lambda_t - \partial H_t / \partial P_t \tag{3.14}$$

$$P(t_0) = P_0 \; (\textit{initial condition}) \tag{3.15}$$

$$H(t_T) = 0 \; (\textit{transversality condition}) \tag{3.16}$$

Note that the functions φP_t and $f(P_t)$ are not a function of p_t, and are therefore zero when differentiated by p_t. At the point of efficient pollution control, the marginal abatement costs equal the marginal damage costs. A single point of maximum efficiency is found in the case of convex abatement and damage costs curves. Non-convexity may lead to the presence of several points of local maximum and minimum pollution control efficiency (e.g. Mäler et al., 2003). The case specified above is a simple, single-pollutant case. Examples of the optimisation of pollution control in the case of multiple pollutants are given in, among others, Schmieman (2001) and Brink (2003).

Efficient land cover change

The basis for analysing the economic value of land use was laid by David Ricardo in the early 1800s. His 'Law of Rent' states that the rent of a land use unit equals the economic advantage obtained by using the site in its most productive use, relative to the advantage obtained by using marginal (i.e. the best rent-free) land for the same purpose, given the same inputs of labour and capital (e.g. Van Kooten and Bulte, 2000). The Ricardian or Differential concept of rent was extended and modified in the mid-nineteenth century by Von Thünen, who elaborated on the spatial component of rent (Von Thünen, 1863). Von Thünen related rent to yields, production costs, prices and transportation costs from the field to the (urban) market, and explained that, given uniform physical production factors (soils, water availability, etc.), land nearest to a city will be used for growing vegetables, subsequently followed by concentric rings used for staple crops, pasture land and forest land. Hence, following the classical models, land rent depends on the quality of the land as well as the physical access (depending on distance and infrastructure) to markets. In a market economy, this is reflected in land prices, which will depend on land quality as well as proximity to markets. Market prices for land, either with regards to sale or lease, commonly only reflect the (current or potential) benefits that can be obtained from the land by the land owner, and not the positive or negative externalities generated by land use.

Following Figure 3.2, land cover change may have economic efficiency, sustainability and equity implications, at different scales (ranging from local stakeholders to global externalities). A modification of land cover will normally lead to a change in the capacity of the area to provide ecosystem services. The economic efficiency of changes in ecosystem service supply as a function of land cover change can be analysed with the following basic objective function:

$$J = \int_{t=0}^{\infty} e^{-rt} \left\{ \sum_{l=1}^{L} \sum_{s=1}^{S} (B_{ls} - C_{ls}) - K_l \right\} dt, \qquad (3.17)$$

where e^{-rt} is the discount factor, $l = 1$ to L is the different land cover units that each provide various ecosystem services (s), B_{ls} is the benefits of each service per land cover unit, and C_{ls} is the production costs of each service, as defined per land cover unit. K_l is the costs related to land use change, for example, related to converting a forest to cropland. This equation expresses that different land cover units produce different ecosystem services, and that production costs for each ecosystem service may vary per land use unit. It also covers the cost of land cover change.

Note that, in addition to economic, sustainability and equity criteria, land cover change will in most cases be subject to strong boundaries set by a number of other criteria. First, land cover change options will be subject to a range of boundary conditions set by the physical aspects of the ecosystem.

Soils, climate and (irrigation) water availability will restrict the potential options for land cover change because of the physical requirements of specific vegetation types. In addition, legal and customary criteria will normally apply and further limit possibilities for land use change. Other land uses and associated land cover types (e.g. recreation) require the presence of infrastructure for access. Finally, certain land cover types and/or ecosystem services supplies may be spatially incompatible, for instance, in the case of a highway and a recreational forest – the zone besides the highway will have a relatively low potential for ecotourism (see Willemen et al., 2008 for more details on how ecosystem services interact in the landscape).

Analysing the economic efficiency of land cover changes will normally require a GIS-based approach. Specifying spatial units, the biophysical characteristics including land cover of these units and the ecosystem services they supply. Finding the economic efficient land cover lay-out, accounting for the costs of land cover conversion and both private and public goods supplied by the landscape, will, except in very simple or stylised cases, not be possible with algebraic optimisation, in view of the large number of variables and relations involved. Hence, a general set-up of the Hamiltonian can not be proposed, and a simulation optimisation approach is the method of choice for comparing the efficiency of different land covers, and for identifying the economic efficient land cover option given a set of biophysical, economic and institutional constraints.

Efficient intervention in the ecosystem

Ecosystem intervention is the fourth and final category of ecosystem change, covering all residual types of change not covered in the previous categories. Interventions may either lead to rehabilitation of the ecosystem or, normally unintentionally, to ecosystem degradation. Examples of interventions in the ecosystem are human-controlled burning of dead wood in forests susceptible to fire, changing groundwater levels (e.g. through more or less drinking water extraction, or by changing water levels in rivers or canals), eradicating alien invasive species, etc. These interventions can also be designed to assist the ecosystem in adapting to external pressures such as climate change.

In view of the diversity of possible ecosystem interventions, the efficient level of ecosystem intervention can only be analysed in general terms in this section. If the evaluation concerns only one, discrete measure, the basic criterion, in terms of efficiency, is whether the discounted benefits of the measure exceed the discounted costs of the measure. Benefits include the potential impact of the measure on the supply of all relevant ecosystem services, and the costs include investment costs, operation and maintenance costs, and possible negative impacts on the supply of other ecosystem services.

In the case where a range of measures is possible, the efficient intervention level corresponds to implementation of those measures that minimise the sum of the total costs of the measures and the costs resulting from a loss of environmental quality. A loss of environmental quality may cause a loss of ecosystem services, and bring costs related to compensation payments to stakeholders impacted by that loss (Hueting, 1980). For concave benefit and convex cost functions, at the point of maximum efficiency, the marginal benefits of implementing the measure equal the marginal costs of adverse environmental quality (Tietenberg, 2000). In case of non-concave benefit, or non-convex cost functions, there may be several local maximums, which need to be compared in order to find the efficient management option. This is illustrated in Chapter 5 (in Figures 5.9 and 5.10, which present a non-concave benefit function).

3.4.2 Analysing the Sustainability of Ecosystem Management

A large number of studies and assessments have addressed the measurement of sustainability. As discussed in Section 2.1.3, in the approach laid out in this book, sustainability is considered in addition to economic efficiency and equity criteria. Therefore, a strong sustainability criterion is proposed, since there is a risk of overlap between weak sustainability criteria and economic efficiency. The section below, consequently, focuses on measuring strong sustainability at the level of the ecosystem, with sustainability defined as 'management that maintains the capacity of the ecosystem to provide future generations with the amount and type of ecosystem services at a level at least equal to the current capacity'. The analysis of sustainability requires two key conceptual steps: selecting a reference situation; and selecting sustainability indicators, as discussed below.

Selecting a reference situation

Sustainability involves preserving environmental qualities compared to a reference situation. It is usually not straightforward to select a reference situation, for which there are three basic options:

- **The present.** The Brundtland definition was formulated with a focus on assessing sustainability at coarse scales (e.g. at the national or global level). This allows for the degradation of some ecosystems if this is compensated by rehabilitation in other places (WCED, 1982). However, application of the concept at the scale of ecosystems becomes problematic since it is unclear whether national or global trends require rehabilitation of the particular ecosystem involved, or if there is room to allow degradation while maintaining national or global sustainability. The alternative is then to consider the ecosystem in isolation, and to assume that sustainability requires the maintenance of the qualities of the ecosystem compared to its present state. However, in this case, if the ecosystem is currently in a heavily degraded state due to recent ecosystem changes, it would, according to most commonly used definitions of sustainability including the one above, be sustainable to leave the ecosystem in its degraded state. This would be contrary to the general perception that restoring recently degraded ecosystems would contribute to sustainability.

- **A historical situation.** An alternative that would circumvent the risk described above is to compare sustainability with the ecosystem quality in a year in which the ecosystem has a desired environmental quality. For instance, for water quality in northwest European waterbodies, 1960 can be taken as a reference year; at this point in time, nutrient and agro-chemical pollution levels were generally low, water was relatively unpolluted, and biodiversity and potential to supply ecosystem services were high. In this case, restoring water quality to the 1960 level can be interpreted as environmentally sustainable. However, clearly, the choice of the reference year may be perceived as arbitrary. Since the large majority of the world's ecosystems has undergone gradual or rapid change as a function of human management during centuries or millennia, the selection of a reference year without human disturbance is generally not feasible.
- **Defining a reference situation based on ecosystem properties.** A different approach is to define sustainability at the ecosystem level on the basis of the properties of the ecosystem itself (e.g. its biodiversity, capacity to provide services, resilience, habitat for specific species, etc.). For instance, sustainable forest management could in specific cases be defined as forest management that conserves the species diversity of the forest, or the numbers and diversity of specific, highly threatened species. For example, remaining forest patches in Kalimantan tend to have orang-utan densities above their long-term carrying capacity because they serve as refuges for displaced orang-utan from nearby forests that have been converted to oilpalm plantations. In this case, sustainable management could be interpreted as management that support the forests in harbouring these orang-utan populations, for instance by reforesting degraded spots in the forest with trees that provide forage for these animals.

Hence, defining a reference situation to assess sustainability at the ecosystem level requires consideration of the properties and management history of the ecosystem, and will often require stakeholder involvement in order to select the appropriate reference basis. In addition, the selection of a reference situation needs to consider that there may be variation in ecosystem qualities between years. For instance, the productivity and species composition of semi-arid rangelands depends strongly on annual rainfall, and the reference situation needs either to correct for the impact of annual fluctuations or to use an average over a number of years to define a reference situation.

Selecting sustainability indicators

A large number of indicators for sustainability has been developed. For instance, national level indicators for measuring sustainability have been developed by the OECD (2003), European Environment Agency (2009) and various national governments, such as the UK (Defra, 2008). At the national level, sustainability indicator sets generally represent three levels of the DPSIR framework: Pressures (e.g. CO_2 emissions), State (e.g. forest cover) and Responses (e.g. share of the land area marked as protected area).

In view of the large variability in ecosystems, indicator sets for measuring sustainability at the scale of the ecosystem need to be fine-tuned to the characteristics of the ecosystem (type, uses, data availability, degree of disturbance, etc.). In general, sustainability indicators need to capture changes in the ecosystem over time, be scientifically robust, require minimum data collection and preferably be easy to comprehend by policy makers and/or the general public. In order to capture long-term changes in the ecosystem's capacity to supply ecosystem services, this will normally require a limited set of singular and or composite indicators. In order to be scientifically robust, the indicators need to cover the key aspects defining the quality of the ecosystem and the key drivers for ecosystem change. Potentially, at the scale of the ecosystem, there is scope to focus the indicators on measurement of the state of the ecosystem (rather than pressures and responses) – as sustainability, as defined above, relates particularly to the changes in the state of the ecosystem.

Since sustainability implies maintaining the ecosystem's capacity to supply ecosystem services, including the biodiversity conservation service, indicator sets need to capture state indicators that relate to ecosystem services supply. In order to limit the number of indicators to be measured, one or a few indicators may be selected from each of three different groups: (1) indicators related to key services supplied by the ecosystem, such as forest cover in the case of a forest used for timber harvesting; (2) indicators reflecting environmental quality and/or functioning of the ecosystem; and (3) indicators reflecting biodiversity, for instance population trends in flagship conservation species of an area.

In order to make information more easily available to stakeholders, aggregated sustainability indicators may be used. Aggregation relies on the reduction of multidimensional effects to a single unit, which may have specific units, such as land area, or may be an index value without specific units (see e.g. Hammond et al., 1991). Hence, aggregation formalises what is often done implicitly because, ultimately, when making a decision, the decision maker must go through a process of condensing information to make simple comparisons. Proponents of aggregate indices argue that it is better to make this process explicit through an aggregation function than relying on the implicit aggregation that inevitably happens using an indicator profile. However, it may be difficult to find a transparent and methodologically satisfactory way of aggregating variables, and care needs to be taken that aggregate indicators are not disturbed by interrelationships between individual variables (Jollands et al., 2003).

Hence, singular and/or aggregate indicators need to be defined for each ecosystem separately on the basis of an understanding of ecosystem dynamics and ecosystem service supply. The selected indicators should be influenced as little as possible by natural variations in the ecosystem. For instance, as elaborated in Chapter 6, rather than productivity which is strongly dependent on annual rainfall, the functioning of semi-arid ecosystems is better reflected in their rain-use efficiency, i.e. the capacity of the ecosystem to use rain for the production of biomass (expressed as kg/ha/mm or rainfall).

3.4.3 Equitable Ecosystem Management

Equitable ecosystem management has two main axes: stakeholder representation and involvement in designing and implementing ecosystem management strategies, and sharing the benefits generated by ecosystems under different forms of management. In line with the focus of this book on quantitative techniques, below an overview is provided of two basic approaches that can be used to measure income effects of environmental management at the level of the ecosystem. For more information on stakeholder representation in ecosystem management, see Pirot et al. (2000).

Lorenz curves and Gini coefficients

Income distribution is often measured by means of the Lorenz curve and the Gini coefficient. In the classical Lorenz curve, people are ranked by income, from low to high. The resulting information is plotted in a diagram, with the cumulative percentage of the number of people in a country on the horizontal axis, and the cumulative percentage of national income on the vertical axis. The closer the curve lies to the diagonal, the more equal the distribution. The Gini coefficient is obtained by dividing the surface between the curve and the diagonal with the surface under the diagonal. In this way, a value between 0 and 1 is obtained which indicates the inequality of the distribution. The Gini coefficient is 0 if everybody earns the same amount of money, and its value is 1 if all income is earned by only one person and the rest of the population has no income at all.

The Lorenz curve and Gini coefficient are usually applied at the level of the country, but they can also be applied at lower scales. For instance, Van der Veeren and Lorenz (2002) calculate the Gini coefficient related to the costs of nutrient abatement in the Rhine Basin. The Rhine basin is divided into seven regions, and the costs of nutrient abatement according to different strategies are calculated for each region. The assumption of Van der Veeren and Lorenz (2002) is that a more equal distribution of these costs over the regions increases the chance of the strategy being adopted by policy makers.

In another example, Adger et al. (1997) examine the income distribution impacts of privatisation and subsequent land use change from common property mangrove forest to privately owned aquaculture ponds, using, among others, Gini curves. Results of a household survey of coastal villages reveals that the poorer households, who exhibit greater reliance on mangrove resources, are most likely to experience negative livelihood impacts as a result of state appropriation of common resources. The results demonstrate how privatisation of the Vietnamese common property wetlands examined leads to increasing inequality within the local population.

Hence, comparison of Lorenz curves and Gini coefficients can demonstrate the income distribution impacts of different ecosystem management options. These measures can be used to show the level of inequality, and potential changes in equality, of different ecosystem management options, but – clearly – not the acceptable level of equality.

Adjusted cost–benefit analysis

Another approach to analyse equity of ecosystem management options is to apply CBA adjusted for the income impacts on different income groups. This is undertaken by adding distributional weights to the costs and benefits, estimated separately for different income groups (see Adger et al., 1997):

$$NPV = \int_{t=1}^{T} \sum_{i=1}^{I} a_i \cdot (B_{i,t} - C_{i,t}) \cdot e^{-rt} dt, \tag{3.18}$$

for all I income groups, where a_i = distribution weight for income group i. The weight $a_i = (\overline{Y}/Y_t)^{-\eta}$, where $\overline{Y}$ is the mean income of the total population and Y_i is the income of the ith income group. Both costs and benefits to groups with a lower than average income are weighted higher. The factor η represents the aversion to inequality as the marginal elasticity of income (Squire and Van der Tak, 1975). If η is 0, the weights are ignored and the equation collapses to the standard equation for CBA. For $\eta = 1$, unitary marginal income elasticity is assumed (Squire and Van der Tak, 1975). For instance, with $\eta = 1$, the incidence of costs and benefits to a group with a quarter of the average income would be four times that of the group with average income (Adger et al., 1997). It has been argued that it may be impossible to observe η in practice (Layard and Walters, 1994) so η is often used to demonstrate the implications of different degrees of aversion against income inequality. In addition, it is possible to calculate threshold values indicating which degree of aversion against inequality renders an ecosystem management option (in)efficient – for a certain discount rate. Rather than forcing decision makers to state their aversion to inequality, this allows decision makers to evaluate whether they believe the combined income and efficiency impacts are justifiable.

In principle, income group does not have to be the only criteria on the basis of which additional weights may be given to a certain group of people. For instance, the income of traditional resource users may be values higher than that of newcomers. In this case the NPV may be corrected for this – with the remaining complexity of finding an appropriate weight for the different groups. Alternatively, Pareto efficiency may be applied, i.e. traditional resource users should not be made worse off than under the current management.

3.5 UNCERTAINTY

Following the framework described in this chapter, efficiency, sustainability and equity aspects of ecosystem management can be analysed with a dynamic systems ecological–economic modelling approach. In view of the number of relations and variables involved, this type of modelling can be subject to significant levels of uncertainty. Key principal sources of uncertainty include (1) the input data used to describe the system; and (2) the equations and the structure of the model. For the analysis of the impact of uncertainty in *input data*, sensitivity analysis is probably the most widely applied method. Sensitivity analysis studies the influence of variations in model parameters and initial values on model outcomes, usually by applying statistical techniques and/or by running the model for a range of different values of the variables and parameters assumed to be most uncertain. For a description of a number of other approaches to analyse uncertainty in input data, see, for example, Rotmans and van Asselt (2001). However, clearly, analysis of the implications of variations in input data with sensitivity analysis does not indicate the likelihood that such variations occur, and the uncertainty in ecological–economic models may be hard to predict.

Regarding uncertainties in the *equations* and *set-up* of the model, particular issues are how to deal with potential threshold values, responses of the ecosystem to multiple drivers, and the relation between changes in the state of the ecosystem, and its capacity to supply ecosystem services. A principal approach that has been developed to deal with uncertainty in model structure is model validation. Toth (1995) proposed three routes for model validation: (1) check against historical records; (2) adopt models and codes from other modelling groups for conceptual verification; and (3) model inter-comparisons (see Toth, 1995; Van der Sluijs, 1997).

In analysing ecosystem–economy interactions, two aspects are particularly subject to uncertainty. These are thresholds in ecosystem dynamics and the values estimates for different ecosystem services. Thresholds in ecosystem dynamics are discussed in Section 3.3 and uncertainties related to ecosystem services valuation are specified in Section 2.4.6. Thresholds occur at all scales, ranging from the melting of the Greenland icecap to the thresholds guiding the bifurcations between clear and turbid water in shallow lakes. The implication of uncertainty in a threshold is that major differences in ecosystem responses can occur as a function of only minor changes in ecosystem management. Sensitivity analysis can assist in determining the impacts of different values of the threshold but, as above, in itself does not indicate the likelihood of specific threshold values.

Hence, as with any modelling study, uncertainty is a key concern. As much as possible uncertainty levels, and the implications of uncertainty, should be indicated, based on sensitivity analysis or verifications of (part of) the model with data series. For further information and examples of how to deal with uncertainty in ecosystem management, see Rastetter (1996) and Peterson et al. (2003).

4. Modelling the efficiency and sustainability of forest management

4.1 INTRODUCTION

This chapter shows how dynamic systems ecological–economic modelling, as presented in Chapter 3, can be applied to analyse the efficiency and sustainability of ecosystem management options. The case of a hillside forest supplying two ecosystem services, wood production and erosion control, is used as an example. In this chapter, a basic model of a hypothetical forest ecosystem is developed, and the model parameters are quantified on the basis of representative values based on literature. The forest model comprises two components: forest cover and topsoil. In order to obtain consistency between the parameters, as much as possible, values related to a US Douglas fir forest stand are used. The specific objectives of the chapter are to: (1) model the productivity of a forest, in two cases, a reversible and an irreversible response to high harvesting pressure; and (2) demonstrate the difference between efficient and sustainable rotation periods, and identify intermediate management options ensuring higher economic efficiency as well as long-term sustainability.

The ecological–economic model developed in this chapter presents a deviation from the Faustmann models that have often been used to optimise rotation periods (Tahvonen, 1991). Faustmann models are algebraic optimisation models, and the basic principle of this type of models is that, for the economic efficient rotation period, the marginal value of the growth of the timber stock equals the marginal opportunity costs of not harvesting. The opportunity costs depend upon the costs of capital and the interest foregone on the site value of the land (Faustmann, 1849; Brazee, 2001). Assuming perfect markets and perfect foresight, the model leads to a constant rotation period that maximizes the present value of forest land over an infinitely long time horizon (Samuelson, 1976). In the last decades, a large number of enhancements to the original Faustmann model have been developed. For instance, Hartman (1976) extended the Faustmann model with the flow of non-timber forest benefits, Van Kooten et al. (1995) incorporated the benefits of carbon dioxide storage in the model, and Creedy and Wurzbacher (2001) presented a Faustmann model for a forest ecosystem that provides three different ecosystem services (timber, water and carbon sequestration). However, none of these models explicitly analyses efficient management in the case of a forest that responds irreversibly to overharvesting and where the ecosystem manager does not necessarily want to maintain the forest stock in the long term.

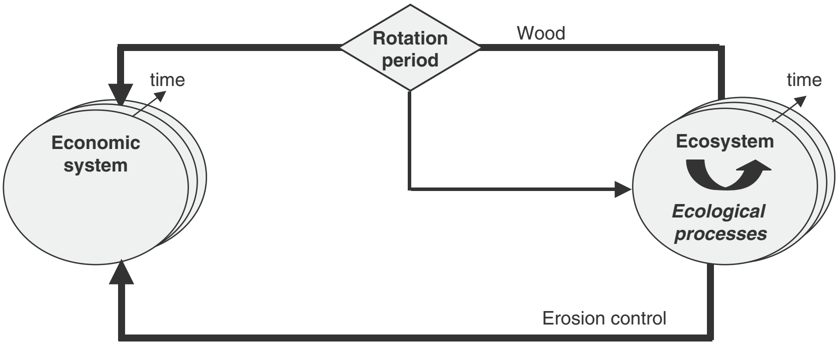
Since it is very complex to solve the Faustmann equations for a forest model including forest cover and topsoil and the various processes related to these two components (see Appendix 4.1), this chapter uses the dynamic systems modelling approach described in Chapter 3 in combination with simulation modelling (cf. Hein and Van Ierland, 2006). Two basic dynamic systems models of a hypothetical forest ecosystem are developed, one representing an ecosystem that responds reversibly to stress, and the other an ecosystem that responds irreversibly to stress. The ecosystems provide two services: wood and erosion control. The benefits of both services are considered in the calculation of the net present value (NPV) of the forest under different rotation periods. The control variable is the rotation period applied to harvest wood. Both fixed and variable rotation periods are examined.

The chapter is organised as follows. The two forestry models are developed in Section 4.2. The efficiency and sustainability of different felling rates are compared, for the two models, in Section 4.3. A discussion and the main conclusions are presented in Section 4.4. Note that, although an attempt is made to find representative values for the different components and variables of the forest ecosystem models, this case study deals with a hypothetical ecosystem. The case study is meant to demonstrate the Assessment Methodology described in Chapter 3, and does not yield efficient or sustainable rotation periods for a specific ecosystem.

4.2 DESCRIPTION OF THE ECOSYSTEM MODELS

4.2.1 The Modelling Framework

The modelling framework for this study is shown in Figure 4.1. The two models that are developed in the next paragraphs represent a hillside ecosystem that supplies two services: wood and erosion control. Erosion control is derived from the capacity of the forest to maintain soil cover and to prevent downstream deposition of sediments. Both models comprise two components: forest cover and topsoil. In the models, these two services are included in both physical (ton) and monetary units (US$). The ecosystems contain two processes, 'vegetation growth' and 'erosion', and the control variable is the rotation period. Wood extraction reduces the forest cover of the ecosystem, and a reduced forest cover leads to



Notes: Both services are first expressed in a physical unit (ton), which is then converted into a monetary unit (US$). The harvesting of wood reduces the forest standing stock, and represents a feedback in the ecosystem (thin arrow).

Figure 4.1 Modelling framework for a forest ecosystem supplying wood and erosion control

higher erosion rates. The models represent hypothetical ecosystems, with assumed values based upon literature.

The economic system is considered exogenous. It is assumed that, for a given, constant price for wood and the erosion control service, demand will not be saturated. Constant, average prices are used for the two ecosystem services included in the model. In order to ensure maximum possible consistency in the parameters, wood prices, costs of erosion and harvesting costs of wood are taken from the same economic setting. All costs and prices are derived from literature (LeDoux and Huyler, 2000; Uri and Lewis, 1998). Erosion costs are the average costs of water erosion in the USA, and wood prices and wood harvesting costs are also averages for the (Eastern) USA. In the models, a constant discount rate is assumed and the capital value of land is neglected (but see Penttinen, 2000 for more information on the implications of variable prices).

Based on the dynamic system modelling framework described above, two ecosystem models have been developed. The first model represents an ecosystem that responds *reversibly* to stress, the second model responds *irreversibly* to stress. In the first model, erosion is a function of forest cover and vegetation growth a function of standing biomass only. In the second model, a refinement is added. Regrowth of the forest cover now also depends upon the topsoil depth (e.g. because topsoil commonly contains a large part of the soil nutrients). There is no recovery if topsoil and forest cover are removed below a critical threshold. For simplicity, the models do not contain spatial heterogeneity and assume homogenous slopes, soils and species composition. They operate at the scale of the plot (30 by 30 m). The ecosystem's carrying capacity (maximum forest cover) is assumed to be 200 ton wood per hectare.

The models are used to analyse the efficiency and the sustainability of different rotation periods in the two forest types. *Efficient* ecosystem management is interpreted as management that maximises an objective function that includes the benefits of wood supply and the costs of erosion (based on Pearce and Moran, 2000; Turner et al., 2004). Future benefits and costs are discounted, using a fixed discount rate, in order to compare them with current benefits and costs. *Sustainable* ecosystem management is interpreted as management that maintains the capacity of the forest ecosystem to provide future generations with the amount and type of ecosystem services at a level at least equal to the current capacity. Biodiversity in the forest is not considered in this case study.

4.2.2 Ecosystem Model 1: Reversible Response to Stress

Model 1 represents a stylised model of a forest ecosystem that contains two components, described by two state variables. These are topsoil and forest cover, described respectively by topsoil depth (*TS*) and forest cover (*FC*). Topsoil depth has been identified as one of the key indicators for the assessment of the impact of erosion on the ecosystem (Cammeraat et al., 2002). The variable forest cover (*FC*) expresses both the standing biomass that can be harvested, and the soil cover that reduces erosion rates. The ecosystem provides wood (*W*) and erosion control, expressed (inversely) as the amount of sediments eroded (*E*). The amount of wood that can be harvested at a specific time depends upon the forest cover. The amount of erosion depends upon the forest cover and causes a reduction of the topsoil depth. The formulas used in the model to describe the development of the ecosystem are presented below.

Development of the topsoil (*TS*)

The topsoil decreases due to erosion, but recovers as a consequence of the accumulation of sediment and litter (analogous to Imeson and Cammeraat, 2002). Application of the model requires defining the initial value of the topsoil depth, which is assumed to be 3 cm.

$$TS_{(t)} = TS_{(t-1)} - E_{(t)} + RE_{(t)}, \tag{4.1}$$

where:
$TS_{(t)}$ = topsoil depth in metres at time t;
$E_{(t)}$ = erosion in metres at time t; and

$RE_{(t)}$ = recovery of the topsoil from erosion through accumulation of sediment and organic material in year t, expressed in metres.

Development of forest cover (*FC*)

The forest cover is expressed as percentage cover, compared to the carrying capacity of the plot. It is assumed that 100% cover represents a total biomass of 18 ton on the 30 by 30 m plot (which equals 200 ton/ha). Forest cover develops as a function of the harvest of wood, and the natural growth of the vegetation:

$$FC_{(t)} = FC_{(t-1)} - W_{(t)} + G_{(t)}, \tag{4.2}$$

where:
$FC_{(t)}$ = forest cover in % in year t;
$W_{(t)}$ = harvest of wood in year t, expressed in %-points; and
$G_{(t)}$ = growth of the forest cover of the ecosystem, expressed in %-points.

Wood harvest (*W*)

Wood is harvested with a certain rotation period, for example, once every 15 years. In line with the original Faustmann models, a fixed percentage of the standing wood stock is harvested with every felling. The variable forest cover (*FC*) represents both the standing biomass and the soil cover that reduces erosion, and it is assumed that, with every felling, 60% of the forest cover is harvested. This implies that, after a felling, 40% of the soil is left bare and is susceptible to erosion. Wood harvest is represented through the following formula:

$$W(t) = 0.6 \cdot FC(t) \mid \textit{once every R years}, \tag{4.3}$$

where:
$W(t)$ = wood harvest (%-points)
$FC(t)$ = forest cover (%-points)
R = rotation period (years)

Wood harvest and forest cover are expressed in %-points, and wood harvest is subsequently converted to ton wood on the basis that full forest cover represents 18 ton biomass for the 30 by 30 m plot.

Erosion (*E*)

The erosion control service is expressed through the amount of erosion (*E*) taking place. For a fixed slope, rainfall, and slope length, the relation between erosion and forest cover can be expressed as follows (cf. Nearing et al., 1989; Morgan, 1995):

$$E_{(t)} = \alpha \cdot e^{-2.5 \cdot FC(t)}, \tag{4.4}$$

where:
$FC_{(t)}$ = forest cover in % in year t; and
α = a constant for the ecosystem.

In line with Nearing et al. (1989) and Morgan (1995), an exponential relation is assumed between forest cover and erosion. The constant 2.5 is an empirical factor reported in Nearing et al. (1989). The impact of the logging activities themselves (e.g. through disturbance of the remaining vegetation, or compaction of the topsoil) is not further considered in this model.

Recovery from erosion (*RE*)

In the model, a gradual recovery of the topsoil takes place through the accumulation of sediment and plant litter, in metres/year. This process depends upon the sedimentation of soil particles by water or wind; and the deposition of organic material from standing biomass. Both processes vary substantially between different ecosystems, but each process is related to the forest cover. Forest cover reduces the speed of run-off and wind in the ecosystem, and causes deposition of sediments, and organic material is directly derived from (nearby) plants and trees (Morgan, 1995). A literature review did not reveal any formula describing the accumulation of sediments as a function of forest cover. Assuming that (1) the accumulation increases with forest cover; and (2) marginal increases will progressively diminish, the following logarithmic relation is assumed:

$$RE_{(t)} = \gamma \cdot FC_{(t)}{}^{\varphi}, \tag{4.5}$$

where γ and φ constants for a specific ecosystem ($\varphi < 1$), depending in particular on soil type are and climatic conditions.

Growth of the forest cover (*G*)

In ecosystem 1, growth of the forest cover depends upon standing forest cover in relation to the ecosystem's carrying capacity. It follows a simple logistic growth pattern (Pielou, 1969; Clark, 1976). The formula describing net growth of the vegetation cover is:

$$G_{(t)} = r_{max} \cdot FC_{(t)} \cdot (1 - FC_{(t)}/K), \tag{4.6a}$$

where r_{max} represents the maximum relative regrowth rate and K the carrying capacity of the ecosystem for vegetation (which equals a 100% cover).

Table 4.1 Parameters used for the valuation of the ecosystem services

Parameter	Value	Source
Price of wood	US$ 197/ton wood	Average export price of Douglas fir in 1997, Western hemlock and other softwoods exported from Washington, Oregon, northern California and Alaska (IMF, 2003)
Harvesting costs	US$ 27/ton wood	Average marginal costs of three different harvesting methods for Northeastern US forests (LeDoux and Huyler, 2000)
Costs of erosion	US$ 16/ton eroded soil	Estimated average off-site costs of sheet and rill erosion in the US in 1997 (Uri and Lewis, 1998)

Calculation of the net present value on the basis of the supply of ecosystem services

The ecosystem supplies two services: wood and erosion control. The following formula is used to calculate the net present value (NPV) of the services supplied by the ecosystem, over a 100-year period:

$$NPV = \sum_{t=1}^{100} (p_w . W_{(t)} - c_e . E_{(t)})\, \delta^t, \tag{4.7}$$

where:
p_w = net price of wood: the price of wood minus the extraction costs (US$/ton wood)
$W_{(t)}$ = the amount of wood harvested (ton wood)
c_e = the costs of erosion (US$/ton eroded soil)
$E_{(t)}$ = the amount of erosion (ton eroded soil)
δ^t = the discount factor

The values assumed for the parameters in equation (4.7) are shown in Table 4.1. For simplicity, the net price of wood and the costs of erosion are average, constant values derived from the literature. The NPV of different rotation periods is calculated for discount rates of 2.5% and 5%.

Overview of the variables and parameters used in the model

An overview of the *variables* used in the model is presented in Table 4.2. The various *parameters* used in the model are shown in Table 4.3. The parameters have been selected in such a way that potentially realistic erosion, sedimentation and regrowth rates are obtained (as explained in Table 4.3).

Table 4.2 Variables used in the ecosystem model

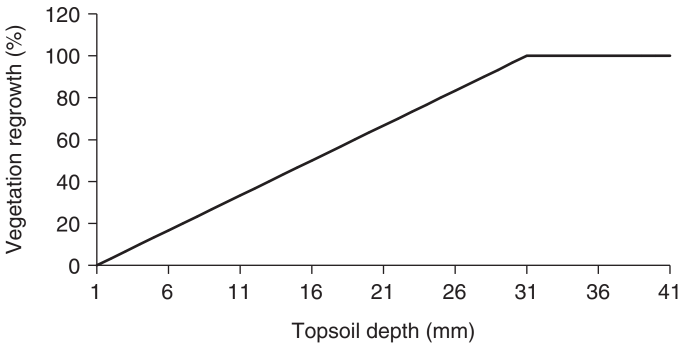
Variable	Type	Units	Notation
Topsoil depth	State	millimetres	*TS*
Forest cover	State	percentage-points	*FC*
Erosion	Process	millimetres	*E*
Wood harvest	Pressure	percentage-points	*W*
Rotation period	Control	years	*R*
Recovery from erosion	Process	millimetres	*RE*
Growth of the forest cover	Process	percentage-points	*G*

Table 4.3 Parameters of the ecosystem model

Parameter	Value	Unit	Comments
Initial topsoil depth	30	millimetre	
Initial forest cover	100	% cover	
α	3	millimetre	With α = 3 mm, erosion varies from around 0.3 mm/year (5 ton/ha) for full forest cover to around 3 mm/year (60 ton/ha) for bare soil – in line with for example Zanchi (1983) in Morgan (1995).
γ	0.1	millimetre	With γ = 0.1 mm and φ = 0.5, the recovery from erosion varies between 1 mm/year for total forest cover to 0 mm/year for zero forest cover.
φ	0.5	–	
r_{max}	0.1	–	This equals a maximum annual regrowth of 2.5 percentage points, reached at a forest cover of 50%.
K	100	% cover	K represents full forest cover.

4.2.3 Ecosystem Model 2: Irreversible Response to Stress

Ecosystem model 2 is equal to ecosystem model 1, with one refinement. It is assumed that a degradation of the topsoil reduces the regrowth capacity of the forest cover, for example, because a degraded topsoil prohibits the



Source: Hein and Van Ierland (2006).

Figure 4.2 Relation between topsoil depth and vegetation regrowth

establishment of new seedlings and because a less fertile soil profile leads to reduced growth of the trees (e.g. FAO, 1992; Williamson and Nielsen, 2003). Smith et al. (2000) found, within certain boundaries, a linear relation between fertility status of the soil (expressed as the C:N ratio) and the growth of pine seedlings (*Pinus radiata*): a C:N ration of 55 reduced growth of seedlings with 10 to 30% as compared to a C:N ratio of 20. Based upon Smith et al. (2000) and Williamson and Nielsen (2003), a linear relation between topsoil depth and forest regrowth is assumed up to a topsoil depth of 30 mm. For a topsoil of 30 mm or more, it is assumed that there is no further change in forest growth. In the model, it is also assumed that the absence of topsoil leads to a complete stop of the regrowth of the vegetation (see Figure 4.2). This alters equation (4.6a) to equation (4.6b), as described below. All other equations remain the same.

Growth of the forest cover ($G_{(t)}$)

As explained above, the growth of the forest cover in ecosystem 2 depends upon both standing forest cover and the soil conditions. Growth of the forest cover proceeds according to a logistic growth curve, whereas a linear relation between topsoil depth and growth is assumed – up to a topsoil depth of 30 mm (Figure 4.2):

$$G_{(t)} = r_{max} \cdot FC_{(t)} \cdot (1 - FC_{(t)} / K) \cdot Min(1; 0.0333 \cdot TS_{(t)}) \qquad (4.6b)$$

where r_{max} represents the maximum relative growth rate and K the carrying capacity. $FC_{(t)}$ is the forest cover, and $TS_{(t)}$ the topsoil depth in millimetres.

4.3 EFFICIENT AND SUSTAINABLE MANAGEMENT OPTIONS FOR THE MODELLED ECOSYSTEMS

4.3.1 Ecosystem Model 1: Reversible Response to Stress

The two models, each representing a different type of forest dynamics, are used to calculate the economic efficiency and sustainability of three types of management. The three examined ecosystem management options are: (1) a fixed rotation period that provides maximum economic benefits given the specified objective function; (2) the shortest rotation period that qualifies as sustainable; and (3) an alternative management option that provides a compromise between economic efficient and sustainable forest management. The model is run for a 100-year period. Both ecosystem models start with full forest cover (100%) in year 0, and harvest starts in year 1.

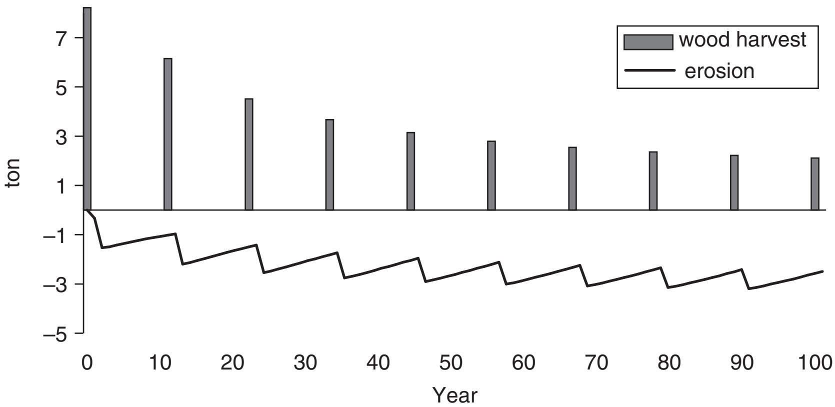
For ecosystem 1, the alternative management option entails a period of intensive harvesting, followed by a period of recovery. As degradation of the ecosystem is reversible, recovery of the ecosystem is not constrained by total depletion of the forest cover and topsoil layer. The most efficient rotation scheme is calculated, based upon a period of intensive harvesting and a recovery period, that qualifies as sustainable over a 100-year period.

The efficient rotation period

The model was run for a range of rotation periods, and the generated NPV was calculated for each felling rate, considering both the benefits of wood supply, and the costs of erosion. These calculations show that, at a 5% discount rate, an 11-year rotation period generates the maximum NPV. This maximum NPV is US$ 845, for the 30 by 30 m plot. If a discount rate of 2.5% is used, the optimal felling rate is 16 years, and the corresponding NPV is US$ 1604. This difference reflects that the NPV of the long-term benefits of an ecosystem is much higher if a low discount rate is used. The supply of ecosystem services for an 11-year rotation period is presented in Figure 4.3, and the development of the forest cover and the topsoil is shown in Figure 4.4. It is assumed that, once the topsoil is depleted, erosion of the subsoil will continue with the same erosion rate.

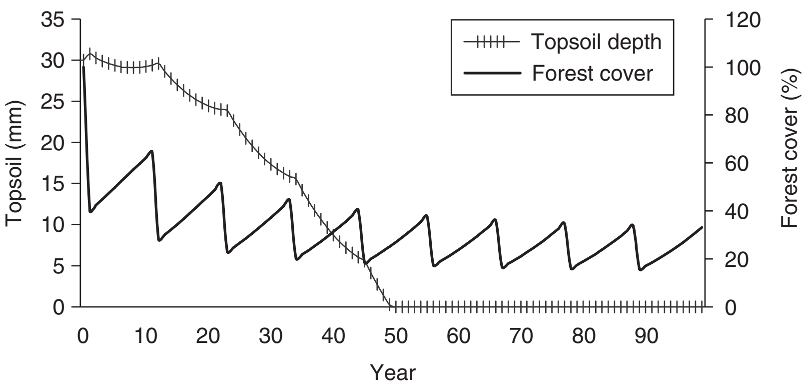
The sustainable rotation period

Clearly, the 11-year felling cycle, that generates maximum NPV at a 5% discount rate, is not sustainable. With this rotation period, topsoil is depleted in year 50, and forest cover is gradually reduced to around 35% in year 100. Because the loss of topsoil does not reduce vegetation regrowth in



Source: Hein and Van Ierland (2006).

Figure 4.3 Supply of two ecosystem services, wood and erosion control, at a felling rate of 11 years

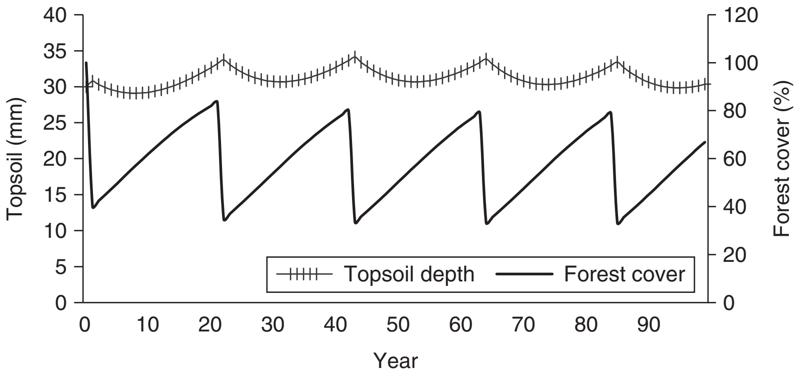


Source: Hein and Van Ierland (2006).

Figure 4.4 Development of the forest cover and topsoil for an 11-year felling cycle

ecosystem 1, regrowth of the forest cover continues even when the topsoil is totally eroded. With the 16-year rotation period, efficient in the case where the 2.5% discount rate is used, the topsoil is reduced to around 5 mm in year 100, and, hence, this management option is not sustainable either.

Ecosystem model 1 has also been run to reveal the most efficient,



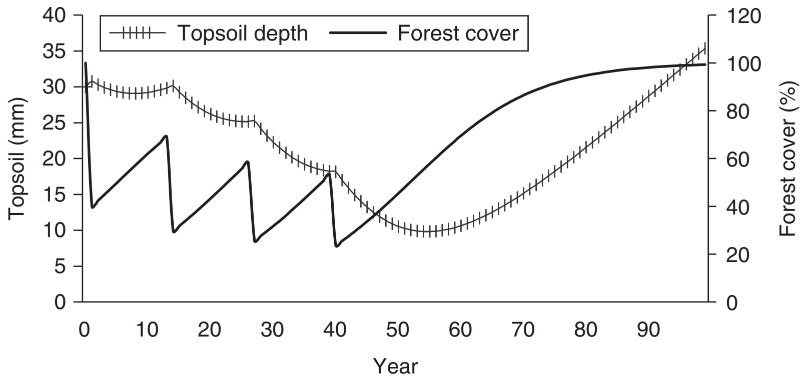
Source: Hein and Van Ierland (2006).

Figure 4.5 Development of the topsoil and forest cover under a sustainable, fixed rotation period of 21 years

sustainable rotation period. The shortest, and most efficient, rotation period that maintains the topsoil at around 3 mm, and allows the forest cover to recuperate following each felling cycle, is calculated to be 21 years. Figure 4.5 shows the development of the two ecosystem components under this form of management. At a 5% discount rate, this generates a NPV of US$ 478, and at 2.5% discount rate the NPV is US$ 1297.

Intensive harvesting followed by a recovery period

For ecosystem model 1, it is also possible to achieve sustainability, over a 100-year period, by intensive wood harvest in the first years, followed by a period of recovery. During the period of intensive wood harvesting, a fixed rotation period is used. Using this management approach, at a 5% discount rate, the optimal rotation period is 13 years during the first 50 years (enabling 4 felling cycles), followed by a recovery period of 50 years. The corresponding development of the ecosystem is shown in Figure 4.6. The resulting NPV is US$ 692 for the plot. This is some 20% less compared to the maximum NPV achieved at a felling cycle of 11 years during 100 years – but some 40% more compared to management based upon a sustainable, fixed rotation period. The use of a variable rotation period is most attractive for high discount rates. At a 2.5% discount rate, the benefits of this approach are small – US$ 1373 compared to US$ 1297 for the sustainable, fixed rotation period. The results for ecosystem 1 are summarised in Table 4.4.

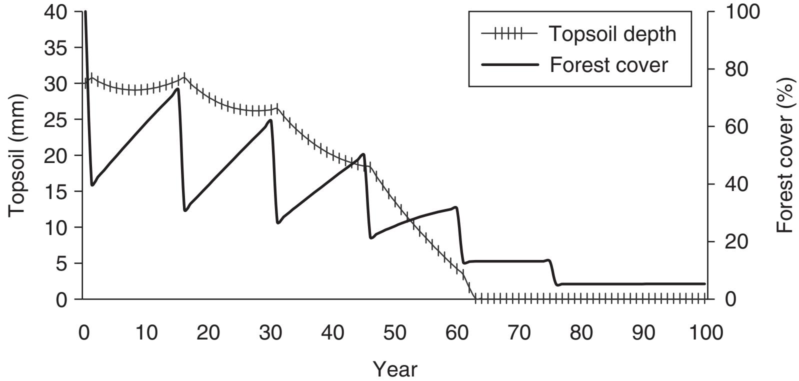


Source: Hein and Van Ierland (2006).

Figure 4.6 Sustainable management with a variable rotation period

Table 4.4 Efficient versus sustainable management of ecosystem 1

Management strategy	**Felling cycle (years)**	**NPV (US$)**	**Felling cycle (years)**	**NPV (US$)**	**Development of the topsoil and vegetation**
	Discount rate = 5%		**Discount rate = 2.5%**		
Profit maximisation	11	845	16	1604	Depletion of topsoil and forest cover
Sustainable management: no harvest until forest cover and topsoil are fully recovered from the previous felling	21	478	21	1297	Dynamic stabilisation of the topsoil and vegetation
Long-term sustainable management: intensive felling during the first period, no wood harvest in the remaining years	13 (during the first 50 years)	692	18 (during the first 80 years)	1373	Full recovery of topsoil and vegetation in year 100



Source: Hein and Van Ierland (2006).

Figure 4.7 Development of the topsoil and vegetation cover of ecosystem 2, for a 15-year felling cycle

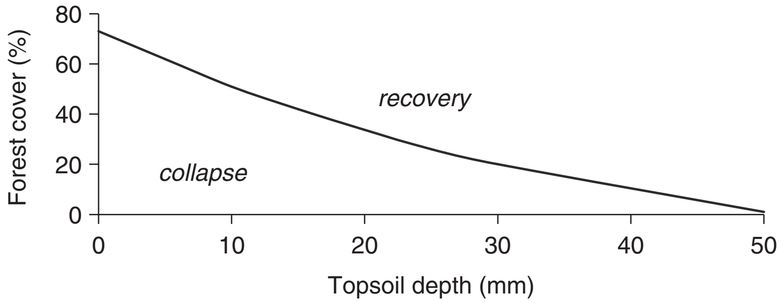
4.3.2 Ecosystem Model 2: Irreversible Response to Stress

The efficient rotation period

Compared to ecosystem model 1, model 2 contains an extra feedback that reduces the growth of the forest cover at low topsoil depths. For this model 2, the economic efficient rotation period comprises one felling per 15 years, resulting in an NPV of US$585 at a 5% discount rate. The corresponding development of the topsoil and forest cover is shown in Figure 4.7. For a discount rate of 2.5%, the optimal felling rate would be 19 years, resulting in an NPV of US$1349. The additional feedback in model 2 causes a slower recovery of the vegetation from wood harvest, which leads to a lower NPV for the different rotation periods.

The sustainable rotation period

As with ecosystem 1, the efficient rotation period is not sustainable. At the 15-year rotation period, the topsoil is depleted in year 63, and the forest cover in year 77. The 19-year rotation period, efficient if a discount rate of 2.5% is used, also leads to an, albeit slower, depletion of the topsoil and forest cover. For this ecosystem, the most efficient sustainable felling rate would be 21 years, with an NPV of US$475 at a 5% discount rate, and US$1291 at a 2.5% discount rate.



Notes: At values below the curve, the topsoil and forest cover of the ecosystem will irreversibly collapse to 0.

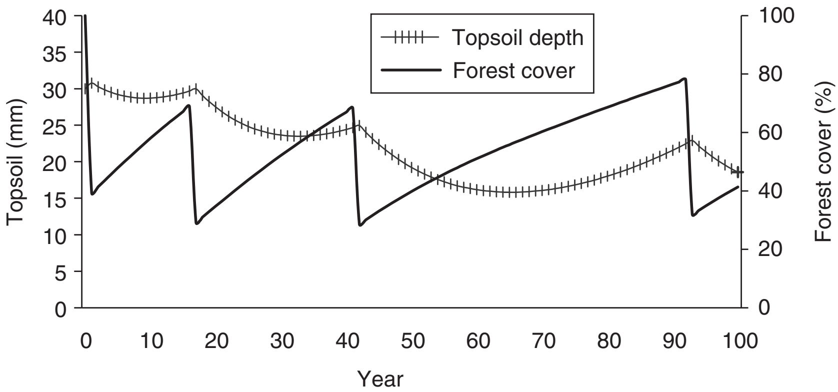
Source: Hein and Van Ierland (2006).

Figure 4.8 Minimum sustainable stock levels for ecosystem 2

Profit maximising while maintaining the minimum sustainable stock levels

For ecosystem 2, the strategy of clear-felling followed by a period of recovery is not suitable – recovery will not take place once the topsoil and forest cover have been depleted. However, an approach may be followed with intensive harvesting subject to maintaining the minimum sustainable stock. The minimum sustainable stock reflects the combinations of topsoil depth and forest cover that need to be maintained to allow recovery of the system. These combinations have been calculated with the ecosystem model, and are shown in Figure 4.8. In order to calculate the efficiency of profit maximisation subject to the condition that forest cover and/or topsoil are not depleted to below the minimum sustainable stock levels, ecosystem model 2 has been expanded with a simple if/then routine. Basically, this routine states that no harvesting is allowed if this would reduce the forest cover and the topsoil to below the minimum sustainable stock level.

Subsequently, the most efficient rotation scheme that does not deplete forest cover and topsoil levels to below the minimum sustainable stock level has been calculated. The resulting rotation scheme is shown in Figure 4.9. After a first harvest in year 1, the second harvest takes place in year 18, followed by a third harvest in year 40 and a fourth harvest in year 92. The corresponding NPVs are US$ 572 for a discount rate of 5%, and US$ 1308 for a discount rate of 2.5%. Both discount rates lead to the same optimal rotation scheme. The results for ecosystem 2 are summarised in Table 4.5.



Source: Hein and Van Ierland (2006).

Figure 4.9 Harvesting of forest ecosystem 2, subject to maintaining the minimum sustainable stocks

Table 4.5 Efficient versus sustainable management of ecosystem 2

Management strategy	**Felling cycle (years)**	**NPV (US$)**	**Felling cycle (years)**	**NPV (US$)**	**Development of the topsoil and vegetation**
	Discount rate = 5%		**Discount rate = 2.5%**		
Profit maximisation	15	585	19	1349	Depletion of topsoil and forest cover
Sustainable management	21	475	21	1291	Dynamic stabilisation of topsoil and forest cover
Maintaining the minimum sustainable stock	variable	572	variable	1308	Maintaining the capacity of the ecosystem to recover

4.4 DISCUSSION AND CONCLUSIONS

4.4.1 Uncertainties in the Model

Clearly, the identified efficient and sustainable rotation periods are very short compared to those rotation periods actually applied in temperate forest stands. Nevertheless, provided that the models are calibrated appropriately for a specific ecosystem, they allow the calculation of the efficiency and sustainability of rotation periods for a forest stand. Compared to the Faustmann models (Faustmann, 1849; Brazee, 2001), the models characterise the ecosystem through two, instead of one, ecosystem component: forest cover and topsoil. This allows a more refined modelling of the response of the ecosystem to harvesting: the development of the topsoil influences the establishment of forest seedlings and the regrowth of the forest stand (cf. FAO, 1992).

Incorporating two ecosystem components and their interactions in the model allows for the simulation of irreversible ecosystem dynamics. Hence, with a relatively simple modelling approach, irreversible responses and thresholds follow from the model, rather than having to be exogenously forced on the model. Although the models provide enhanced ecological accuracy compared to the original Faustmann models, a number of important simplifications have been made in the models, including:

1. Tree biomass and vegetation cover are expressed through one parameter: percentage forest cover. This is accounted for by adapting the parameters used to model the capacities of the ecosystem to supply wood and control erosion as a function of the forest cover.
2. The impact of logging itself is not considered. In reality, logging activities may be a major cause of erosion because they cause disturbance of the vegetation layer and compaction of the soil (e.g. Croke et al., 2001).
3. All logging costs are considered variable costs in the model – the average logging costs are independent of the amount of wood harvested. This is one of the factors that leads to the relatively short efficient rotation periods found in this hypothetical case study. In reality, logging companies also have to consider the fixed costs of harvesting a particular site.
4. The model assumes constant prices for the two ecosystem services involved.

If the stylised model is applied to a specific forest ecosystem, these four factors would need to be addressed. In particular, the relation between forest cover and erosion would need to be specified as a function of standing forest biomass, the impacts of logging would have to be included, and variable and fixed costs of logging would need to be considered.

4.4.2 Implications for Ecosystem Management

In general, the inclusion of the full set of ecosystem services in the assessment of management options is required in order to obtain a correct picture of the private and societal benefits supplied by the ecosystem under various management regimes (Turner et al., 2003). In the case study, the models show that intensive harvesting increases erosion of the plots. This reduces the growth of the forest stock, and brings additional costs through sedimentation of rivers and reservoirs downstream. Hence, consideration of the topsoil component and erosion results in a longer efficient rotation period as compared to the optimal period that would be identified on the basis of the original Faustmann model (cf. Creedy and Wurzbacher, 2001).

The analysis also illustrates the large discrepancies that may occur between efficient and sustainable management of an ecosystem. Indeed, such discrepancies frequently occur in decision making on ecosystems (e.g. Munasinghe and McNeely, 1994). Clearly, the selected discount rate plays a major role (see also Hueting, 1991; Freeman, 1993; Howarth and Norgaard, 1993; Hanley, 1999). Because the erosion control service is included in the model, the most efficient rotation period is longer, and somewhat more sustainable, compared to a model in which only the wood supply service of the forest is considered. However, inclusion of all relevant services may not be enough to reconcile efficient and sustainable management (Hueting, 1980). In addition, even if the total benefits including all relevant ecosystem services provided by a privately owned ecosystem are larger for the ecosystem in its natural state, private land owners may still prefer land conversion or clear-felling if they are not rewarded for the public services supplied by the ecosystem (Kishor and Constantino, 1993). In this case, ecosystem valuation may be used to underpin the transfer of funds from stakeholders benefiting from ecosystem services to the stakeholders maintaining the ecosystem (e.g. Chomitz et al., 1998).

The case study also shows two alternative management options in addition to pursuing the efficient and the sustainable management option. Application of variable rotation periods allows partial reconciliation of efficiency and sustainability considerations. If an ecosystem responds reversibly to stress, variable rotation periods provide the possibility to exploit the ecosystem intensively during an initial period, and allow it to recover in the subsequent years. When sustainability is evaluated over a specific, long-term period, this exploitation can be seen as sustainable if the final topsoil and forest cover are not lower than the initial values. This option is particularly favourable if a relatively high discount rate is used. Obviously, one of the consequences is that there will be no income during the recovery period – but the lack of income from the ecosystem in this period may be compensated by income generated from investments made during the first period (e.g. Pezzey and Toman, 2002).

Partial reconciliation of efficiency and sustainability through intensive harvesting in an initial period, followed by a rest period, is not possible in the case where the ecosystem responds irreversibly to stress. Once the topsoil and vegetation have been depleted to below the minimum sustainable stock levels, the system will no longer recover. An alternative option for the irreversible ecosystem is to optimise harvest rates subject to maintaining the minimum sustainable stock levels. This offers a compromise solution between efficient and sustainable exploitation of the ecosystem. This option is more profitable than pursuing sustainable management through a fixed rotation period, particularly if high discount rates are used. Although this option is not sustainable in the sense that there is a gradual decline in the natural capital stock, it avoids an irreversible collapse of the system. It leaves future generations the option to fully recuperate the ecosystem by temporarily reducing the harvest levels (at the cost of not harvesting during a certain period).

The occurrence of irreversible responses to management is of particular importance in the case where variable rotation periods are modelled. Once the minimum sustainable stock of the ecosystem is surpassed, a reduction of the rotation period does not lead to recovery of the system. In the model of the irreversible ecosystem (2), the development of the two components is mutually dependent, and the minimum sustainable stock is expressed through both state indicators (topsoil depth and forest cover). Hence, for ecosystems with interconnected, mutually dependent components (which will often be the case, e.g. Levin, 1992; Mooney et al., 1995), the threshold between collapse and recovery may depend upon a combination of indicators. If this is the case, consideration of only one indicator (for example, forest cover alone) will provide inaccurate information on the state of the ecosystem and the risk of potential collapses of the system.

In the deterministic model developed in this chapter, the minimum sustainable stock levels were known with certainty. This will usually not be the case for real-world ecosystems (Cole, 1954; Smith, 1990). In the case of uncertainty in ecosystem behaviour, the concept that indicates the minimum ecosystem stock that should be preserved in order to maintain the functioning of the ecosystem is the 'safe minimum standard' (SMS). Ciriacy-Wantrup (1968) proposed a 'safe minimum standard of conservation' as a means of incorporating uncertainty and irreversibility in the appraisal of natural resource utilisation. The safe minimum standard concept was later modified by Bishop (1978), who stated that irreversible environmental loss should be avoided unless this bears 'unacceptable' social costs. The SMS threshold increases with higher uncertainty (see also Randall and Farmer, 1995). This case study shows how the minimum sustainable stock, and hence the SMS, may depend upon two ecosystem state indicators and their interactions, and that measuring the SMS with regards to only one or few state indicators may not be accurate. Hence, defining an indicator and a threshold value for the SMS needs to be based on a thorough understanding of the dynamics of the ecosystem, and may need to include a set of interrelated indicators rather than one singular indicator.

4.4.3 Conclusions

This chapter shows how dynamic systems ecological–economic modelling can be applied to assess the economic efficiency and sustainability of different ecosystem management options. A stylised model of a hillside forest stand providing wood and erosion control is used as a case study. The model is not calibrated for a specific forest, and the retrieved efficient and sustainable rotation periods are short compared to those rotation periods actually applied in temperate forests. Nevertheless, the study highlights some of the potential strengths and challenges related to ecological–economic modelling. Among the strengths of the approach is the flexibility to apply the models to include a wide range of ecosystem components and processes, and a wide range of ecosystem services. Another strength is the possibility of including complex dynamics, such as irreversible responses to stress. These dynamics are, in the case of this study, not imposed on the system through an exogenous threshold, but result from the modelled interactions between the ecosystem components. Furthermore, by specifying variables and parameters for different grids, and modelling interactions between grids (for instance slope and erosion rates, transport of sediment between grids), the models may be integrated in a spatial model. Potential weaknesses are the need to come to a detailed understanding of the processes driving the ecosystem and the relatively high data requirement if the modelling approach is applied to a real world ecosystem.

APPENDIX 4.1 FAUSTMANN EFFICIENCY CONDITIONS FOR A FOREST SUPPLYING TWO ECOSYSTEM SERVICES

For a hillside forest ecosystem that supplies two ecosystem services, wood and erosion control, the Faustmann efficiency condition needs to be expanded to reflect the supply of both services. The expanded efficiency condition is as follows:

$$p_w \cdot dFC/dR + c_e \cdot \Delta E = i \cdot p_w \cdot FC + i \cdot \Pi$$

where:
p_w = net price of timber (the price of wood minus the harvest costs) (US$/ton wood)
FC = forest cover, representing the standing stock of wood (converted to ton wood).
R = rotation period (years)
c_e = costs of erosion (US$/ton soil)
ΔE = increase of erosion in the case of logging (ton soil)
i = discount rate
Π = the site value of the land on which the forest is located.

On the left-hand side of the equation are the net marginal benefits, which consist of the marginal benefits of wood harvesting plus the marginal benefits of the erosion control service (which equals the costs of erosion times the marginal amount of erosion avoided by delaying logging). On the right-hand side are the marginal opportunity costs of not harvesting, which consist of the marginal opportunity costs of the capital that could be gained through harvesting and the marginal costs of the land on which the forest is located.

In the ecosystem models developed in this chapter, it is assumed that $\Pi = 0$. The efficiency equation becomes:

$$p_w \cdot r_{max} \cdot FC\,(1 - FC/K) + c_e \cdot (\alpha.e^{-2.5*FC2} - \alpha.e^{-2.5 \cdot FC}) - i \cdot p_w .\ FC = 0$$

where:
p_w = net price of timber (the price of wood minus the harvest costs) (US$/ton wood)
r_{max} = maximum relative regrowth rate
FC = forest cover before harvesting (%).
$FC2$ = the forest cover after harvesting (%)

K = carrying capacity
c_e = costs of erosion (US$/ton soil)
α = a parameter, set at 3 mm in the models
i = discount rate.

5. Case study: eutrophication control in the De Wieden wetland, the Netherlands

5.1 INTRODUCTION

This chapter aims to apply the ecological–economic modelling approach described in Chapter 3 to a real-world ecosystem. The framework is used to construct an ecological–economic model to compare the costs and benefits of eutrophication control measures in the lakes of De Wieden, the Netherlands. Eutrophication of lakes is caused by the inflow of nutrients, in particular nitrogen and phosphorus, that are released from agricultural or urban sources. Eutrophication often leads to a reduction in the supply of ecosystem services. For instance, it may affect recreation, fisheries or nature conservation in and around the water body (Carpenter et al., 1999; Mäler, 2000). The response of a freshwater lake to changes in nutrient loading is generally subject to multiple states and thresholds. These multiple states are determined by different factors in deep and shallow lakes. In deep lakes, a critical aspect of the lake ecosystem dynamics is whether the deep part of the lake is in an aerobic or anaerobic condition. In shallow lakes, different states are characterised by different lake visibility and different plant and fish communities (Timms and Moss, 1984; Scheffer, 1998). This case study focusses on a shallow lake ecosystem, accounting for the complex dynamics of this ecosystem.

The identification of efficient eutrophication control strategies involves the comparison of the costs and benefits of eutrophication control measures. The costs relate to the investment, operation and maintenance of pollution control equipment, the benefits to an increased supply of ecosystem services following reduced eutrophication. A crucial factor to be considered in eutrophication control are steady states and thresholds in the ecosystem, because these have an overriding impact on the lake's response to (reductions in) nutrient loading. Temperate shallow lakes are, in general, in a clear-water state with abundant water plants and a fish community dominated by piscivorous fish at low nutrient levels. At high nutrient levels, shallow lakes tend to convert to a turbid state dominated by phytoplankton and benthivorous fish (Jeppesen et al., 1990; Scheffer, 1998; Van Nes et al., 2002). The change between the clear and turbid water states is normally abrupt, and proceeds at a certain threshold in nutrient concentrations. This threshold is specific for each lake (Jeppesen et al., 1990).

Although a range of theoretical models have been developed to identify optimum eutrophication control strategies (e.g. Nævdal, 2001; Brock and Starrett, 2003; Mäler et al., 2003), there are few studies that determine optimum eutrophication control strategies for specific waterbodies (but see e.g. Wulff et al., 2001 for a study on eutrophication control in the Baltic Sea). In order to identify an efficient eutrophication control strategy for an individual lake, it is necessary to analyse the supply of ecosystem services of the lake, the costs of eutrophication control measures and the response of the lake to reduced nutrient loading, including the threshold involved. The challenge lies in combining the above three issues into one ecological–economic model, and to calibrate and apply the model using water quality data that is measured in a monitoring program. From a policy perspective, such studies can support local authorities with concrete advice on eutrophication management of specific ecosystems.

The aim of this chapter is to identify efficient and sustainable eutrophication control strategies for 'De Wieden', a Dutch shallow lake ecosystem (see also Hein, 2006a). The lake is currently in a eutrophic, turbid state but, because of its major importance for biodiversity conservation and recreation, local authorities are considering rehabilitating the ecosystem. Ecological–economic modelling is applied in order to analyse the ecosystem's response to a reduction in nutrient loading and to compare the costs and benefits of a range of eutrophication control measures. The lake dynamics are modelled by means of a set of (differential) equations obtained through regression analysis of long-term water quality data. These data were provided by the Waterboard Reest and Wieden (2003), which has the policy mandate for water quantity and quality management in the area. Lake dynamics were combined with information on the supply of ecosystem services and the costs of measures in order to construct an integrated ecological–economic model for the De Wieden ecosystem.

Section 5.2 presents a description of De Wieden and the ecosystem services it provides. The ecological–economic model is developed in Section 5.3, and the efficiency, sustainability and equity implications of different management options are compared in Section 5.4. Finally, Section 5.5 analyses the uncertainties in the model, and presents a number of recommendations for the management of De Wieden as well as shallow lake ecosystems in general.

5.2 THE CASE STUDY AREA

5.2.1 Location and Water Management

The De Wieden wetland is located in the northeastern part of the Netherlands (52º42'N; 06º03'E). The lakes and canals of the area have been created through peat extraction activities that started in the late Middle Ages and continued up to the nineteenth century. This study considers water quality in the four biggest lakes of De Wieden – see Figure 5.1. The lakes are located in close proximity to each other and there is frequent exchange of water between them. The lakes total 1640 ha, and their average depth is around 1.8 m. The most important source of water is a canal entering the lakes from the north. This canal is fed by two small rivers that drain from the agricultural area located to the northeast of De Wieden, as well as by excess water released from a number of nearby polders. The main discharge of the lakes is to the downstream Lake Zwartewater (around 210 million m^3/year).

Up to the 1960s, the lakes were oligotrophic and the transparency was over 2 m, sufficient to see the lake bottom in most of the area. Since then, however, population pressures in the region increased and the agricultural production around the lakes intensified. This resulted in a rapid increase in

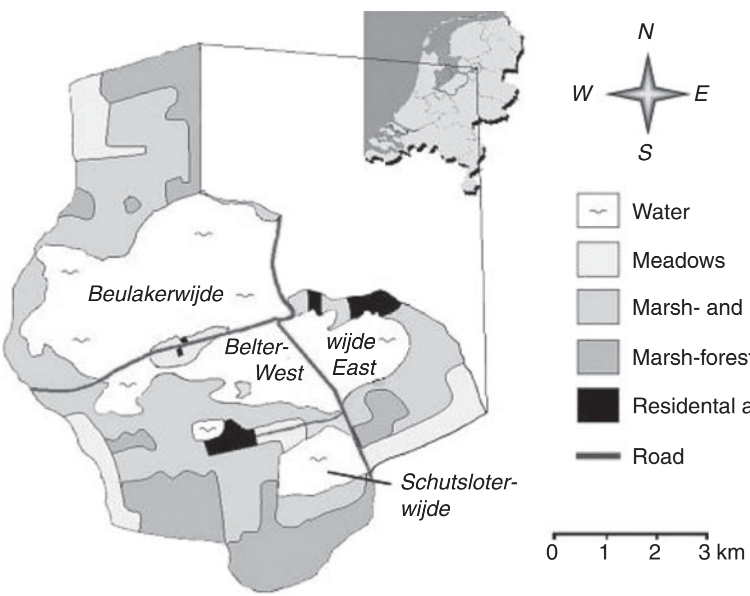


Figure 5.1 The De Wieden wetland

the input of nutrients in the area, which caused major ecological changes in the lakes. The original burbot (*Lota lota*) and roach (*Rutilus rutilus*) fish community was replaced by a bream (*Abramis brama*) dominated community with high phytoplankton biomass and low transparency.

Since the mid 1970s, nutrient influxes have decreased as a result of the construction of a sewage treatment facility in the nearby town of Steenwijk and the gradual implementation of policies aimed at reducing excess use of fertilisers by farmers. Consequently, the water quality has gradually improved and the current summer averages for total N and total P are around 2 mg/l and 0.1 mg/l, respectively. The current main sources of nutrients are two rivers originating in upstream agricultural areas, excess water from nearby polders and the effluent of a sewage treatment facility in the town of Steenwijk (Van Berkum, 2000). In spite of the reductions in nutrient loading in the last three decades, the water is still turbid, with an average summer transparency in the lakes of around 40 cm. The fish community remains dominated by bream.

5.2.2 Ecosystem Services

The four main ecosystem services of De Wieden are: (1) the provision of reeds for cutting; (2) the provision of fish (both provisioning services); (3) the provision of opportunities for recreation; and (4) nature conservation (both cultural services). The water purification service is based upon the breakdown and absorption of pollutants in the wetland, but is not very important for the economic value of De Wieden. To avoid double counting, its positive impact on recreation and nature conservation in De Wieden should not be included and its impact on the water quality of downstream Lake IJsselmeer downstream is very small because the De Wieden lake supplies only around 0.7% of the water flowing into Lake IJsselmeer. The four services are described below (they have been analysed in more detail in Hein et al., 2006).

Provision of reed (for cutting)

The reed of De Wieden has been cut for several centuries, and is mainly used for thatched roofs. Reed cutting is practiced on some 1400 ha (Natuurmonumenten, 2000), and is locally an important industry, employing around 220 people (De Bruin et al., 2001). Harvests are in the order of 665 kg/ha/year (De Bruin et al., 2001). Most of the reed cutting is done in combination with farming and/or fisheries, a suitable combination because most of the reed cutting takes place in the period October–March, and most farming and fishing activities are conducted in the period April–September. The total turnover from the reed cutting is around 800 000 euro, and the net value added (taken as a proxy for the value of the service) is around 480 000 euro (De Bruin et al., 2001). For the valuation in this study, it can be assumed that an increase or decrease in reed production in De Wieden can be compensated by other producers without changes in the price or quality of the product on the market, and it is therefore assumed that the consumer surplus resulting from reed production is zero.

Provision of fish

Professional fishermen fish each of the four lakes of the case study area, which comprise in total around 1600 ha open water. There are in total 11 professional fishermen working in the area (Van Dijk, 2003). The most important species is eel (*Anguilla anguilla*), which is fished with hoop nets. Fishermen also collect the whitefish that ends up in the nets, including pike, perch pike, bream and roach, although the prices of these fish are relatively low (Klinge, 1999). Total annual turnover of the fishery sector is estimated to be only around 215 000 euro (Klinge, 1999; De Bruin et al., 2001; Van Dijk, 2003). Investments are small, and the value added is estimated at around 140 000 euro (De Bruin et al., 2001; Van Dijk, 2003). In comparison with the total eel fisheries in the Netherlands, the contribution from De Wieden is small; less than 1% of the Dutch market is supplied by De Wieden (Klinge, 1999). As with reed cutting, it can be assumed that the consumer surplus generated by the fisheries activities in De Wieden is zero.

Recreation

De Wieden is an important area for recreation, attracting visitors that come for short holidays as well as day-trips. Some 170 000 visitors per year enjoy a range of activities including boating, sailing, hiking, fishing, canoeing, surfing, swimming and sun-bathing. Benefits of the recreational opportunities of De Wieden also accrue to the local companies offering recreational services. These include boat and canoe rental agencies, hotels, camping sites, marinas, and bars and restaurants. Both companies located in the study area and companies located in the immediate surroundings of the study area benefit from the visitors to De Wieden.

Hein et al. (2006) carried out a simple travel zone method to analyse the demand curve for and consumer surplus generated by the recreation service of De Wieden. The area under the demand curve, equalling the consumer surplus, is around 880 000 euro (which equals around 5 euro per visit). The value added generated by the recreation sector is calculated on the basis of the total turnover and the net value added of the recreation sector. For the recreational companies in De Wieden, the net value added is around 22% of turnover (De Bruin et al., 2001) and the value added generated by De Wieden is around 800 000 euro per year. The total value of the recreational service of De Wieden can be found by summing the benefits accruing to the visitors and the net value added of the recreational sector in the immediate surroundings of De Wieden.

Nature conservation

De Wieden is very important for biodiversity conservation in the Netherlands. It provides a habitat to a wide range of water- and meadow-birds, dragonflies, butterflies, fish, etc., and it contains, together with the adjacent wetland 'De Weerribben', the world's only population of a specific subspecies of the Large Copper butterfly (*Lyacena dispar*), see Figure 5.2. The Eurasian river otter (*Lutra lutra*), which became extinct in the Netherlands some 12 years ago, was reintroduced in the area in June 2002. The area is protected under national laws, is included in the EU Habitat and Birds directives and was appointed a Ramsar site in 2002. The non-use value associated with the nature conservation service can be analysed with CVM (Arrow et al., 1993; Hailu et al., 2000). Although CVM has increasingly been applied to analyse the non-material benefits derived from ecosystems, there are still a number of methodological constraints (e.g. Carson, 1998), and the implementation of a well-designed CVM study is outside the scope of this chapter. Instead, in order to obtain a crude approximation of the economic value of the nature conservation service, it is assumed that the amount of money contributed to the NGO

Source: M. Grutters.

Figure 5.2 Large Copper butterfly in De Wieden

'Natuurmonumenten' that manages De Wieden provides an indication of the willingness-to-pay (WTP) of the Netherlands' public for nature conservation in De Wieden. An advantage of this approach is that it measures actual payments instead of a stated willingness-to-pay. However, the estimate only indicates the minimum amount the Dutch public is willing to pay. The actual amount will be higher because some members of the NGO may be willing to pay a larger sum if this would be necessary to preserve De Wieden, and because some non-members may also be willing to pay for nature conservation in De Wieden.

The main objective of the NGO 'Natuurmonumenten' is to preserve Dutch nature for present and future generations by managing and conserving a number of natural parks in the Netherlands. In the year 2002, the NGO received in total around 29 million euro in donations (Natuurmonumenten, 2003). To estimate the WTP for conservation of De Wieden, it is assumed that the WTP for this area is proportional to the aerial surface of De Wieden in comparison to the total area of the sites managed by the NGO. The total area of the sites managed by the NGO is 71 200 ha (June 2002), of which 5400 ha (7.6%) are located in De Wieden (Natuurmonumenten, 2003). Hence, the minimum value of the nature conservation service of the De Wieden wetlands is estimated at around 2.2 million euro per year.

All services have been valued in monetary terms. However, different indicators have been used to indicate the surplus generated by the services (value added, consumer surplus and payments to Natuurmonumenten). This restricts the possibilities to add and compare the values. Nevertheless, the values of the four services have been added to provide a crude indication of their total value, as presented in Table 5.1. The approximate, combined monetary value of the four selected ecosystem services provided by De Wieden is in the order of 4 500 000 euro per year, or 830 euro per ha per year.

Table 5.1 Economic value of the ecosystem services supplied by the study area

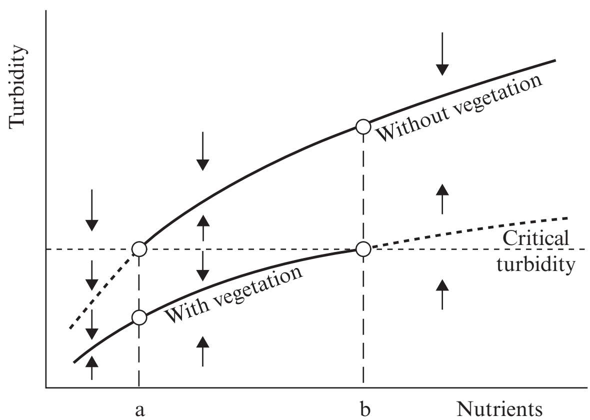
Ecosystem service	**Economic value (euro/year)**
Reed cutting	480 000
Fisheries	140 000
Recreation	1 680 000
Nature conservation	2 200 000
Total value of the selected services	4 500 000

5.3 THE ECOLOGICAL–ECONOMIC MODEL

5.3.1 Dynamics of Shallow Lake Ecosystems

Eutrophic shallow lakes can, under certain conditions, be in either of two states: a vegetated state with clear water and an unvegetated, turbid state dominated by phytoplankton (Timms and Moss, 1984; Jeppesen et al., 1990; Scheffer, 1998; Meijer, 2000; Van Nes et al., 2002). The two states represent alternative equilibriums that exist over a certain range of nutrient conditions. At lower nutrient levels, only the vegetation-dominated state exists, whereas at the highest nutrient levels, there is only a turbid state.

Shallow lakes tend to be subject to hysteresis (Scheffer, 1998). Vegetated lake bottoms promote the development of more vegetation by keeping the lake water clear through (1) stabilizing lake sediments; (2) providing a shelter to *Daphnia* (waterfleas) that graze upon the phytoplankton; and (3) promoting a piscivorous fish community that controls the numbers of fish species feeding upon Daphnia. In a turbid water state, the lack of light penetration prevents the establishment of water plants, and a fish community dominated by benthivorous fish enhances the suspension of lake sediments, further increasing turbidity. Hence, the vegetated state of clear lakes is robust during eutrophication, but to restore the vegetated clear state once the lake has switched to a turbid state, the nutrient level must be reduced to a much lower level than the one at which vegetation collapsed, see Figure 5.3.



Source: Scheffer (1998).

Figure 5.3 Hysteresis in shallow lake ecosystems

The presence of multiple steady states and hysteresis has important implications for lake management. Shallow lakes that are currently in a turbid state without vegetation, can be restored to a clear water state through (1) reduction of the nutrient loading to point ‘a’ in Figure 5.3; or (2) partial reduction of nutrient concentrations to below point ‘b’ in Figure 5.3 in combination with biomanipulation (Scheffer, 1998). Biomanipulation involves the removal of a substantial part (>75%) of the benthivorous fish in order to evoke a switch from a turbid to a clear water ecosystem (Klinge et al., 1995; Meijer, 2000). Removal of the benthivorous fish allows *Daphnia* to graze the phytoplankton, reduces the resuspension of sediments through activities of the fish and causes a period of clear water during which water plants can develop (Meijer, 2000).

5.3.2 Model Structure and Data

The ecological–economic model describes the response of the ecosystem to eutrophication control measures. Total-phosphorus (P) concentrations are used as the control variable because P is the main limiting nutrient in the lakes (Van Berkum, 2000; Waterschap Groot Salland, 2000), and because reduction of nitrogen only may enhance blooms of toxic *Oscillatoria* species that are able to fix atmospheric nitrogen (Van der Molen et al., 1998; Scheffer, 1998; Hosper, 1997). The model contains two approaches to eutrophication control: without and with biomanipulation. In the model, biomanipulation is expressed through a modified relation between P loading and algae growth that reflects the different ecological processes that occur after biomanipulation. Application of biomanipulation is also reflected in the costs of the eutrophication control measures.

The model follows a steady state approach to analyse the impacts of eutrophication control measures on turbidity and macrophyte grow. However, once macrophyte growth exceeds a minimum cover of the lake bottom (25%), it is assumed that the whole lake is converted to a clear water state in a period of five years – in line with current experiences in large Dutch lakes (Meijer, 2000). This leads to a gradual increase in the supply of ecosystem services by the lake. The benefits of this transition are expressed as net present value (NPV) in order to compare them with the costs of eutrophication control measures.

The various steps included in the model are presented in Figure 5.4 and described below.

Eutrophication control measures; costs and impacts on P loading

Potential eutrophication control measures have been examined by the local waterboard (Van Berkum, 2000). In collaboration with the main

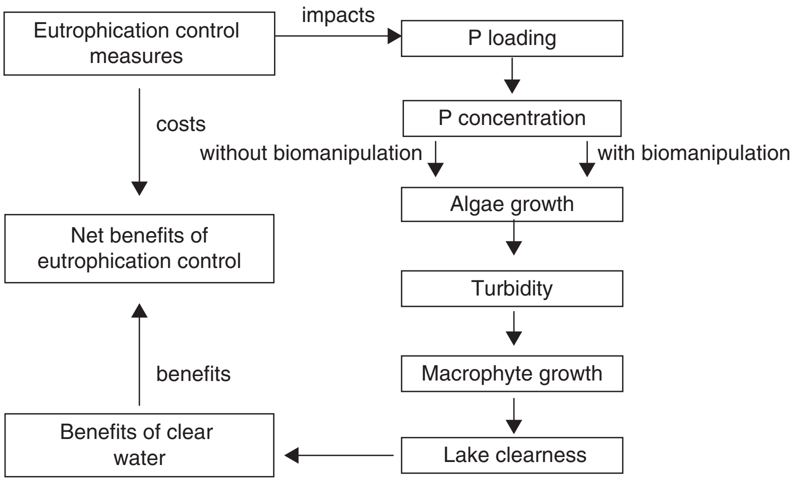


Figure 5.4 Outline of the model

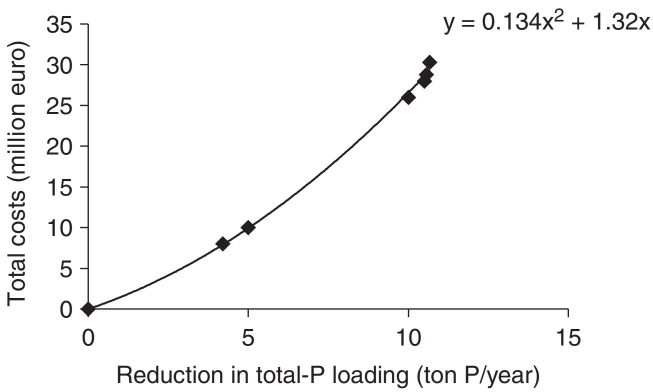
stakeholders in the area (nature conservationists, farmers and representatives from the tourist sector), they have identified the most feasible measures in terms of cost-effectiveness and acceptance for local stakeholders. For this study, the six measures that have an impact on the four central lakes of De Wieden have been selected, see Table 5.2. All measures have an impact on the inflow of phosphorus through canals and streams, except Measure 5 that addresses the supply of phosphorus through recreation in the lakes. Measure 3 involves the chemical removal of phosphates from the two small rivers that drain the agricultural plateau northeast of De Wieden. This has also been applied in rivers flowing into other wetlands in the Netherlands (e.g. the Nieuwkoopse Plassen) and requires the construction of a new treatment facility. The measures are independent of each other and their impacts are additive (Van Berkum, 2000). The costs have been recalculated on the basis of Van Berkum (2000) and are expressed as net present value (NPV) including investment, and operation and maintenance costs, using a discount rate of 5% and a discounting period of 25 years. The measures have been ranked according to their cost-effectiveness. The costs of biomanipulation in the Netherlands are around 230 euro/ha (Hosper et al., 1992), hence 380 000 euro for the four lakes together.

The measures in Table 5.2 have been plotted in order to obtain an approximate cost curve (Figure 5.5). For reasons of simplicity, the model uses a continuous cost curve, rather than the discrete set of measures to calculate the efficient pollution reduction. This also reflects that some of the measures can be partially implemented. For example, the first

Table 5.2 Eutrophication control measures for De Wieden

Measure	P-reduction (ton/year)	Costs (NPV) (million euro)	Cost-effectiveness (million euro/ ton P)
1. Diverting eutrophic polder water	4.2	8	1.90
2. Enhancing sewage treatment plant at Steenwijk	0.8	2	2.50
3. Phosphorus reduction inflowing rivers	5	16	3.20
4. Increased connection to sewage system	0.5	2	4.00
5. Reducing P-loading from recreation through information of visitors and enhanced sanitary facilities in the area	0.06	0.8	13.33
6. Reducing sewage spill-over	0.1	1.5	15.00
Total	10.7		

Source: Adapted from Van Berkum (2000).



Source: Hein (2006a).

Figure 5.5 Total discounted cost curve for P control measures

measure, diversion of eutrophic polder water, can be implemented for one up to a total of four surrounding polders. Based on the continuous cost curve, the following equations are used in the model to describe the costs of eutrophication control measures.

$$\text{Without biomanipulation: } TC = 0.134\,\Delta L^2 + 1.32\,\Delta L \qquad (5.1a)$$

$$\text{With biomanipulation: } TC = 0.134\,\Delta L^2 + 1.32\,\Delta L + CB \qquad (5.1b)$$

where:
TC = Total costs of the reduction in P loading (euro, expressed as NPV)
ΔL = Reduction in total-P loading; the current loading is 14.8 ton/year
CB = Costs of biomanipulation (380 000 euro)

From P loading to P concentration

The relationship between P loading and the P concentration in the lakes has been modelled using a steady-state approach. This approach does not fully reflect the dynamic behaviour of P in shallow lakes, which is driven by a range of processes such as the uptake and release by algae, reversible or irreversible absorption and adsorption by the lake sediments, etc. (Scheffer, 1998). Nevertheless, it can be used to indicate the longer term trends in the P concentrations of a lake as a function of P loading (Vollenweider, 1968; Hosper, 1997; Scheffer, 1998). Using data from 14 Dutch shallow lakes, Hosper (1997) updated the original Vollenweider equations and provided the following equation :

$$P = L \cdot [0.201\, log\,(z/\tau) + 0.322]\,/\,(z/\tau) \quad (n = 63;\, r^2 = 0.72,\, t = 1.7) \qquad (5.2)$$

where:
P = Phosphorus concentration (mg/l)
L = Loading (g P/m^2/y)
z = Average lake depth (m)
τ = Hydraulic residence time (y)

The applicability of the formula for De Wieden has been tested by comparing the current P loading and P concentration of De Wieden. In the period 1998–2002, the average P concentration of the four lakes was 0.010 mg/l, z was 1.82 m and τ was 0.43 (Van Berkum, 2000). Inserting these numbers in Equation (5.1) yields a P loading of 0.9 g P/m^2/y, which equals 15 ton P/year. This corresponds well with the P loading calculated in the P balance, and it is concluded that Equation (5.2) also provides a valid equation to model the relation between P loading and P concentration in De Wieden.

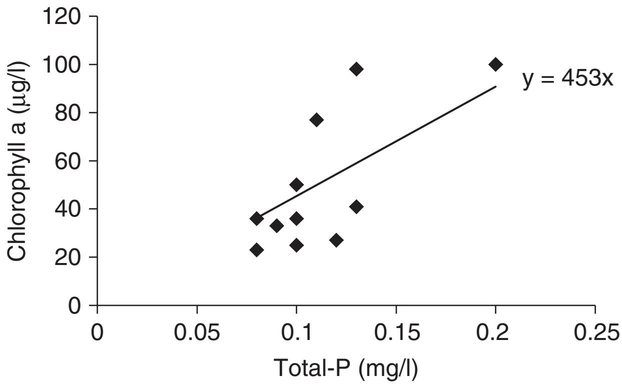
It is assumed that there is no significant resupply of P through the sediments following a reduction of P concentrations in the water column. Two factors underlie this assumption: (1) the build-up of sediments has been relatively small in the lakes, only around 20% of the lake bottom currently has a sediment layer over 10 cm depth; and (2) the water column of

De Wieden is very rich in iron (concentrations range from 0.4 to 2 mg/l), which enhances the immobilisation of P in lake sediments under the aerobic conditions that prevail at the lake bottoms of De Wieden (Boers et al., 1998; Scheffer, 1998).

From P concentration to algae growth

In this model, chlorophyll a is used as indicator for algae biomass. In general, there is a positive correlation between the concentration of the most limiting nutrient and the chlorophyll a concentrations (Vollenweider, 1968; Dillon and Rigler, 1974). Hence, if P loading is substantially reduced, it can be assumed that P becomes the main limiting nutrient. As P is, under current conditions, already the limiting nutrient in the De Wieden lakes (Van Berkum, 2000), a positive relation between total-P and chlorophyll a is assumed in the model. However, the relation between total-P and chlorophyll a is complex. The total-P concentration determines the algae growth, but the algae concentration also partly determines the total-P concentration because a substantial part of the total-P in the water column can be contained in the algae. The uptake of soluble P in the water column by the algae evokes a new equilibrium between the P contained in the lake sediments and the P in the water column, and can cause the release of P from the sediment as the algae biomass increases in the course of the year (see Scheffer, 1998 for details). Therefore, for the case *without biomanipulation*, the yearly average total-P and chlorophyll a concentrations have been analysed. This reduces the error in the regression analysis because it excludes the annual variation between total-P and algae concentrations. The annual variation, driven by the release of phosphorus from the sediment as a consequence of algae growth, tends to increase the slope of the curve depicting the relation between total-P and chlorophyll a (Scheffer, 1998). For the case *with biomanipulation*, data availability was insufficient to allow for the analysis of the relation between total-P and algae on the basis of yearly averages, and all data were used. The specific equations included in the model for the two approaches are defined below.

Without biomanipulation The relation between total-P and chlorophyll a is established on the basis of existing data for the four lakes, which were available for the period 1992–2002, see Figure 5.6. The unexplained variation is caused by variations in turbulence, light regime and grazing (Scheffer, 1998). A linear relation between algae growth and total-P concentration is assumed (cf. Hosper, 1997). The trend line inserted in Figure 5.6 is used as the function describing the development of chlorophyll a as a function of P concentrations. The equation used in the model is:



Source: Hein (2006a).

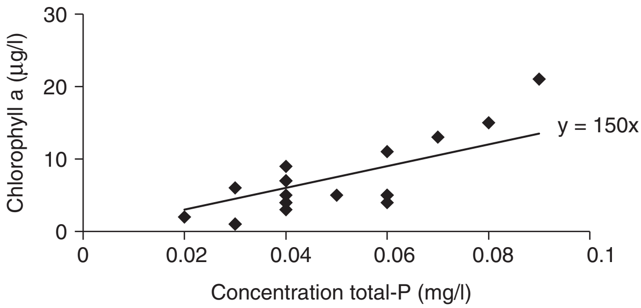
Figure 5.6 Relation between total-P and chlorophyll a in De Wieden

$$Ch = 453 \cdot P \quad (n = 12;\ r^2 = 0.88;\ t = 8.6) \qquad (5.3a)$$

where:
Ch = chlorophyll a concentration (µg/l)
P = total phosphorus concentration (mg/l)

With biomanipulation It is assumed that the main impact of biomanipulation, i.e. the removal of the majority of the benthivorous and zooplankton-eating fish, is a strong increase in *Daphnia* concentrations and, hence, the substantial reduction of algae concentrations in relation to total-P concentrations. In order to obtain quantitative insight in this effect, Lake Duinigermeer is used to establish an alternative relation between these two factors. Lake Duinigermeer is located near to the four lakes considered in the case study (around 4 km distance). Through biomanipulation, it was restored to a clear water state in 1994. During the period 1994–2002, the lake has been in a clear water state with reduced fish stocks and high concentrations of *Daphnia*. Figure 5.7 shows the chlorophyll a concentrations of Lake Duinigermeer as a function of total-P concentrations, and a trend line. Because of the more limited availability of data, all available samples are used, instead of yearly averages. However, all samples that were potentially nitrogen limited (with a N:P ratio below 20) have been excluded from the analysis. For the approach with biomanipulation, Equation (5.3), which describes the relation between total-P and chlorophyll a, becomes:

$$Ch = 150 \cdot P \quad (n = 15;\ r^2 = 0.92,\ t = 10.2) \qquad (5.3b)$$



Source: Hein (2006a).

Figure 5.7 The relation between total-P and chlorophyll a for Lake Duinigermeer

where:
Ch = chlorophyll a concentration (µg/l)
P = total phosphorus concentration (mg/l)

From algae growth to transparency

Transparency is generally measured in terms of Secchi depth (Scheffer, 1998). Secchi depth samples are available for the period 1991–2002. Secchi depth is determined by two main factors: algae and sediment concentrations in the water. The sediment concentrations in the water column depend, among others, on the wind speed. Because the wind speed varies from one day to the next, the relation between algae growth and Secchi depth is subject to considerable intra-annual variation. Therefore, the equation is based on the yearly averages of algae concentration and inverse Secchi depth (Figure 5.8). A linear relation between chlorophyll a concentrations and the inverse of the Secchi depth is assumed (cf. Hosper, 1997):

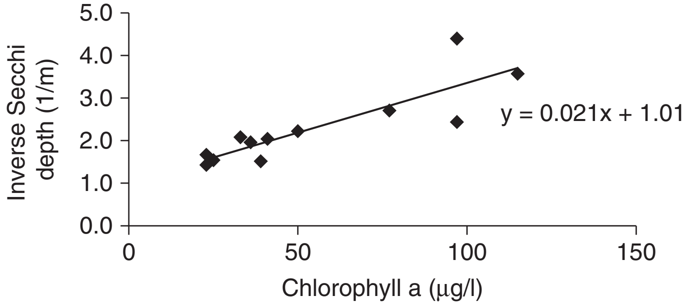
$$SD = 1 / (0.021 \,.\, Ch + 1.01)$$

$$(n = 12;\ r^2 = 0.74;\ t(\text{coefficient}) = 5.2;\ t(\text{constant}) = 3.6) \qquad (5.4)$$

where:
SD = Secchi depth (m)
Ch = Chlorophyll a concentration (µg/l)

From transparency to lake clearness

The Secchi depth is converted to the depth at which sufficient light penetrates to allow the growth of charophyte waterplants. Charophyte plants



Source: Also published in Hein (2006a).

Figure 5.8 Relation between chlorophyll a and the inverse Secchi depth

grow on the lake bottom, rooted by rhizoids, and need light for photosynthesis. They stabilise the sediment on the lake bottom and provide a shelter to *Daphnia*. It is assumed that charophytes require 10% of the surface light to grow (Hosper, 1997). The downward irradiance of light diminishes with depth according to the following formula (Kirk, 1994):

$$E(z) = E(0) \cdot e^{-K \cdot z}$$

or:

$$\ln E(z) / E(0) = -K \cdot z \tag{5.5}$$

where:
$E(z)$, $E(0)$ = the values of downward irradiant at depth z and just below the surface
z = depth
K = vertical extinction coefficient

Although there are substantial variations between lakes in terms of the relation between the light attenuation and the Secchi depth (Scheffer, 1998), K can be approximated by the following formula (Kirk, 1994):

$$K = PA/SD \tag{5.6}$$

where:
PA = Poole–Atkins coefficient
SD = Secchi depth

There are no measurements of the PA for the De Wieden lakes. Therefore, it is assumed that the PA of De Wieden is 1.5, which is the average of four nearby Dutch lakes for which the PA is available (Loosdrecht, IJsselmeer, Wolderwijd, Nuldernauw) (Hosper, 1997). These lakes are comparable to De Wieden in terms of size, depth and the composition of the inflowing surface water. Hence, at depth z_{ch}, at which 10% of the surface light remains, ln $[E(z)/E(0)] = -2.3$, and:

$$z_{ch} = 1.53 \cdot SD \tag{5.7}$$

where:
z_{ch} = the maximum depth at which charophytes can develop (m)
SD = Secchi depth (m)

The part of the lake that can be covered with plants at a given light penetration depends on the depth profile of the lakes. The depth profile of the lakes of De Wieden has been studied by Van Berkum (2000). The depth profile of the four lakes has been plotted in a scatter diagram and an S-curve has been fitted to the profile. The fitted curve is used to indicate how much of the lake bottom will be covered by waterplants at a certain light penetration. This results in the following equation:

$$PL_1 = 100\,/\,(1 + 375.e^{-4 \cdot z_{ch}})\ (n = 10;\ F = 21;\ r^2 = 0.98) \tag{5.8}$$

where:
PL_1 = percentage of the lake bottom covered with waterplants in year 1
z_{ch} = depth at which 10% of the surface light penetrates

In large lakes, there can be a combination of clear and turbid water states, with the water above the water plants clear and the water in the other parts of the lake turbid (Scheffer et al., 1994; Meijer, 2000). However, if the percentage of the lake covered with waterplants passes a critical cover, the whole lake turns into a clear water state (Meijer, 2000). Based on the review of Dutch lake ecosystems, Meijer (2000) suggests that a cover of around of 25% of the lake bottom is sufficient to gradually turn the whole lake into a clear water system. Obviously, this critical assumption is subject to high uncertainties. In reality, the percentage will differ for each lake, depending on the physical and biological characteristics of the lake. However, specific data for De Wieden are not available (and may not be before it is attempted to bring the lakes into a clear water state) and the 25% threshold is used in the model, while sensitivity analyses are provided for a threshold of 20% and 30%. Hence, it is assumed in the model that if waterplants cover at least 25% of the lake bottom following a reduction in nutrient loading without or in combination with biomanipulation, the whole lake switches to a clear water state. It is also assumed that the establishment of charophyte waterplants in the whole lake (and hence the transition of the whole lake to a clear-water state), would take five years, in line with the development of vegetation in nearby Lake Wolderwijd. If the threshold is not passed, charophyte cover is assumed to remain constant. An exponential growth curve has been fitted to the development of charophyte cover in Lake Wolderwijd (Meijer, 2000), and it is assumed that the development of waterplants in De Wieden will proceed accordingly, as a function of time and the waterplant cover in year 1:

$$PL_t = \frac{81}{1 + 70 \cdot e^{-1.5 \cdot [t + (PL_1 - 0.25)/0.1875]}} + 19 \quad (\text{for } t \geq 2 \text{ and } PL_1 > 0.25) \tag{5.9}$$

where:
PL_t = percentage of the lake bottom covered with waterplants in year t

The benefits of clear water

It is assumed that only the two services, nature conservation and recreation, will benefit from a switch to clear water. Regarding nature conservation, analysis of the habitat requirements of all 108 threatened species occurring in the area revealed that 35 of them would benefit from a switch to clear water, and that no threatened species would be negatively affected by it. The species benefiting include waterbirds, dragonflies, fish, vascular plants and two mammals (the Eurasian river otter, *L. lutra*, and the European water shrew, *Neomys fodiens*). As for the recreation service, swimmers, in particular, but also sailors and surfers appreciate clear water, provided that waterplants do not hamper the access of the boats to the lakes (Van der Veeren, 2002). Fisheries and reed cutting will probably not significantly benefit from a transition to clear water. For local fisheries, the most important species is eel, which is relatively insensitive to modest changes in P concentrations or a potential shift to clear water (Svedang et al., 1996). Reed growth also does not respond to such changes (Clevering, 1998; Romero et al., 1999).

The monetary benefits of a switch to clear water are difficult to quantify. Regarding the nature conservation service, it is very difficult to translate the potential changes in species occurrence into a monetary value (Spash and Hanley, 1995; Nunes and van den Bergh, 2001). Concerning the recreation service, it is not known if, and by how much, visitor numbers would increase following an increase in water transparency. In addition, there is no accurate information on the willingness-to-pay of visitors for clear water. Therefore, the study does not embark on a valuation of these benefits. Instead, the model calculates the net benefits of a reduction in total-P loading for a range of assumed values of the increased supply of the nature conservation and recreation service following a switch to clear water. In other words, the net benefits of eutrophication control measures are calculated as a function of both (1) the level of eutrophication control and the type of measures implemented (without or with biomanipulation); and (2) the assumed value of the increase in the supply of the two ecosystem services.

The formula used to calculate the net benefits is shown below. The net benefits (*U*) and the total costs (*TC*, from equations (5.1a) and (5.1b) of the eutrophication control measures are expressed as NPV (discounted over 25 years, using a 5% discount rate). It is assumed that the benefits increase proportionally with the percentage of the lake that has clear water (which may vary over time).

$$U = \sum_{t=0}^{25} 1/(1+r)^t \cdot (TB \cdot PL_t) - TC \tag{5.10}$$

where:
U = Net benefits of the eutrophication control measures (euro)
r = Discount rate at time t (5%)
TB = Annual benefits as a result of a switch to clear water (euro/year)
PL_t = Percentage of clear water in the lake (%) in year t
TC = Total costs of the eutrophication control measures (euro)

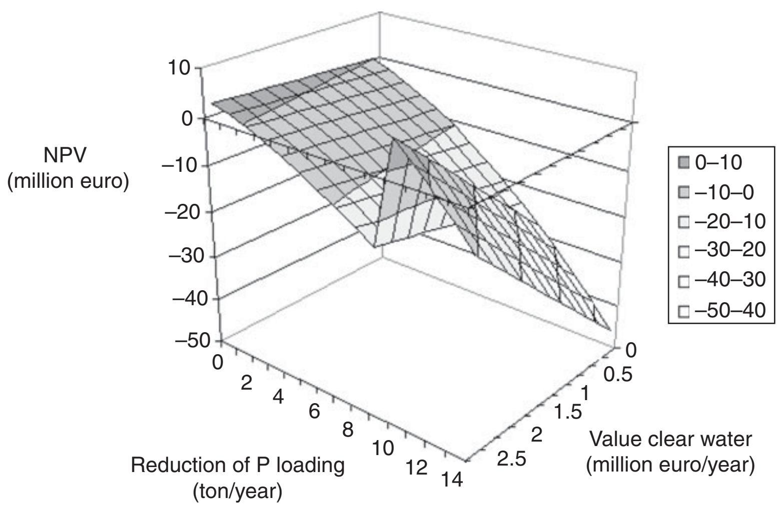
5.4 RESULTS

5.4.1 Costs and Benefits of Management Options

The costs of eutrophication control relate to the costs of the P control measures and, in the second approach, also to the costs of biomanipulation. The benefits increase proportionally with the part of the lake that is in a clear water state. They are derived from increased opportunities for nature conservation and recreation in clear water. As explained in the previous section, these benefits are difficult to value in monetary terms, and the net benefits of eutrophication control are calculated for a range of assumed values of the annual benefits of a switch to clear water. The cost-efficiency of each level of eutrophication control is expressed as a NPV, with all annual cost and benefits converted on the basis of a 5% discount rate, and using a 25-year discounting period.

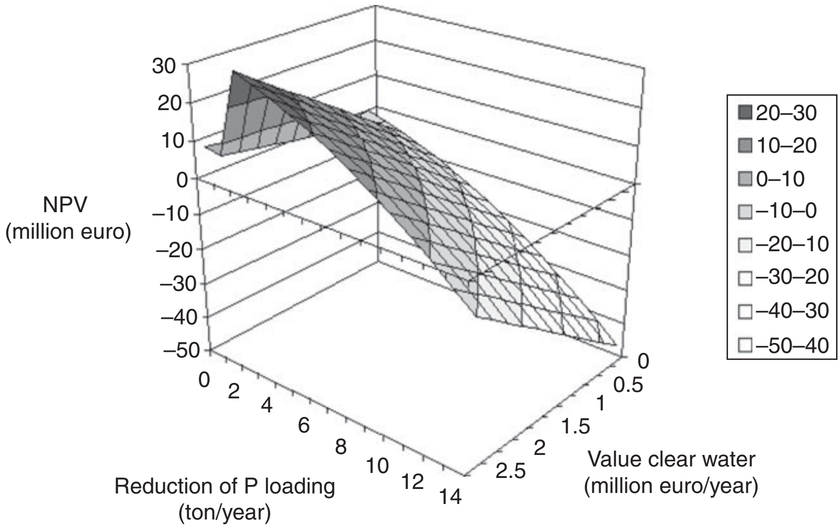
Efficiency of reducing P loading: without biomanipulation

Figure 5.9 shows the economic efficiency of reducing the inflow of P in De Wieden without biomanipulation. The figure has three axes, representing (1) the pursued reduction in P inflow; (2) the annual value attributed to a switch to clear water; and (3) the net benefits (expressed as NPV) as a function of the two previous variables. If the annual benefits of clear water are larger than zero, there is a bimodal distribution with respect to the value of eutrophication control. In this case, there is a point of local maximum efficiency at a zero reduction in P loading, and a second point of local maximum efficiency for a reduction in P loading of around 9 ton/year. The second local maximum corresponds to the minimum P inflow reduction at which the complete lake changes from the current turbid water state to a clear water state. The corresponding P concentration in the lake is predicted to be 0.03 mg/l. Hence, in the absence of biomanipulation, the model indicates that the lakes can be expected to switch to a clear water state at a total-P concentration of 0.03 mg/l. Which of the two local maximums is preferable depends on the benefits attributed to clear water. If the annual benefits provided by clear water (through enhanced biodiversity protection and more opportunities for recreation) are valued at at least 1.75 million euro/year, it is economically efficient to reduce the inflow



Source: Hein (2006a).

Figure 5.9 Net present value of eutrophication control measures in De Wieden (without biomanupilation)



Source: Hein (2006a).

Figure 5.10 Net present value of eutrophication control measures in De Wieden (with biomanupilation)

of total-P by 9 ton/year in order to obtain clear water. This involves total investment costs of 22 million euro.

Efficiency of reducing P loading: with biomanipulation

The approach normally followed in the Netherlands to rehabilitate shallow lake ecosystems to a clear water state is through biomanipulation, which may or may not be applied in combination with a reduction in P loading (Perrow et al., 1997). Figure 5.10 shows the economic efficiency of reducing the inflow of P in De Wieden with biomanipulation. When biomanipulation is applied, there is a point of local maximum efficiency at a zero reduction in P loading, and a second point of local maximum efficiency for a reduction in P loading of 3 ton/year. The second local maximum corresponds to the minimum P inflow reduction at which the complete lake changes from the current turbid water state to a clear water state. The model indicates that the corresponding P concentration in the lake is 0.08 mg/l. Hence, biomanipulation is predicted to be successful if the lakes have reached a total-P concentration of 0.08 mg/l. Which of the two local maximums is preferable depends on the benefits attributed to clear water. If the annual benefits provided by clear water (through enhanced biodiversity protection and more opportunities for recreation) are valued at at least 0.5 million euro/year, it is economically efficient to reduce the inflow of total-P by 3 ton/year and apply biomanipulation in order to obtain clear water. This involves a total investment cost of 5 million euro.

5.4.2 Comparison of Management Options

Based on the analysis presented above, three management options for De Wieden can be analysed with regards to their efficiency, sustainability and equity implications. The management options are, respectively, no further action, nutrient pollution reduction without biomanupilation and nutrient pollution reduction in combination with biomanupilation. The degree to which the three criteria for ecosystem management are fulfilled for each of the ecosystem management options is discussed below.

Efficiency

The presence of two steady states of the ecosystem creates two points of local maximum efficiency in ecosystem management, each belonging to one state. For the shallow lake ecosystem studied, one point of local maximum efficiency belongs to the current, turbid water state and involves no reduction in eutrophication levels. The other local maximum corresponds to implementation of the cheapest management strategy that would cause a bifurcation to a clear water state. For the lakes of the De Wieden wetland, this involves reducing the inflow of total-P by 2 ton/year, in combination with biomanipulation.

The study shows that an approach that combines nutrient pollution control with biomanipulation (removal of >75% of the bream in the lakes) is the cheapest option of restoring clear water in the lake. The removal of an extra amount of nutrients in the case where biomanipulation is not applied is more expensive than the biomanipulation. Whether it is economically efficient to select the second local maximum and pursue a clear water state depends upon the ratio between the costs of the measures and the benefits of an increased supply of ecosystem services following rehabilitation. The implementation of eutrophication control measures that do not lead to a rehabilitation of the clear water state is not cost-effective.

In the case of De Wieden, it appeared difficult to assess how ecosystem services supply would change following rehabilitation of the ecosystem. Whereas the present value of the four key services of De Wieden could be quantified using the available data, there was a high degree of uncertainty on how much the performance of the services would increase if the lakes were restored to a clear water state. As analysed above, where nutrient concentrations would be reduced and the water would be restored to a clear water state, there would be few impacts on fishing and reed cutting, but an increased supply of the recreation and biodiversity services can be anticipated. It proved difficult to quantify these increases. Estimates of the willingness-to-pay of visitors for experiencing clear rather than turbid swimming water are available for the Netherlands (Van der Veeren, 2002), but it is uncertain how the number of visitors coming to De Wieden would increase if the lakes shifted to a clear water state. Clearly, this has a major impact on the recreational service of the area (turnover of the tourism industry, visitors' consumer surplus). A positive impact can also be anticipated for the biodiversity service. Interviews with ecologists working in the area showed that many threatened species would benefit from a switch to clear water, and no rare or threatened species would be expected to be worse off (see below). However, the precise increase in species abundance would be very difficult to predict and very difficult to quantify in monetary terms. In principle, these data deficiencies could be overcome with regression analysis of tourism numbers in lakes with different properties including different water qualities, and by looking at species occurrence in otherwise comparable lakes with clear water. However, these methods are data intensive and prone to a considerable degree of uncertainty, and they have not been pursued for this case study.

Sustainability

Reducing nutrient pollution and restoring the clear water state of De Wieden would enhance the ecological value of the area. The ecological value of De Wieden is indicated by the occurrence of species that are considered of national importance for nature conservation, the so-called 'target' species. Target species have been defined by a large panel of experts as 'species that need specific consideration in Dutch nature policy on the basis of their rarity, and/or a negative trend in occurrence nationally and/ or internationally' (Bal et al., 2001). In total, 1006 target species have been distinguished for the Netherlands, including mammals, birds, reptiles, fish, insects, bivalves, snails, worms, plants and mosses. Of the Dutch target species, 108 occur in De Wieden.

The impact of a shift to clear water on each of the target species present in De Wieden has been analysed based on Natuurmonumenten (2000), the Netherlands Association for Dragonfly Studies (2002) and Noordhuis et al. (2002), cross-checked with ecologists working in the area. It appears that 35 target species would benefit from a shift to clear water, because their opportunities to forage or reproduce would increase, or because they depend upon species that would benefit from a transition to clear water (Table 5.3). There are no target species that are likely to decline following a switch to clear water in De Wieden.

Analysing the sustainability of water management policy options in De

Table 5.3 List of target species occurring in De Wieden that would benefit from a shift to clear water

Species	
English	**Latin**
Birds	
Tundra swan	*Cygnus colombianus spp. bewickii*
Marsh harrier	*Circus aeruginosus*
Common bluethroat	*Luscinia svecica*
Black tern	*Chlidonias niger*
Purple heron	*Ardea purpurea*
Great bittern	*Botaurus stellaris*
Little bittern	*Ixobrychus minutus*
Common kingfisher	*Alcedo atthis*
Spotted crake	*Porzana porzana*
Night heron	*Nycticorax nycticorax*
Little grebe	*Tachybaptus ruficollis*
Savi's Warbler	*Locustella luscinioides*
Sedge Warbler	*Acrocephalus schoenobaenus*
Bearded tit	*Panurus biarmicus*
Great reed warbler	*Acrocephalus arundinaceus ssp. arundinaceus*
Fish	
Crucian carp	*Carassius carassius*
Burbot	*Lota lota*
Miller's thumb	*Cottus gobio*
Barbel	*Barbus barbus*
Atlantic salmon	*Salmo trutta*
Butterflies	
Grizzled Skipper	*Pyrgus malvae*
Large Copper	*Lycaena dispar*
Dragonflies	
Green hawker	*Aeshna viridis*
Large White-faced dragonfly	*Leucorrhinia pectoralis*
Yellow-spotted dragonfly	*Somatochlora flavomaculata*
Siberian Winter Damsel	*Sympecma paedisca*
Scarce chaser	*Libellula fulva*
Hairy dragonfly	*Brachytron pratense*
Norfolk hawker	*Aeshna isosceles*
Mammals	
Eurasian river otter	*Lutra lutra*
European water shrew	*Neomys fodiens*
Vacular plants	
Fen orchid	*Liparis loeselii*
Floating bur-reed	*Sparganium natans*
Intermediate bladderwort	*Utricularia intermedia*
Lesser bladderwort	*Utricularia minor*

Wieden requires defining the reference year and the situation with which environmental quality is compared. If the base year is the present state of the ecosystem, i.e. before implementation of the pollution control measures, all three options (doing nothing, nutrient pollution control, nutrient pollution control in combination with biomanipulation) would be sustainable, as none of them would lead to a further decline in biodiversity or the capacity of the ecosystem to supply ecosystem services.

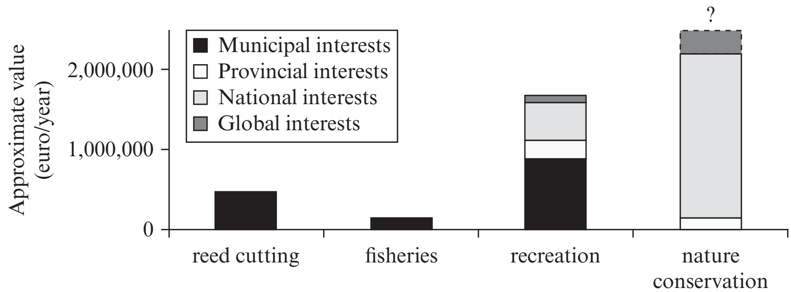
However, in the case where sustainability in De Wieden is analysed with regards to the bigger picture of environmental change in the Netherlands – which may be justified because of the national importance of the area for biodiversity conservation and recreation – a different conclusion can be drawn. In the Netherlands, as a whole, there is an ongoing decline in biodiversity and numbers of target species (PBL, 2008). If the national perspective is taken, i.e. an ongoing negative trend in the numbers of target species in the country, restoring the lakes to a clear water state will contribute to sustainable ecosystem management.

Hence, the case study shows the difficulty of interpreting sustainability at the level of the ecosystem. If the present condition of the lake itself is taken, rehabilitation would not be required to reach sustainability, but if the reference case is the Netherlands, as a whole, or a historical situation (prior to the shift to a turbid ecosystem around 1970), sustainable management would require rehabilitation.

Equity

There is no basic difference between the two different options available for restoring clear water in the lakes, in terms of equity aspects. However, a shift to a clear water ecosystem would, compared to the present situation, increase the supply of two ecosystem services (recreation and biodiversity conservation). Ecosystem services analysis can be instrumental in identifying impacts of ecosystem change on different stakeholders. A brief analysis of how ecosystem services accrue to local versus other stakeholders in the De Wieden case study area is presented below.

People engaging in fisheries and reed cutting are almost exclusively local people, living in the municipality of Steenwijk. The benefits resulting from these two service therefore accrue mainly at the local level. As for recreation, the benefits are distributed over a range of stakeholders present at different scales. There is local industry (hotels, restaurants, boat rental, etc.) owned by, or employing, local people. In addition, there is the benefit of having recreation opportunities for people living nearby and at greater distances from De Wieden. Finally, the nature conservation service is relevant for the local people, but also, given the importance of the area, at the level of the country and above.



Source: Hein et al. (2006).

Figure 5.11 The relation between institutional scale and the value of ecosystem services

The value of the four services supplied by the area has been assessed in order to reveal how the benefits accrue to stakeholders at different scales. For reed cutting and fisheries, it is assumed that all benefits are generated at the level of the local municipality. For recreation, the benefits for the tourist industry are assumed to be generated locally, but the consumer surplus accruing to visitors has been attributed to different spatial scales as a function of the place of residence of the visitors, as recorded with a survey (Van Konijnenburg, 1996). The value of the nature conservation service at different scales is estimated by analysing where the members of the NGO maintaining the site (Natuurmonumenten) are living, distinguishing members in the municipality, the province and in the rest of the country. The value of the nature conservation service at the global scale is not known, although its ecological value is confirmed through its appointment as Ramsar wetland and through its inclusion in the areas designated under the EU Habitat and Birdlife Directives.

Figure 5.11 shows how the values of the four main ecosystem services generated in De Wieden are distributed over four institutional scales. It is clear that the provisioning services benefit local people, whereas much of the benefits of nature conservation and, to a lesser extent, the recreation service accrue to other parts of the country. Hence, the interests of the national and the municipal stakeholders in managing the ecosystem are not aligned.

The analysis indicates that local stakeholders would benefit from restoring the lakes to clear water because they would have increased recreational opportunities nearby, and there may be a positive impact on the local recreation sector. None of the stakeholders would be negatively affected by a shift to clear water. In terms of payments for restoring clear water, the policy mandate resides with the water board, which operates at the level of the province, but a case could be made, based on the analysis, for national financial support for restoring the water quality in the lakes, given the benefits of clear water to two ecosystem services relevant in particular at the national scale (recreation and biodiversity).

5.5 DISCUSSION AND CONCLUSIONS

5.5.1 Uncertainties in the Model

There are several sources of uncertainty in the models, related to: (1) inaccuracy of the data; (2) uncertainty in the model equations; and (3) uncertainty in the threshold value (see also Rotmans and Van Asselt, 2001). These three aspects are briefly discussed below.

Inaccuracy in the data

The ecological data is relatively reliable as all variables (P concentrations, chlorophyll a concentrations, Secchi depth, etc.) can be accurately measured, and because the available data set is large. However, the costs and impacts of the P control measures are based on expert judgement and are known only by approximation. The level of uncertainty related to the cost figures is not known.

Uncertainty in the model equations

There are considerable uncertainties related to the parameters of several of the empirical equations used in the model. In particular, this applies to the equation relating phosphorus loading and phosphorus concentration, which was based upon Hosper (1997). With a t-value of 1.7, this equation is significant only at the 0.15 level (following Blalock, 1987). Another main source of uncertainty is the Poole–Atkins coefficient (PA) assumed for De Wieden in Equation (5.7) ($PA = 1.5$). The PA of shallow lakes in the Netherlands varies from around 1.0 to 2.1 (Hosper, 1997), and it is not certain how well the average PA of four nearby lakes of comparable size represents the PA of the De Wieden lakes.

Uncertainty in the threshold value

A sensitivity analysis has been conducted for the threshold at which the switch to a clear water system occurs (Equation (5.9)). In the model, the threshold was set at 25% cover of the lake bottom with waterplants. At a cover exceeding 25%, the whole lake would develop into a clear water system (cf. Meijer, 2000). The results of the sensitivity analysis, for thresholds of

Table 5.4 Sensitivity analysis for the threshold level

Factor	**Critical plant cover**		
	20%	**25%**	**30%**
Without biomanipulation			
Reduction of P loading required (ton total-P)	9	10	12
Critical P concentration (mg/l)	0.04	0.03	0.02
Minimum yearly benefits that justify transition to clear water (million euro)	1.8	2.0	2.8
With biomanipulation			
Reduction of P loading required (ton total-P)	0	2	5
Critical P concentration (mg/l)	0.11	0.09	0.08
Minimum yearly benefits that justify transition to clear water (million euro)	0.1	0.5	0.8

20% and 30%, are shown in Table 5.4. Table 5.4 shows that the threshold value has a substantial impact on the critical P concentration and the value of the benefits required to justify a policy aimed at obtaining clear water.

Clearly, the uncertainties of the analysis are substantial, with a main factor being the threshold level, and the model cannot be used to *predict* the development of the lakes following eutrophication control. However, the model does reflect the mechanisms that occur in a shallow lake ecosystem following implementation of eutrophication control measures. The accuracy of the model and the data is sufficient to demonstrate the implications of an ecological threshold for the formulation of an efficient management strategy. In addition, the model provides an order-of-magnitude estimate of the minimal value of clear water that would – from an economic perspective – justify a strategy aimed at recovering the clear water state of three lakes in De Wieden.

5.5.2 Implications for Ecosystem Management

Management of ecosystems subject to multiple steady states

The presence of alternative steady states has been recognised in a range of different ecosystems (Scheffer et al., 2001). Commonly, the state of the ecosystem determines its capacity to supply ecosystem services (Mäler, 2000; Limburg et al., 2002). This is illustrated by the study of the De Wieden ecosystem. The opportunities for recreation and nature conservation are substantially higher in the clear water state. The supply of two other services, reed cutting and fisheries, is not significantly affected by a switch in ecosystem state at the considered range of nutrient levels.

Rehabilitation of lakes currently in a turbid water state is, from an economic perspective, justified if the benefits outweigh the costs of a switch to clear water. In other words, the increase in the supply of ecosystem services should be at least equal to the costs of the eutrophication control measures. Although biomanipulation is relatively cheap, the costs of reducing the inflow of phosphorus are often high (see also Carpenter and Cottingham, 1997; Mäler, 2000). Because the increased supply of ecosystem services strongly depends upon the state of the ecosystem, these benefits are only obtained following a switch to a clear water state. The implementation of measures that do not lead to such a shift is, therefore, not cost-effective. Hence, the presence of two multiple steady states forces the ecosystem manager that wants to implement an economically efficient management strategy to make a choice on which state to pursue. In the case of De Wieden, there are basically two options, each represented by a local maximum in Figures 5.9 and 5.10: maintaining the current state (low benefits, no costs) or rehabilitating the ecosystem (higher benefits, higher costs). The choice between the two options depends upon the ratio between the costs of the measures, and the benefits of the increased supply of ecosystem services.

Implications for the management of De Wieden

A number of specific recommendations for De Wieden can be formulated. First, it is more cost-effective to rehabilitate the lakes through a combination of reducing the inflow of phosphorus and biomanipulation, compared to an approach in which only phosphorus loading is reduced. This conforms to the current experiences with the rehabilitation of shallow lakes in the Netherlands (RIZA, 1997; Meijer, 2000). Second, it is probably necessary to reduce the phosphorus concentrations in the lakes to allow biomanipulation to be successful. The model indicates that a reduction from the current 0.10 to 0.09 mg total-P/l would be required, based upon a threshold value of 25% water plant cover. However, as the uncertainty in the model is high, further research is required to confirm this recommendation (see Jeppesen et al., 1990). Third, the annual benefits of a switch to clear water in the three selected lakes would have to be valued at around 0.5 million euro per year to economically justify implementation of the eutrophication control measures. In comparison, the current annual expenditure of the NGO Natuurmonumenten for the management and conservation of biodiversity in De Wieden amounts to 1.5 million euro (Natuurmonumenten, 2000) and the annual net benefits provided by the four lakes are around 4.5 million euro per year (see Table 5.1). In view of the substantial uncertainties in the response of the ecosystem to eutrophication control measures, the waterboard could, following reduction of the phosphorus inflow, consider the gradual application of biomanipulation starting with the shallowest lake (i.e. the Schutsloterwijde) where the chance of success is highest

(RIZA, 1997; Meijer, 2000). If this is successful, biomanipulation could be extended to the other lakes.

National and EU policies

The study indicates that the current Dutch policies, as expressed in the 'Fourth National Policy Document on Water Management' (VW, 1998), may not be efficient. These policies aim at a reduction of total-P concentrations in lakes classified as important for nature conservation to 0.05 mg total-P/l. However, at low total-P concentrations, the supply of the ecosystem services 'recreation' and 'nature conservation' depends upon the transparency of the water rather than on the total-P concentrations. Furthermore, where biomanipulation is applied, it will in many lakes be possible to achieve clear water at total-P levels above 0.05 mg/l (Meijer, 2000). Jeppesen et al. (1990) indicate that biomanipulation can be applied at a total-P concentration of 0.08–0.15 mg/l, depending upon lake characteristics. For Dutch lakes, in general, clear water is achieved much more cheaply through biomanipulation than through reduction of total-P concentrations only (Hosper et al., 1992; Klinge et al., 1995; RIZA, 1997). Therefore, it is more cost effective to enhance water quality through reducing nutrient loading in combination with the application of biomanipulation, than through setting a standard for total-P concentrations only.

In policies regarding water quality, the Dutch government could consider increasing the allowable total phosphorus concentration in waters with the main function 'nature' to 0.08 mg/l (which should in most cases be sufficient to reach clear water through biomanipulation), and combine this norm with a norm for water transparency. For instance, a summer visibility norm of 1 m would allow the establishment of water plants up to a depth of around 1.5 m, depending upon lake characteristics such as substrate and wind exposure. Considering that the large majority of Dutch lakes are very shallow, with average depths of 1.5 to 2 m (Hosper, 1997), this norm would result in clear water in a substantial number of the Netherlands' lakes. Since it appears that biodiversity and recreation are a function of water transparency rather than total-P levels (at least within the examined range of total-P values), this would bring greater benefits at lower costs compared to the current 0.05 mg total-P/litre norm.

The findings are also relevant for the implementation of the EU Water Framework Directive that requires member states to specify water quality standards as a function of the uses of the river, i.e. the ecosystem services supplied by the river. For waters identified as being important for recreation and nature conservation, whether visibility can be included as water quality indicator should be examined, as well as whether overly strict targets for total P loads should be avoided.

6. Case study: rangeland management in the Ferlo, Senegal

6.1 INTRODUCTION

In this chapter, the ecological–economic modelling approach described in Chapter 3 is used to analyse grazing strategies in a semi-arid rangeland in Senegal. Semi-arid rangelands are the vast tracts between deserts and the agricultural zones where rainfall is generally too low or unreliable for cropping, and where livestock keeping is the most important source of income (Walker and Noy-Meir, 1982; Walker and Abel, 2002). In recent decades, a range of models have been developed that aim to provide guidance on how to maximise income from livestock keeping in semi-arid zones while maintaining the natural resource basis. This has been a particularly urgent question for the Sahel, which has around 70 million pastoralists that are food insecure in years of drought (FAO, 2001). Furthermore, climate change may lead to a reduction of rainfall in the Sahel in the coming decades (Held et al., 2005), making it even more urgent to understand the sustainability and economics of Sahelian pastoralism.

Rangelands have been modelled for several decades. Initial economic rangeland models comprised simple ecosystem dynamics that implicitly assumed exogenous forage production of the rangeland (e.g. Dillon and Burley, 1961; Hildreth and Riewe, 1963; Walters, 1968). In subsequent years, these models were adjusted to better reflect management strategies of pastoralists and/or ecosystem dynamics. For instance, McArthur and Dillon (1971) present a stochastic single period model with a risk averse manager, and Karp and Pope (1984) and Rodriguez and Taylor (1988) present dynamic models that consider the effects of the stocking rate on forage production. In the last 20 years, ecological research has shown that rangelands tend to be subject to multiple stable states as well as stochastic responses to rainfall variability (e.g. Friedel, 1991). In response, a range of more sophisticated rangeland management models has been developed. For example, Perrings and Walker (1997) present a model for rangelands driven by fire and grazing that allows for multiple states, and Janssen et al. (2004) model the productivity of a rangeland dominated by shrubs and perennial grasses as a function of two control variables, stocking rate and fire.

A major issue in rangeland modelling is how to account for the simultaneous impacts of stochastic rainfall and the feedback effect of grazing on vegetation production and the efficiency of different grazing strategies (Fernandez-Gimenez and Allen-Diaz, 1999; Briske et al., 2003). Annual rainfall determines the year-to-year variation in rangeland productivity, whereas the long-term stocking density determines the composition and density of the plant cover, which, in turn, determines biomass production under certain rainfall conditions (Le Houérou, 1984). Most of the Sahel is dominated by annual rather than perennial grasses, and the rangeland models should be adjusted to this characteristic of the Sahel. A crucial difference between annual and perennial vegetation is that, during the dry season, the standing biomass in annual vegetation is carried over to the next year in terms of soil nutrients rather than plant biomass. This leads to much larger interannual variations in the species composition of plant communities compared to rangelands dominated by perennial grasses.

In this chapter, a model for semi-arid rangelands dominated by annual grasses is developed that accounts for stochasticity in rainfall, as well as the feedback of high grazing pressures on the vegetation (see also Hein and Weikard, 2008). The model is developed for the Ferlo semi-arid rangeland in northern Senegal. In line with recent ecological insights, the model assumes that the impacts of high grazing pressure on the vegetation of the Ferlo cannot be reversed by a few years of low grazing pressure (cf. Le Houérou, 1984; Walker, 1993; Ludwig et al., 2001; Walker and Abel, 2002). Instead, how the livestock grazing regime will influence the long-term capacity of the land to produce animal feed is examined.

The decision variable of the model is the long-term stocking density applied by the pastoralist society (cf. Walker, 1993; Batabyal et al., 2001). This chapter follows a simulation modelling approach, where the differential equations governing rangeland dynamics and pastoralists' income are captured in a dynamic systems model. Subsequently, the model is run for a range of long-term stocking rates in order to produce a simple profit function and determine the stocking rate that provides the highest profits to the pastoralist society. Using an algebraic optimisation approach, the first order conditions for economic efficient grazing can also be determined. This approach, and the resulting conditions for efficient grazing strategies are not included here, but can be found in Hein and Weikard (2008).

The model calculates an efficient stocking rate for the pastoralist society as a whole. However, in reality, the problem of the commons is a key factor in rangeland management. Stocking decisions in northern Senegal are made by several tens of thousands of pastoralists who do not act as one profit-maximising entity. In addition, profit maximisation is only one of their decision-making criteria; their strategy will also be aimed at, for instance, reducing risks. This chapter shows how the economic efficient grazing stocking rate can be determined, rather than the optimal strategy from the perspective of the individual pastoralist. Hence, the model is useful to support policy formulation on rangeland management, rather than explaining the behaviour of individual pastoralists, as further discussed in Section 6.5.

The model and the case study are based on ecological data from a 10-year grazing experiment conducted in the Ferlo, Senegal (1981–1990) (Miehe, 1997). These data reveal the joint impact of rainfall variability and long-term stocking rate on rangeland productivity. The study does not address the issue of multiple states in semi-arid rangelands (Friedel, 1991; Walker and Abel, 2002). The grazing experiments show that the grazing regimes currently applied in the Ferlo are unlikely to lead to a switch to a new rangeland state with the exception of the zones around boreholes. This is further elaborated in Section 6.5.

6.2 THE CASE STUDY AREA

The study area is the Ferlo Region, Northern Senegal (Figure 6.1). The natural vegetation consists of dry grassland with scattered trees and bushes. The herbaceous layer comprises a mix of grasses, leguminous species and other plants. While both annual and perennial species occur in the Ferlo, annual species strongly dominate the herbaceous layer (Breman and De Ridder, 1991; Hein, 2006b). Soils are mainly of aeolian origin and

Figure 6.1 Location of the Ferlo

are predominantly sandy, with variable but generally small amounts of loam and clay. Annual rainfall varies between around 120 and 450 mm, with an average of 291 mm. The rainy season lasts only three months, from July to September (Andre, 1998).

With an average population density of around five people per km^2, the total rural population of the Ferlo can be estimated at around 110 000 people (Direction de la prévision et de la statistique, 1997). Livestock keeping is the main economic activity in the Ferlo and it is essential for local food security. The principal animals kept are cattle (Zebu), sheep and goats. The rangeland also provides a number of other ecosystem services, such as timber and fuel wood, fibres for making baskets, medicinal plants, wildlife, etc. The most important off-farm activities are the selling of crafts (2% of income) and the collection and sale of arabic gum (3% of income) (Sutter 1987). Hence, the contribution of these other ecosystem services to the local income is small, and this chapter focuses on the provision of fodder for cattle grazing.

While there is considerable variation among the local pastoralists as a consequence of family size and composition, herd size, etc., transhumance remains the most common production system among the Fulani (Adriansen and Nielsen, 2002). Families spend the wet season in the Ferlo, with the herds feeding on the green pastures and water being provided by ponds or boreholes. In recent decades, a substantial number of new wells have been constructed in the Ferlo, drawing water from deep aquifers, and perennial water sources are now available throughout the Ferlo (Ministère de l'Hydraulique, 1987). The drilling of the boreholes, in combination with government policies aimed at settling local people in order to facilitate their incorporation in the administrative system, caused a concentration of grazing pressures and led to the creation of denuded zones around the wells (Sinclair and Fryxell, 1985).

During the dry season ponds dry out, but water is still provided by the boreholes, unless there is a pump failure. Feed resources strongly decline during the dry season, and many of the pastoralists migrate southwards to the more humid Sudan zone, where fallow lands and crop residues provide food for the animals and where more perennial water resources are available (Sutter, 1987; Breman and De Ridder, 1991; Adriansen and Nielsen, 2002). However, since the early 1990s, there has been an expansion of agricultural activities in the Sudan zone, which increasingly limits the possibility for pastoralists to migrate south in the dry season (Guerin et al., 1993; Adriansen, 2006).

In recent years, there has been an increase in the number of small ruminants, which are more drought tolerant. In addition, there has been a trend to increased connection with the livestock markets, with the Fulani becoming more actively engaged in selling and buying livestock on local and regional markets (Adriansen, 2006). Currently, average livestock densities in the Ferlo are in the order of 0.15–0.20 Tropical Livestock Units (TLU) per hectare (De Leeuw and Tothill, 1990; Miehe, 1997). A TLU is an indicator that has been developed to measure livestock production and that corresponds to 250 kg of animal weight (Boudet, 1975). In the Ferlo, a Zebu cow equals on average 0.73 TLU and a sheep or a goat is around 0.12 TLU (Boudet, 1975).

6.3 THE ECOLOGICAL–ECONOMIC MODEL

6.3.1 Dynamics of Semi-arid Rangelands

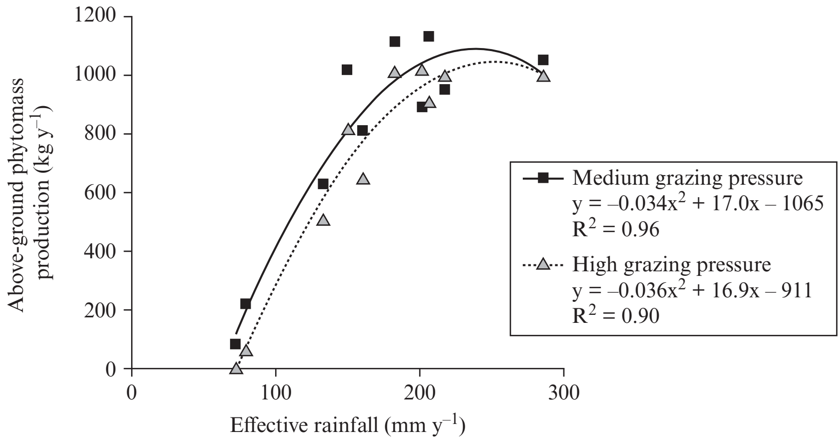
Early models of rangeland dynamics were based upon the Clementsian theory of ecological succession (Clements, 1916; Weaver and Clements, 1938). These models assumed that succession to a climax is a steady process that can be reversed by grazing, drought, fire or other disturbances. A given stocking rate was generally assumed to result in an equilibrium state of the vegetation (Walker, 1993). These early models were not capable of adequately predicting impacts of changes in rangeland management (Walker, 1993) and new insights in rangeland dynamics have emerged. These new models encompassed such concepts as multiple steady states, stochasticity in rangeland dynamics and irreversible responses to stress (Westoby et al., 1989; Friedel, 1991).

Whereas the understanding of rangeland dynamics has greatly increased in recent years, the impact of high grazing pressures on rangelands is still strongly debated. A number of authors state that plant and animal dynamics are largely independent of one another and that high grazing pressures do not have a significant long-term impact on the composition and functioning of the rangeland. In this view, rangeland development is largely driven by year-to-year variation in abiotic drivers, primarily rainfall (Ellis and Swift, 1988; Scoones, 1994; Sullivan and Rohde, 2002). However, others stress that high grazing pressures do have an impact on the ecosystem, particularly in the medium and long term, and may affect composition, functioning and productivity of the ecosystem (Le Houérou, 1984; Sinclair and Fryxell, 1985; Illius and O'Connor, 1999; Fynn and O'Connor, 2000). It has also been shown that the impacts of a high grazing pressure can strongly vary between different rangelands (Fernandez-Gimenez and Allen-Diaz, 1999). The two approaches to rangeland dynamics relate to, respectively, the 'non-equilibrium' and 'equilibrium' paradigms in ecology (Wiens, 1984; Sullivan and Rohde, 2002). In recent years, the insight has emerged that understanding rangeland dynamics requires both concepts, with the applicability of equilibrium or non-equilibrium models dependent on the temporal and spatial scale under consideration, the amount of stress the rangeland is exposed to and the specific characteristics of the rangeland (Briske et al., 2003).

Rain-use efficiency (RUE) is a central concept in analysing rangeland dynamics. RUE is expressed in kg dry weight biomass ha^{-1} mm^{-1}, and is calculated by dividing above-ground phytomass production with the annual rainfall. Hence, the RUE efficiency expresses the capacity of the vegetation to use water for NPP. Note that there is a difference between rain and water availability to plants in semi-arid sites, as an important part of the rainfall may evaporate or run-off. The amount of rainfall that is available to plants is called the effective annual rainfall, which is a better indicator of the amount of rain available to plants than total annual rainfall (Snyman, 1998). The ratio effective/total rainfall varies between years depending on the rainfall pattern. In the Sahel, a key aspect is how much rain falls outside of the growing season, before the plants have germinated or after they have wilted. Since the germination of plants depends on the vegetation composition, and runoff depends upon the site-specific interception, infiltration and surface storage rates, the RUE is an indicator of the degradation of the rangeland. In general, degraded rangelands have a lower RUE because of a lower plant cover and root system and/or because the soil has been compacted and run-off is increased.

For the Ferlo, in the period 1981–1990, the effective rainfall appears to be, on average, around 65% of the total rain (Miehe, 1997). There are large differences between the various years, effective rainfall varies from 53% of total rainfall in 1990 to 72% of total rainfall in 1983. The RUE of the herb layer in the Ferlo under both grazing regimes, plotted as a function of effective rainfall, is presented in Figure 6.2. A second order polynomial has been fitted through the observed values. Regression analysis showed this fit to be significant at the $p < 0.05$ level, whereas neither linear nor *S*-curve relations could represent the measurements in a statistically significant way. The curve demonstrates that RUE is highest at an effective rainfall of around 180 to 200 mm. This corresponds well with the actual amount of effective rainfall in the study site. Assuming that also in the long term 65% of the average annual rainfall (290 mm) can be considered as effective rainfall, the long-term average effective rainfall in the area is around $0.65 \cdot 290 = 189$ mm per year.

A quadratic function relation between RUE and rainfall has also been found to represent the best fit to datasets for several other African and other semi-arid rangelands (e.g. O'Connor et al., 2001; Hein and De Ridder, 2006), but its general validity is still being debated. However,



Notes: See also Hein (2006b).

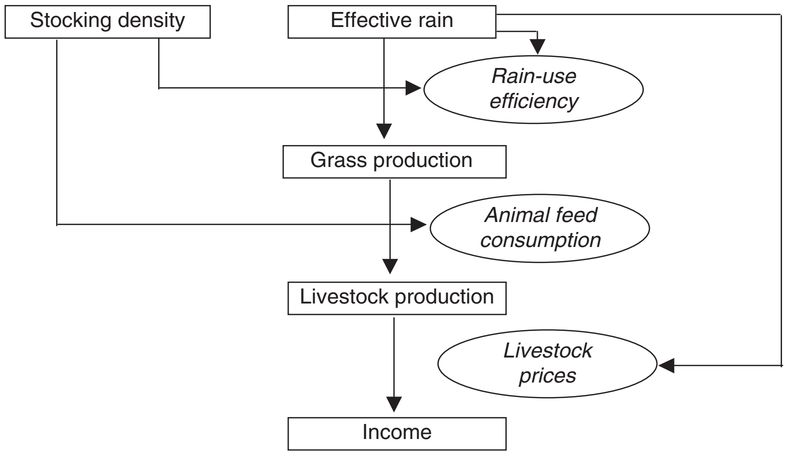
Figure 6.2 RUE as a function of effective rainfall, for sites with medium (0.10 TLU ha^{-1}) and high (0.15–0.20 TLU ha^{-1}) grazing pressure

since the ecological–economic model developed in this chapter concerns the Ferlo, and the data for the Ferlo point to the existence of this quadratic relation in this area, it is used as a basis for modelling the rangeland dynamics in the following sections.

Figure 6.2 shows that the RUEs of plots with medium and high grazing pressure do not markedly differ for *normal* and *wet* years – defined as having at least 150 mm of effective rain (Andre, 1998). However, for the dry years, there is a substantial difference between the plots. The differences between the two types of plots have also been tested with a matched-pairs test, which show that for the overall period 1981–1990, the difference in RUE between plots with medium and high grazing pressure is significant at the $p = 0.05$ level (Hein, 2006b). Hence, high grazing pressures have a significant impact on the RUE of the herbaceous layer of the Ferlo, but this impact is concentrated in the dry years. This mechanism is incorporated in the model developed for the Ferlo in the next section.

6.3.2 Model Structure and Equations

An ecological–economic model has been constructed on the basis of local data on vegetation production under different grazing densities and rainfall conditions. These data were collected during a 10-year period



Source: Hein and Weikard (2008).

Figure 6.3 Structure of the ecological–economic model

(1981–1990) in the Widou-Thiengoly research station in the western part of the Ferlo (see Figure 6.1), as reported in Klug (1982), Miehe (1992, 1997) and Andre (1998).

The model is a dynamic systems model, running with time increments of one year. The model calculates the annual income for the pastoralists as a function of annual rainfall and the long-term stocking rate maintained by the pastoralists. The model is spatially homogeneous, and changes in livestock routes or other adaptation strategies not involving changes in stocking densities are not considered. The structure of the model is shown in Figure 6.3. The different steps are explained in detail below. All prices are expressed in CFA, the West-African Franc.[1]

Input variables

The model is based on the assumption that it is not water, but grass biomass that is the critical limiting factor for livestock grazing in the Ferlo. The reason is that, in recent decades, a large number of boreholes have been constructed in the area and drinking water for livestock is now generally also available in the dry season. However, in the dry season, grass resources are becoming increasingly scarce and, during drought years, animals are suffering from a lack of feed (Ministère de l'Hydraulique, 1987; Adriansen, 2006). During a drought, feed resources are also difficult

[1] On 1 June 2006, 1 US$ = 530 CFA.

to access by migrating the herds south because of the agricultural expansion that has taken place here and because, in times of drought, fewer crop residues are available for animal feed (Guerin et al., 1993; Adriansen, 2006).

The model contains two input variables: effective annual rainfall and long-term stocking density. Measurements by Miehe (1997) have shown that the average annual rainfall of 291 mm year^{-1} results in an effective annual rainfall of 190 mm year^{-1}. The effective annual rainfall is the rain not lost through evaporation or run-off, providing a better indicator of the water availability for plant growth. The model uses a 50-year simulation of annual rainfall, including its variability, based on the actual annual rainfall variability during the period 1961–1990 (as reported in Andre, 1998). Long-term stocking density is a key management variable for semi-arid rangelands (Batabyal et al., 2001; Briske et al., 2003). While pastoralists decide on a year-to-year basis how they move their herds in a spatially heterogeneous rangeland, the long-term stocking density is the key driver for the development of the structure and composition of the plant cover over time (Le Houérou et al., 1988; Walker 1993; Vetter, 2005). The herds in the model comprise a mix of cattle, goats and sheep, and the size of the herd is expressed in Tropical Livestock Units (TLU, see above).

Rain-use efficiency

Rain-use efficiency indicates the effectiveness of vegetation to transfer rain into biomass (Le Houérou, 1984). Rain-use efficiency is normally expressed as the amount of biomass produced per hectare per year per mm of (effective) rain. Analysis of 10 years of grazing data in the Ferlo (Hein, 2006b) demonstrated that RUE is a quadratic function of rainfall (cf. O'Connor et al., 2001). RUE appears to be relatively high at intermediate levels of rainfall, and RUE declines in years of drought and in years with high rainfall (Hein and De Ridder, 2006). In addition to rainfall, the RUE is also affected by long-term grazing pressure. This reflects that a few years of high grazing pressure have limited impacts on the vegetation, but that sustained high grazing pressure leads to changes in the ecosystem (species composition, cover, etc.) (Le Houérou et al., 1988). Impacts of ecosystem changes on the productivity are expressed through changes in the RUE. Specifically, high long-term grazing pressure shifts the quadratic curve downwards, reducing the RUE and, consequently, herbaceous biomass production, for each amount of rainfall (see for details: Hein, 2006b; Hein and De Ridder, 2006). In the model, the following formula is used:

$$\rho = [\alpha r^2 - 2\alpha R \cdot r + \beta] - [(s^o)^\vartheta(\mu r^2 - 2\mu \bar{r} \cdot r + \nu)] \qquad (6.1)$$

where ρ denotes the rain-use efficiency (kg ha^{-1} mm^{-1}), r the effective annual rainfall (mm $year^{-1}$), R the average rainfall, and s^o the long-term stocking density (TLU ha^{-1}). α, β, μ and ν are scaling parameters, which are estimated on the basis of the grazing data. The left-hand part of the equation is a simple quadratic function showing the relation between rain-use efficiency and rainfall without grazing (cf. O'Connor et al., 2001; Hein and De Ridder, 2006). The right-hand part of the equation shows how the long-term grazing pressure affects the rain-use efficiency by moving the curve towards a lower rain-use efficiency, while maintaining a quadratic relation (cf. O'Connor et al., 2001). Further explanation of the function is provided in Hein and De Ridder (2006) and Hein and Weikard (2008).

Grass production

Grass production is the product of annual effective rainfall and rain-use efficiency. Note that this implies that F is a third power function of the rainfall (cf. Le Houérou et al. 1988; Palmer 2000). It can be specified as follows, with F denoting the grass production (above-ground NPP, expressed as kg ha^{-1} $year^{-1}$):

$$F = \rho \cdot r \tag{6.2}$$

Livestock production. A pasture's annual grass production F can be translated into the annual grazing capacity s_{max} (Hildreth and Riewe 1963). Let φ be the amount of plant biomass required to allow the subsistence of a livestock unit. Then

$$s_{max} = \frac{1}{\varphi}F \tag{6.3}$$

In the model, it is assumed that pastoralists decide on the stocking rate at the end of the rainy season, i.e. when they have full oversight of the animal feed resources they can expect in the coming year. The actual stocking density in a year, s_t, depends on s_{max} in relation to the selected long-term stocking density, s^o. In years with sufficient rainfall ($s_{max} > s^o$), the pastoralists maintain s^o. In these years, the reproduction of the stock leads to surpluses ($s_t - s^o$), which are sold on the market. However, in years where animal feed resources are insufficient due to low rainfall, the pastoralists maintain only as many animals as can be supported by the grass production in that year: s_{max}. The animals that can not be fed are sold on the market. This is in line with the actual strategies that can be observed in the Ferlo. In years of drought, pastoralists maintain as many animals as possible on the limited grass resources available in order to be able to restock as quickly as possible after the drought, and in years of abundant rainfall, the surplus is sold on the local markets (Guerin et al., 1993). Hence:

$$s_t = Minimum\ (s^o, s_{max}) \tag{6.4}$$

The growth of the livestock herd is assumed to follow a logistic growth process:

$$\Delta s = \lambda\left(1 - \frac{s_t}{s_{max}}\right) \cdot s_t \tag{6.5}$$

where s_t is livestock in the current year, Δs is the gain in livestock, s_{max} is the grazing capacity of the rangeland determined by the annual rainfall and λ is a scaling parameter capturing the potential natural growth in livestock. λ is always greater than zero. Note that during droughts, when s_t equals s_{max} (see Equation (6.4)), there is no gain in livestock. This reflects that during drought the net reproduction rate of the animals is zero.

Pastoralists income

For Senegalese pastoralists, the main source of income is the sale of animals for meat, with milk production coming in second place (Sutter, 1987; Guerin et al., 1993). The role of milk production and agriculture has decreased in recent decades with the increased focus on livestock herding for meat production (Adriansen, 2006). Off-farm income is relatively unimportant in the Ferlo. The most important off-farm activities are the selling of crafts (2% of income) and the collection and sale of arabic gum (3% of income) (Sutter 1987). Therefore, for reasons of simplicity, in the model it is assumed that income is only derived from the sale of animals. Income depends upon the amount of surplus livestock that can be sold annually on the market, as well as on the livestock price. The amount of livestock that can be sold or bought equals the stock in the previous year (s_{t-1}) plus the growth in livestock (Δs) minus the stocking density that is maintained (s_t). The amount of animals the pastoralists sell (positive) or buy (negative) (S) equals:

$$S = s_{t-1} - s_t + \Delta s. \tag{6.6}$$

In years of drought, when there is not enough grass production to feed the livestock number corresponding to the long-term stocking rate, it is assumed that the pastoralists maintain the amount of livestock that can be fed and that they sell the surplus on the market. However, the price will be low, as there will be a high supply, and low demand on the local markets. If a dry year is followed by a wet year, the pastoralists will purchase livestock in the market in order to stock up to the long-term stocking rate. In years subsequent to a drought, prices will generally be high. The profit-function of the pastoralist society is:

$$\pi = S \cdot p - c \cdot s, \tag{6.7}$$

where π is the profit of the pastoralists per hectare and p the price per livestock unit. The factor $c{\cdot}s$ indicates the variable costs per livestock unit (e.g. interest and veterinary services). It is assumed that, in the Ferlo, there are no fixed costs (such as land taxes or fencing, which are virtually absent in the Ferlo).

6.3.3 Data

The parameters in Equation (6.1) were derived through regression analysis on the basis of the grazing data from the Widou-Thiengoly research station covering the period 1981–1990 (see Hein, 2006b). The empirical parameter settings are presented in Table 6.1. The main outcome of the grazing data was that high grazing pressures have relatively little effect during years with normal or high rainfall, but lead to a strong reduction in grass production during droughts (Hein, 2006b).

For Equation (6.3), the amount of plant biomass required to feed one TLU during one year (φ) has been estimated on the basis of the local livestock mix (Thébaud et al. 1995) and the energy requirements per animal (Bayer and Waters-Bayer 1998). The minimum amount of feed that the animals need to maintain themselves is estimated at 4.3 kg TLU^{-1} day^{-1}. Not all herb biomass is available to the animals, due to decomposition, fire or the unpalatability of certain plants. Penning de Vries and Djitèye (1982)

Table 6.1 Parameters for the relation between rainfall, grazing and rain-use efficiency, used to calibrate equation 1 ($n = 30$, $F = 28$)

Parameter	Value
α	−0.00021
β	−1.254
μ	0.00504
ν	210.8
θ	2.0
R (average effective rainfall)	189 mm

Source: Hein and Weikard (2008) and Hein (2006b).

and Breman and De Ridder (1991) show that, in the Ferlo, 50% of plant biomass is available for grazing. The dietary contribution of woody plants to the overall feed supply is estimated at 20% (Breman and De Ridder, 1991). Hence, it is estimated that φ equals $4.3 \cdot 365 \cdot 2 \cdot 0.8 = 2511$ kg herb biomass $\text{TLU}^{-1}\,\text{year}^{-1}$.

The livestock population grows according to a logistic growth curve (Equation (6.5)). Specification of this curve requires estimation of the growth factor of livestock, λ. Boudet (1975) and Mortimore and Adams (2001) estimate a maximum natural growth of herd size of around 20% per year for the western Sahel. It is assumed that this also holds for the Ferlo. This growth rate corresponds to a logistic growth factor λ of 0.6.

The final parameters to be estimated in the model relate to the price of livestock, p, and the variable costs related to livestock, c. Based upon the average price per animal in the Ferlo and the local livestock mix (Thébaud et al., 1995), the average livestock price is assumed to be 24 750 CFA TLU^{-1}. In the Ferlo, livestock prices decrease during a drought, as many farmers want to sell livestock that they cannot feed. Immediately after a drought, livestock prices increase substantially as farmers want to restock (Turner and Williams, 2002). For the Ferlo, data on price fluctuations were not available. Therefore, as the best proxy available, data on price fluctuations from western Niger have been used. This area has a slightly higher average rainfall but otherwise represents a comparable physical, economic and social environment. The production system is also based on transhumant livestock ranging, and the local population is also dominated by Fulani. Based on these data, it is assumed in the model that prices drop to 43% during years with a drought and that they increase to 146% in the two years subsequent to a drought.

Regarding the costs of livestock herding in northern Senegal, it is assumed that all costs are variable costs, related to capital and labour inputs required to maintain the herd. The capital costs per livestock unit amount to the local, real interest rate times the price of a livestock unit. Currently, the average local interest rates are around 18% (Ndour and Wané, 1998), and the annual inflation in Senegal is approximately 2% (IMF, 1999). The capital costs are, therefore, $0.16 \cdot 24\,750 = 3960$ CFA TLU^{-1}. The average labour costs in rural Senegal are estimated at 100 000 CFA per person per year (Direction de la prévision et de la statistique, 1997). These costs only incur during the period January–June when the herds are moved south (Guerin et al., 1993), as during the rainy season herds are taken care of by various family members, including children, at no cost. With an average herd size of 44 TLU per family (Thébaud et al., 1995), the annual labor costs amount to $100\,000/44/2 = 1140$ CFA TLU^{-1}. Therefore, the total variable costs are $3960 + 1140 = 5100$ CFA TLU^{-1}.

6.4 RESULTS

The results of the model calculations are presented in the form of a profit function. The profit function indicates the surplus that pastoralists gain, valuing inputs including labour and outputs at the applicable market rates. In the model, all inputs are assumed to have constant prices, but variable livestock prices have been accounted for. The implications of these assumptions are elaborated in Section 6.5. The functions present the average annual profits over the 30-year period of the simulation as a function of different long-term stocking rates, see Figure 6.4. The model assumes that the long-term stocking rate is constant throughout the 30-year period (but the actual stocking rate, s_t, may be lower during years of drought). The profit function indicates both the long-term stocking rate where the pastoralist community as a whole obtains maximum profits, and the open access equilibrium stocking rate, where profits (but not income) approach zero. In the case of an open-access situation, where each resource user maximises individual profit, profits for society at large tend to decline to 0. The open access stocking rate is indicated by the point where the profit function crosses the vertical axis.

Figure 6.4 shows maximum profits of 450 CFA ha^{-1} year^{-1} at a stocking density of 0.09 TLU ha^{-1}. Note that 0.09 TLU ha^{-1} is consistent with the optimal stocking density of 0.1 TLU ha^{-1} suggested by Boudet (1975) for the Ferlo. It also shows the stocking density where no profit is made, around 0.17 TLU ha^{-1}. This corresponds well with the current stocking in the Ferlo, which is estimated at 0.15 to 0.20 TLU ha^{-1} (De Leeuw and Tothill, 1990; Miehe, 1997). Hence, the calculations indicate that the Ferlo is currently managed as an open access, common property resource. This is in line with Le Houérou (1989) and Guerin et al. (1993), who state that while the migration routes of pastoralists are determined by a complex set of institutions, there is little institutional control of the amount of animals

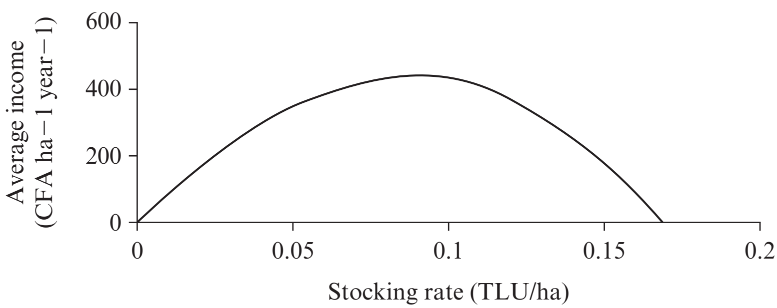


Figure 6.4 Ordinary profit function for the Ferlo

that each pastoralist keeps. Note that, although current profits from livestock keeping appear to be close to zero, the income that pastoralists gain at the current stocking density (the reward for their labour) is around 1140 per TLU · 0.17 TLU ha^{-1} = 194 CFA ha^{-1}.

The stocking rate that provides maximum profits to society as a whole, i.e. the economic efficient stocking rate, has been analysed with an algebraic optimisation approach in Hein and Weikard (2008). As should be expected, this optimisation approach results in the same value of 0.09 TLU ha^{-1} for the efficient stocking rate, generating maximum profits of 450 CFA ha^{-1}.

A critical aspect of analysing ecosystem management options with a profit function (or with algebraic optimisation) is the availability of alternative employment for the people using the ecosystem to earn a living. The use of actual labour costs in the society's profit function is not justified if there are few alternative local income earning opportunities, in which case it is unrealistic to assume a constant price of labour. In the Ferlo, few alternative employment options are available, and this aspect is further discussed in the section below.

6.5 DISCUSSION AND CONCLUSIONS

6.5.1 The Reliability and Limitations of the Modelling Approach

The key assumptions underlying the model are that (1) rain-use efficiency varies with effective rainfall according to a quadratic function (cf. O'Connor et al., 2001; Hein and De Ridder, 2006); and (2) grazing affects the rain-use efficiency of the vegetation, in particular in the long term (cf. Le Houérou, 1989). Consequently, rainfall determines yearly fluctuations in productivity, and grazing pressure affects the long-term productivity. Both relations are non-linear, and the impact of grazing on productivity is most pronounced in years with low rainfall.

The model presented in this study is subject to five key constraints. First, the model does not explicitly include the impacts of fire, which is an additional important driver in many rangelands (e.g. West, 1971; Walker, 1981; Perrings and Walker, 1997; Snyman, 1998). In particular, the model ignores that fire may have a strong impact on the availability of woody plant biomass for browsing. However, the impact of fire on the availability of herbaceous biomass for grazing is accounted for (through modification of the parameter φ). As in the Ferlo only 20% of animal feed is obtained through browsing (Breman and De Ridder, 1991), this simplification has only a small impact on the results of the study.

Second, the model does not account for the spatial heterogeneity of rangelands. Rainfall patterns in the Sahel are not only highly variable per year, but also show strong spatial variability. Pastoralists adapt to this heterogeneity by adjusting their grazing strategy to the local availability of plant biomass. The importance of spatial heterogeneity for calculating optimal stocking rates is elaborated in, for instance, Turner (1999) and McPeak (2003). In the model, accounting for spatial heterogeneity would probably lead to an increase in the optimal stocking density, as the impacts of dry years in one site may be moderated by higher rainfall in other sites. However, in this respect, it is important to note that the two years with the lowest available plant biomass in the period during which the data supporting our model was collected, 1983 and 1984, count as extremely dry throughout the Sahel (Nicholson et al., 1998). This indicates that animal feed availability was very low in most parts of the Sahel, reducing the opportunity to mitigate for drought by adjusting grazing patterns.

Third, the model does not address the risk attitude of pastoralists. In general, pastoralists tend to be risk-averse, and prefer to avoid years with below average income (Anderson and Dillon, 1992; Hardaker, 2000). Accounting for risk-aversion of pastoralists may lead to a reduction of the calculated optimal stocking rate.

Fourth, the model assumes fixed prices for labour. This, however, is not realistic. In the case where there would be a strong decline in livestock grazing in the Ferlo, it is likely that labour costs would go down because there are few alternative sources of income. In general, the point of maximum profits, established at current labour costs, may not indicate the optimal management strategy from the perspective of the local society in the case where there are no or very few alternative employment options. To partly account for this effect, labour costs have been conservatively estimated in this study (see Section 6.3), it is assumed that they account for only 20% of the total variable costs of livestock herding. However, it is likely that a residual impact remains and that the society's optimal stocking rate is higher than predicted by Figure 6.4.

Fifth, the profit function shows the average annual profits, and is as such a very crude metric for optimisation. It does not account for income differences between years, which may be a critical concern for pastoralists since the poor among them may be food insecure in years with low income.

Hence, the model approach illustrates efficient stocking rates in the Ferlo, but needs to be further refined in order to find optimal rangeland management strategies for the pastoralist society. In particular, further refinement is needed in order to account for such factors as risk-aversiveness, lack of alternative income opportunities and a need to minimise food insecure years.

6.5.2 Implications for Rangeland Management

In the Ferlo, as in the Sahel in general, transhumance is the dominant management system. Pastoralists migrate according to specific, seasonal patterns, complemented with more permanent settlements for the less mobile part of the population. The current average stocking rates in the Ferlo are around 0.15–0.20 TLU/ha (De Leeuw and Tothill, 1990; Miehe, 1997). The stocking rate is high relative to other parts of the Sahel with comparable rainfall, due to the presence of a large number of boreholes that provide year-round drinking water for the animals (De Leeuw and Tothill, 1990). The case study shows that the most efficient stocking rate is 0.09 TLU ha^{-1}. Consequently, the optimal long-term stocking rate in the Ferlo is substantially lower than the current stocking rate.

However, the lack of alternative employment opportunities has not been fully considered in the model, and the actual objective function of pastoralists is likely to include other aspects besides income maximisation. Pastoralists in remote areas have limited access to banks and livestock performs the functions of insurance and savings account. Nevertheless, the large difference between the present and the optimal stocking rate indicates that government agencies and development institutes should be focussing on promoting reductions rather than increases in livestock densities in the Ferlo, as any further increases in herd sizes are economically counter-productive. They may also consider improving the functioning of livestock markets, which enhances the capacity of pastoralists to adjust stocking rates to the annual grazing capacities (see Holtzman and Kulibaba, 1994).

It remains to be tested whether the concentration of impacts of degradation in years of drought (Figure 6.2) also occurs in other rangelands. If so, this would have significant repercussions for rangeland management in these areas. In particular, pastoralists or other rangeland managers may in that case underestimate the actual amount of degradation that has occurred in an ecosystem (which is expressed through reduced soil fertility, crusting, changes in plant community, etc.) when looking at ecosystem productivity during wet years or years with normal rainfall. As shown by the case study in the Ferlo, the plant community may have developed under high grazing pressure in such a way that its resilience to cope with drought is affected. Where this effect occurs, i.e. where changes in plant community as a function of high grazing pressure have decreased the ecosystem's resilience to cope with drought, the impacts of future droughts may be particularly severe, and there is a risk that the vulnerability to drought is underestimated by looking at current rangeland productivity and past impacts of drought.

7. Applying the framework in support of environmental management

7.1 INTRODUCTION

This chapter provides a brief overview of how a dynamic systems ecosystem assessment approach can be applied in support of environmental management. Based on the findings of the case studies, the chapter first presents a short review of (1) how the approach can be used to analyse the efficiency, sustainability and equity impacts of environmental management options; and (2) the implications of complex ecosystem dynamics for environmental management. Subsequently, a brief discussion is presented on how the approach may be used in support of a number of environmental management tools, in particular, Environmental Impact Assessment (EIA), Environmental Cost–Benefit Analysis and spatial planning. In line with the focus of the book, the chapter zooms in on environmental management at the scale of the ecosystem, excluding issues at different scales such as climate change and urban environmental issues such as water supply and sanitation.

7.2 ANALYSING ECOSYSTEM MANAGEMENT OPTIONS

7.2.1 Analysing the Efficiency of Ecosystem Management

The economic efficient option is the option that provides maximum net benefits, given a certain objective function, and including the benefits of all services supplied by the ecosystem and the costs involved in providing or accessing these services. Costs include, for instance, the costs of imposing and enforcing maximum harvest levels, or investment, operation and maintenance costs of pollution control measures (e.g. Hueting, 1980). The two important elements in the calculation of the benefits of ecosystem management options are: (1) quantification of the flows of ecosystem services; and (2) ecosystem services valuation. Whereas ecosystem service quantification may be relatively straightforward for the provisioning services, it is often data-intensive for the regulating and cultural services. The appropriate valuation method depends on the ecosystem service under consideration. In the case of large-scale changes in ecosystem services supply that lead to changes in prices for these services, market models need to be developed in order to analyse how consumer and producer surpluses are affected. Valuation may be particularly complex when non-use values have to be quantified, using CVM or related approaches.

For a given discount rate, the efficient management option is the option that provides the maximum Net Present Value (NPV) based upon the current and discounted future flows of net benefits provided by the ecosystem. Alternative criteria in the economic analysis of project or ecosystem management options are the Internal Rate of Return (IRR) or the somewhat more crude measures such as the payback period or the cost–benefit ratio (see OECD, 1995 for guidance on the application of these criteria). The efficiency criterion can be applied to compare the economic returns of two or more ecosystem management options, or to identify the management option that generates the highest net economic benefits, i.e. the '*economic efficient*' management option.

In order to assess the efficient option, two approaches can be applied: (1) a simulation or programming approach; and (2) an algebraic optimisation approach. The simulation approach simulates the development of the ecosystem as a function of the decision variables in order to reveal optimal solutions within the tested range. With the algebraic optimisation approach, optimal solutions are found in an algebraic or numerical manner through the preparation of the Hamiltonian and solving the relevant conditions (Chiang, 1992), or with dedicated algorithms applied in computer software such as GAMS and Mathematica.

The case studies presented in Chapters 4, 5 and 6 provide some general insights in the potential applicability of both optimisation approaches. An advantage of the *algebraic optimisation* approach is that it is more suitable for dealing with stochastic ecosystem behaviour. For example, by applying an algebraic optimisation approach to rangeland management (Hein and Weikard, 2008), the expected values for the income gained from livestock keeping can be derived from a rainfall probability density function. This function can easily be constructed on the basis of an observed rainfall pattern. With the simulation approach used in the main text of Chapter 6, rangeland productivity is modelled based on an observed rainfall pattern. This leads to an error, since the starting year of the simulation run influences the outcomes. However, in the case of the Ferlo case study, the difference between the results obtained with the algebraic and the simulation modelling approach was less than 5%, which is likely to be small compared with the other sources of uncertainty in the analysis.

The *simulation approach* deals more easily with non-linearities in ecosystem behaviour and ecological feedbacks. Solving the first order conditions in the algebraic approach rapidly becomes highly complex, particularly if the costs or benefit functions are non-linear and/or discontinuous. In these cases, it can be very difficult to mathematically solve the first order conditions indicating the efficient level of ecosystem management (cf. Grasso, 1998). In addition, in an algebraic optimisation approach, specific attention is required to deal with sequencing. For example, in the Sahel, it is not unusual to have two or three consecutive years of drought, followed by 5 to 10 years of average or above average rainfall (Andre, 1998; Put et al., 2004). If, in a simulation approach, real climatic data are used, the implications of this sequencing are accounted for in the model runs. The equations used in the algebraic optimisation approach need to be specifically adjusted in order to account for the sequencing effect.

7.2.2 Analysing the Sustainability of Ecosystem Management

In this book, sustainable ecosystem management is interpreted as 'management that maintains the capacity of the ecosystem to provide future generations with the amount and type of ecosystem services at a level at least equal to the current capacity' (based upon WCED, 1987, and Pearce et al., 1989). Ecosystem services include the service 'conservation of biodiversity', which means that maintaining the biodiversity contained in an ecosystem is one of the prerequisites for achieving sustainability. This definition classifies as a 'strong' sustainability criterion (Carter, 2001; Pezzey and Toman, 2002), which facilitates the comparison of sustainability aspects with the efficiency criteria, as elaborated in Section 2.1.3.

A key issue with regards to applying the sustainability concept at the level of the ecosystem is that it is often difficult to define a benchmark state of the ecosystem. For the hypothetical forest ecosystem (Chapter 4), sustainability required the long-term maintenance of the forest and soil cover, which, for this simple ecosystem model, is a sufficient condition to guarantee the sustained supply of the two ecosystem services considered. For the case studies conducted in the De Wieden wetland and the Ferlo semi-arid rangeland, identification of the benchmark for sustainability is less straightforward. In the De Wieden wetland, the present state of the lakes is substantially degraded compared to their natural state. The water of the main lakes is turbid, with an impoverished water plant and fish community, as a consequence of the strong increases in nutrient loading in the lakes since the early 1960s (Van Berkum, 2000). According to the proposed definition, leaving the water quality as it is now would be sustainable. However, if 1960 had been taken as the baseline condition, rehabilitation would be required to reach sustainability. In the case study of the Ferlo rangeland, the same issue applies. Human modification of African savannas started at least 10 000 years ago (Walter, 1971; Walker and Noy-Meir, 1982). In addition, there are large interannual variations in the state of the system due to rainfall and fire (Walker, 1993), which make it difficult to select a baseline condition.

Hence, in practice, society needs to make a choice regarding the benchmark for assessing sustainability. The benchmark may be based on trends occurring at a more aggregated scale than that of the ecosystem being studied. For instance, in the case of De Wieden, which is highly important for biodiversity conservation in the Netherlands, rehabilitation of the lakes could be seen as an important measure for reaching sustainable ecosystem management at the national scale.

Maintenance of the resilience of ecosystems is an important element in ensuring the sustainability of ecosystem management (Levin et al., 1998; Carpenter et al., 2001; Brock et al., 2002). Resilience determines the system's capacity to deal with external disturbance, such as that resulting from weather extremes, climate change or invasions of exotic species. In the De Wieden wetland, for instance, the resilience of the system is related to the amount of total phosphorus that can be absorbed by the system until a threshold value is passed that brings the ecosystem in another state. In semi-arid rangelands such as the Ferlo, maintaining the ecosystem's resilience is required to ensure the productivity of the herbaceous layer during droughts.

As already shown by Common and Perrings (1992) in a formal manner, maintaining resilience is not a sufficient condition for ensuring sustainability. For instance, a certain, small loss of biodiversity in an ecosystem may not necessarily reduce its resilience (in the case where there are other species in the same functional guilds that can take over the role of the lost species, see Mageau et al., 1998), but would compromise the biodiversity conservation service. Or, to give another counter-example, a loss of a species with high pharmaceutical potential would involve a loss of ecosystem service supply, but may not necessarily lead to a reduction in the ecosystem's resilience.

7.2.3 Analysing the Equity Aspects of Ecosystem Management

Equity is relevant with regards to, in particular, the involvement of stakeholders in designing and implementing ecosystem management strategies, and the distribution of benefits from ecosystem management. Whereas larger economic efficiency implies higher overall benefits, it does not necessarily imply that all stakeholders benefit – and a range of studies illustrate that the poorest people in a society may often be most strongly affected by changes in ecosystem management. Poor or indigenous stakeholders may depend most strongly on income from natural resources (e.g. collection of non-timber forest products), and may lack the capacity, network and/or resources to ensure that their interests are sufficiently considered in policy making (e.g. Koop and Tole, 2001). In general, ecosystem services analysis and valuation can assist in revealing (1) the interests of different stakeholder groups in ecosystem management; and (2) the respective impacts of ecosystem change on different stakeholders.

It is generally difficult to define how and when a distribution of benefits can be considered 'equitable'. A benchmark for analysing equitable benefit sharing by stakeholders may be based on a stakeholder consultation process, potentially in combination with an analysis of ecosystem benefits provided in a particular year in the past. However, there is no guarantee that stakeholders will manage to converge to a common understanding of equitable ecosystem management. Alternatively, a citizen's jury may be used to reveal conditions for the equitable management of a specific ecosystem. The citizen's jury comprises citizens that do not have a particular interest in the area, and they are asked to deliberate on a specific topic such as the equitable sharing of benefits provided by an ecosystem, with the perspective of society at large (Renn, 2006). In the case of ecosystems, a challenge is to provide sufficient information to the citizens on the different services at stake. For instance, if biodiversity conservation is to be considered in a citizens' jury, it is essential that the participants are well informed on biodiversity contained in the ecosystem prior to the deliberations. A potential issue with citizen's juries is to what extent stakeholders will accept the findings of a citizen's jury in which they were (by definition) not represented (Creighton et al., 1998).

Scale is a crucial issue in analysing the benefits of ecosystem management. As illustrated in the case of De Wieden, stakeholders at different scales may have an interest in very different ecosystem services. Commonly, local stakeholders will particularly value the provisioning services that provide local benefits. Some regulation and cultural services are most relevant at higher scales, such as carbon sequestration or biodiversity conservation. Hence, with regards to areas of high importance for biodiversity conservation, decision making should not take place at the local level. In areas rich in biodiversity, local people will perceive an abundance of natural ecosystems – possibly in combination with a lack of employment or housing opportunities. Locally, there may therefore be strong and valid arguments for exploiting ecosystems in order to create employment. The same ecosystem may, however, be highly threatened and unique when considered at the national or global scale – and global sustainable development as well as efficient management of ecosystem services may require conservation of the ecosystem in question. Consequently, decentralising ecosystem management may lead to a lack of consideration of biodiversity and other global services, e.g. carbon sequestration, in ecosystem management, and in ecosystem management that is not economically efficient or sustainable at the national level.

In some cases, the local benefits from recreation and tourism may be much larger than the local opportunity costs of nature conservation. This is illustrated by the gradual changes in the attitude of local people towards the Abruzzo National Park in central Italy. The establishment of the park caused a significant increase in visitors to the area, which benefitted local businesses, in particular hotels and restaurants. In addition, farmers were provided compensation payments for losses of sheep killed by wolves. In spite of initial scepticism, these local benefits created a strong basis of support among local people for maintaining and even expanding the park in recent years (Bauman, 2003). It is, of course, not always possible to generate significant additional local income from ecosystem services generated by a transition to sustainable ecosystem management. For instance, a national park may be too remote to attract tourists. In these cases, it can be examined if Payments for Ecosystem Schemes (PES) are an option to financially support sustainable local ecosystem management (e.g. Rosa et al., 2004; Bulte et al., 2008), see also Section 2.4.5.

7.3 IMPLICATIONS OF COMPLEX DYNAMICS FOR ECOSYSTEM MANAGEMENT

7.3.1 Irreversible Responses

Irreversible ecosystem dynamics involves changes in ecosystems that cannot, or only to a very limited extent, be undone through natural processes. For instance, the extinction of a particular species or the loss of an ecosystem can be considered irreversible (Barbault and Sastrapradja, 1995). Irreversibility may also refer to changes in the state of an ecosystem, for example in the case of the transition of a rangeland dominated by palatable grasses to one dominated by unpalatable shrubs (Laycock, 1991). Once a threshold in the ecosystem state is passed and the ecosystem (irreversibly) moves into another state, its capacity to supply ecosystem services is likely to be affected.

A trial-and-error strategy is, in the case of irreversible changes, not a viable pathway towards sustainable, economic efficient resource use. The impacts of a 'collapse' of an ecosystem, which can be described as a rapid change to a degraded ecosystem state that supplies few ecosystem services (see Diamond, 2005 or Costanza et al., 2007), depend on the alternative income-earning opportunities the affected people have. Whereas it may or may not be economically efficient to deplete a local ecosystem to the level of a collapse (see Chapter 4), it is clear that depletion means that sustainability is impaired. Local stakeholders may have few alternative livelihood opportunities and may be particularly affected. The degree of irreversibility of the ecosystem change determines potential future opportunities to use the ecosystem, either through supporting economic activities or, in the case where life-supporting regulation services have been affected, as a place to live. Clearly, the scale at which an ecosystem or ecosystems are affected is a key concern in this respect. The premises of this book are only valid at the level of local ecosystems. In the case of ecosystem degradation at a global scale, different approaches need to be formulated to ensure optimal management of resources. These may involve, for instance, the formulation of policy objectives that involve long-term objectives based on avoiding the passing of (irreversible) thresholds in ecosystem functioning, such as targets for the stabilisation of greenhouse gas concentrations (see Nordhaus, 1999).

The threshold, and even the degree of reversibility, may be difficult to predict before actual changes in the ecosystem have taken place. As Chapter 4 demonstrates, the ecological condition that needs to be maintained to avoid collapse may relate to critical values of *a combination* of related ecosystem state variables. In the case of the hypothetical forest ecosystem presented in Chapter 4, the ecosystem collapses once a certain combination of topsoil depth and forest cover is passed. Hence, establishment of the minimum stock level needs to be based upon a detailed analysis of the drivers for ecosystem change, and may need to define thresholds in relation to multiple ecosystem components. In addition, the threshold level may be influenced by stochastic factors, such as a drought or fire, that may evoke a shift in ecosystem state once its resilience has been reduced.

In the case of uncertainty, the corresponding concept that indicates the minimum ecological condition that should be preserved in order to avoid collapse of the ecosystem, is the 'safe minimum standard' (SMS), as proposed by Ciricay-Wantrup (1968), modified by Bishop (1978). As in the case of the minimum sustainable stock levels, application of the SMS concept to real-world ecosystems with strongly connected components also requires consideration of sets of ecosystem state indicators.

7.3.2 Multiple States and Thresholds

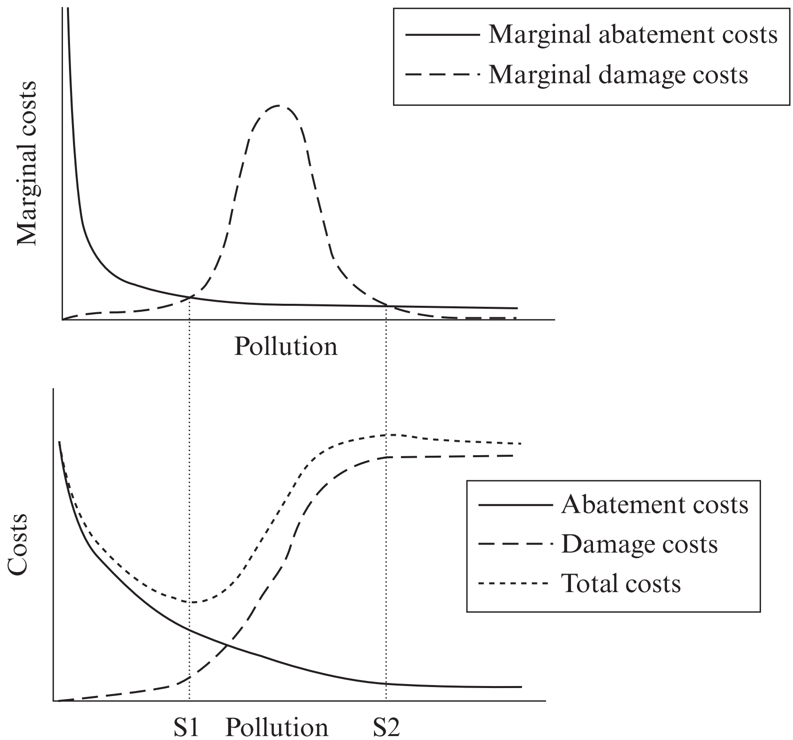
Multiple steady states are relatively stable configurations of an ecosystem, characterised by a certain abiotic and biotic configuration (e.g. Scheffer et al., 1993). If an ecosystem has only one steady state in a certain environmental condition, it will tend to move back towards this state following a disturbance. However, if an ecosystem has more than one state for a certain condition, a disturbance may place the state of the system beyond a threshold leading to a shift to the other state. Shallow lakes are an example of an ecosystem type that is prone to alternative stable states (Scheffer et al., 2001). For a given, intermediate nutrient concentration, shallow lakes can be in a clear water state with low phytoplankton concentrations and a diverse fish community; or in a turbid water state with high phytoplankton concentrations and a species-poor fish community dominated by bream (*Abramis brama*).

The presence of multiple states and thresholds has important consequences for ecosystem management, because the supply of ecosystem services is often strongly linked to the state of the ecosystem. For instance, in the shallow lakes of the De Wieden case study site (Chapter 5), the state of the ecosystem (turbid versus clear water) has a major impact on the conservation of biodiversity and the recreational opportunities provided. The occurrence of ecosystem states needs to be considered in policy making and ecosystem management. As shown in Chapter 5, it is economically *in*efficient to focus policy standards on nutrient levels in shallow lakes solely, without consideration of indicators for the state of the ecosystem. In general, in the management of ecosystems subject to steady states, the benefits of a reduction in the pressure (e.g. pollution) on an ecosystem are small as long as the threshold level at which a shift to a rehabilitated state takes place is not passed.

A general overview of the relation between pollution control and net benefits in the case of an ecosystem subject to two stable states is presented in Figure 7.1. The figure presents the (marginal and total) abatement costs that gradually increase for increasing levels of pollution control (and hence for decreasing levels of pollution). The damage costs rapidly increase when the threshold is passed and the ecosystem switches to a degraded state. Figure 7.1 shows the point of minimum total cost (S1) and maximum total costs (S2). The point of minimum total costs represents the efficient management option. Note that the damage cost curve is non-convex.

7.3.3 Stochasticity and Lag Effects

Stochastic events, such as fire and drought, can be major driving factors for ecosystems (Steele and Henderson, 1984; Friedel, 1991; Bachmann et al., 1999). Lag effects occur where there is a certain amount of time in between the occurrence of the driving factor, and the resulting change in the state of the ecosystem. Often, stochasticity and lag effects act jointly



Notes: The point of minimum total costs (S1) is calculated in Chapter 5 with regards to eutrophication control in a wetland.

Figure 7.1 Pollution damage and abatement costs: marginal and total costs

in determining the dynamics of an ecosystem. In particular, this occurs if a sustained environmental pressure leads to a loss of resilience in the system, which allows a stochastic event to modify the ecosystem's state (Carpenter et al., 2001). The larger the loss of resilience, the smaller the perturbation required to influence the state of the system, and the larger the chance of the occurrence of an event that is big enough to modify the ecosystem. For instance, as demonstrated in Chapter 6, in semi-arid rangelands, the impact of continuous heavy grazing pressures may be lagged in the sense that impacts of overgrazing – in terms of a strong reduction in the supply of animal feed – appear in particular in years of drought. In the Sahel, major widespread droughts have occurred in most of the past eight decades with the exceptions of the 1990s and the first decade of the twenty-first century. It is unclear how climate change will affect droughts in the Sahel in the future, but several of the Global Circulation Models predict a trend towards lower rainfall in the Sahel (e.g. Hein et al., 2009, for an overview). The potential impacts of future droughts are, due to the lag effect, determined by the current grazing strategies.

Stochasticity and lag effects have significant implications for ecosystem management (Reed, 1974; Alvarez, and Shepp, 1998). First, both aspects delay the response of the ecosystem to management. This reduces the net present value of the management measure as the benefits of the measure start occurring at some time in the future (cf. Carpenter et al., 1999). In a comparable manner, they increase the net present value of management options that lead to degradation of the ecosystem, compared to a situation in which these impacts are immediate. For some ecosystems, the approximate delay in ecosystem response to management due to a lag effect can be derived from past experiences. For example, the lag effect occurring in the response of Danish shallow lakes to phosphorus loading and de-loading due to the buffer effect of sediment layers has been quantified through long-term analysis of lake dynamics (Jeppesen et al., 1991 in Scheffer 1998; Søndergaard et al., 1993). However, in the case where the response of an ecosystem is triggered by a stochastic event, the length of the delay may be impossible to predict. For instance, in the Ferlo, the rainfall conditions determine when these impacts become apparent. If the timing of a change in ecosystem conditions cannot be predicted, it is usually more difficult to organise the implementation of mitigation options, for instance, if this involves the stocking of perishable food reserves.

A second impact of stochasticity and lag effects is that they conceal, to some extent, the link between human management (or pressures) and ecosystem responses. In particular, the implications of a long-term decrease in resilience may be underestimated vis-à-vis the effects of a, more obvious, stochastic disturbance. For instance, in the case of the Ferlo, the impact of drought is immediate and dramatic, whereas the impact of high grazing pressures that reduce the resilience of the system to drought is much less obvious. Given the uncertainty that is inherent to ecosystem dynamics and responses of ecosystems to management, this may hamper the enthusiasm of stakeholders for measures that restrict ecosystem use in the short term (such as destocking).

Although the impact of the stochastic event is often obvious, it can be more effective to control the loss of resilience, because stochastic events, such as storms, fires or droughts, are usually difficult to predict or control (Scheffer et al., 2001). For instance, in the case of semi-arid rangelands, droughts and the resulting strong deterioration in food availability for local people attract strong attention from policy makers and the public alike. Obviously, these emergency situations need immediate action to assist people overcoming food shortages. However, in addition to emergency help, there is also a need to assist people in managing their ecosystem in such a way that the resilience of the system to cope with droughts is maintained or improved.

7.4 SELECTED OTHER POTENTIAL APPLICATIONS

7.4.1 Environmental Impact Assessment

An Environmental Impact Assessment (EIA) is defined as 'the process of identifying, predicting, evaluating and mitigating the biophysical, social, and other relevant effects of development proposals prior to major decisions being taken' (e.g. Petts, 1999). A key step in the EIA process is the preparation of the Impact Assessment report, which usually describes the project environment, the proposed activities, potential alternatives, the potential impacts, mitigation measures and an environmental management plan that may include training and/or monitoring activities. The main purpose of the process is to mitigate potential environmental impacts, either through avoiding, reducing or compensating for negative impacts. EIAs are legally required for projects with major environmental impacts in all OECD and most developing countries.

EIAs have been applied for several decades and are a well-established environmental management tool. EIAs do not require analysis of economic consequences of environmental impacts, although in specific cases regulators occasionally request additional economic information from the proponent of the project on top of an EIA in order to evaluate the appropriateness of the proposed mitigation measures. For instance, authorities may require an analysis of the social costs and benefits of potential mitigation measures in order to establish the required mitigation efforts for a certain investment or to remedy a certain amount of pollution.

In an EIA, the concept of ecosystem services can potentially be applied in order to quantify the societal impacts of environmental change resulting from a project's construction or operation activities. For instance, construction of a pipeline through a forest leads to forest loss and dissection and, in case of pipeline failures, to leakage of the medium to be transported. In this case, the first order environmental impacts are, among others, land use conversion and pollution. The second order environmental impacts are an impact on the state of the affected ecosystem, for example, increased concentrations of contaminants. The third order impacts would then be a loss of ecosystem services supply such as a loss

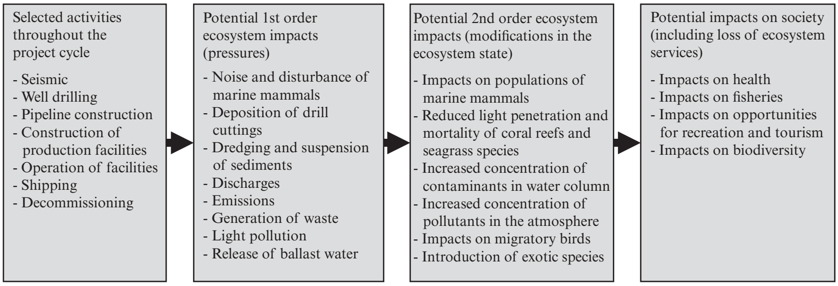


Figure 7.2 Selected potential environmental impacts of a hypothetical off-shore oil and gas exploration and production project

of wood production, recreation opportunities and biodiversity conservation. Analysing changes in the supply of ecosystem services due to project activities makes the project's impacts on society more explicit.

Figure 7.2 illustrates how the ecosystem services approach can be used in the context of EIA, using the case of off-shore oil and gas exploration as an example. Figure 7.2 specifies selected potential impacts of such a project. Note that different impacts occur during different stages of the project cycle, including the exploration and appraisal (of hydrocarbon stocks) phase, and the construction, operation and decommissioning phases. In particular, an ecosystem services approach can facilitate (1) analysis of how project impacts modify the ecosystem and lead to social and economic impacts; (2) identification of mitigation and compensation measures for first, second and third order impacts, and assessing residual impacts; and (3) assessment of ecological, social and economic impacts of different project alternatives.

Analysing third order impacts, i.e. the societal consequences of the environmental impacts of projects, requires analysis of how various drivers such as pollution or land use conversion affect the ecosystem and, subsequently, the supply of ecosystem services, while accounting for the specific dynamics of the ecosystem involved. Clearly, this is more data intensive compared to an approach that requires only assessment of first order impacts. Spending additional resources for a more elaborate approach to EIA involving the analysis of ecosystem services will generally not be justified where effective mitigation measures for first order impacts can be proposed. A potential niche for applying ecosystem services analysis in the context of EIA is where there is insufficient scope for avoiding or mitigating environmental impacts through site selection and/or end-of-pipe measures. For instance, in the case of residual environmental impacts that require compensation of local stakeholders, ecosystem services assessment can be useful in providing insight in the specific impacts and costs born by stakeholders affected by a proposed project and the degree of compensation required.

In addition, ecosystem services analysis can support EIA in the comparison and selection of project alternatives. Project alternatives will often have a different environmental impact as well as requiring different capital expenditures and/or operation and maintenance costs. In this case, economic valuation of changes in ecosystem services supply resulting from different project activities can support comparison of the relative economic benefits of project alternatives (i.e. inclusive of environmental costs and benefits). For instance, alternative pipeline routings generally have different cost figures as well as a different impact on ecosystem services supply. Ecosystem services valuation can facilitate comparison of additional investment costs with the economic value of avoided environmental impacts.

7.4.2 Environmental and Social Cost–Benefit Analysis

Environmental Cost–Benefit Analysis (CBA) is used to evaluate the net economic benefits of an investment option – inclusive of its positive and negative environmental impacts. In an Environmental CBA, environmental impacts need to be expressed in economic terms. In a Social CBA, additionally, in the aggregation of the benefits provided by different ecosystem management options, higher weights may be given to costs or benefits accruing to disadvantaged or low-income groups (see Pearce et al., 2005 for more details). In other words, Environmental and Social CBAs consider the costs and benefits for society at large, rather than the private benefits related to an investment decision.

Environmental and Social CBAs have now been applied in a range of settings. For instance, Environmental CBAs have been applied in relation to transport infrastructure projects in, among others, France, Germany, Japan, the Netherlands and the US (Hayashi and Morisugi, 2000). Impacts that are most commonly included in an Environmental CBA are emissions, noise pollution and spills to soil and groundwater. Emissions can be included on the basis of marginal costs estimates that express societal costs per unit of pollutant emitted. Such cost estimates are available for a range of pollutants, in particular CO_2, CO, NOx, SOx, VOC and Particulate Matter (PM). The marginal costs per unit emitted depend on the environment in which the pollution takes place, but since air pollution disperses and mixes in the troposphere, marginal cost estimates have often been assumed to be valid at the scale of the continent. Most information is available for Europe and the US, where significant differences occur in the marginal cost estimates proposed for specific pollutants (e.g. Forkenbrock, 1999; Sommer et al., 2000; Mayeres et al., 2001).

Noise pollution can be measured in terms of an increase in the level of background or peak noises to which people are exposed. A range of contingent valuation studies have been undertaken in order to reveal the WTP for avoiding noise pollution (see Navrud, 2003 for an overview). This WTP appeared to vary over a considerable range, from 2 to 99 euro per decibel per household per year (euro/dB/hh/year), with a median value around 25 to 30 euro/dB/hh/year. The large range reflects differences in preferences between people, as well as the variety of research approaches and circumstances in which the studies took place. Spills affecting soil and groundwater quality can be analysed on the basis of site remediation costs, which vary considerably as a function of the size of the spill, the type of pollutant, the type of environment, and the country involved (see Khan et al., 2004).

More difficult to include in an Environmental or Social CBA are, for instance, impacts on nature, landscape and cultural aspects, which are often described in biophysical rather than economic terms (e.g. Van Wee et al., 2003). For surface water pollution, generally applicable marginal cost estimates for specific pollutants cannot be provided, since the economic costs of discharges to water strongly depend on the environmental context including the volume and type of waterbody, flows and currents guiding dispersion, temperature and underwater soil properties determining pollutant breakdown rates and the ecosystem services provided by the waterbody.

A particular source of uncertainty in relation to the application of Environmental CBA relates to the valuation of health impacts of projects or policies, including morbidity and mortality impacts. The costs of morbidity can be related to the costs of a reduction in well-being of affected people, medical treatment costs and costs of lost labour days. The expression of mortality, i.e. a premature statistical fatality, in economic terms is a highly debated topic. Two approaches focus respectively on the value of a prevented fatality (VPF) or the costs of life years lost (VOLYs). Whereas a range of value estimates for VPFs and VOLYs are available, there is no consensus yet on the methodologies to be applied for the valuation of health impacts (e.g. Rabl and Spadaro, 1999; Gouveia and Fletcher, 2000; AEAT, 2005).

Hence, considerable uncertainties remain with regards to the expression of several types of environmental impact in monetary terms, and only part of the overall environmental impacts of a project or policy can be meaningfully translated into a cost–benefit estimate. However, a main advantage of Environmental CBA is that (some of the) environmental impacts and their welfare implications can be included in the policy debate at an earlier phase and more prominently, as compared to a situation where only an EIA is conducted, since an EIA is usually done when the main project concepts have already been decided upon. Therefore, Environmental and Social CBAs can be a useful tool to support policy making, provided that the uncertainties and aspects not included in the valuation are made clear (for more detail see Pearce et al., 2005).

Some of the environmental impacts of a project to be included in an Environmental or Social CBA may directly affect human welfare, for instance, pollution affecting people's health. Another type of impact is more indirect; a project may lead to ecosystem degradation, which may subsequently affect people because of a decline in the supply of ecosystem services. The contribution the analytical approach described in this book can make to Environmental and Social CBA is confined to the second case. Ecosystem impacts can be analysed and valued by means of a three step procedure, in line with Figure 3.1: (1) quantifying the impact of the project on the state of the affected ecosystems; (2) quantifying resulting changes in ecosystem service supply; and (3) monetary valuation of the changes in ecosystem services supply. For a discussion of ECBA methodologies related to health impacts, see Ostro and Chestnut (1998), AIChe (2000) and AEAT (2005).

7.4.3 Spatial Planning

Land use maps commonly include land use classes based on goods and services that can easily be linked to a certain land cover unit, such as cropland or forest (e.g., Rounsevell et al., 2005). However, in particular regulating services (such as the hydrological service) and cultural services (such as recreation) may not depend on one single land use unit, or ecosystem, but rather on the complex of ecosystems within a landscape. There is, as yet, relatively limited experience with mapping ecosystem services that cannot be directly linked to land cover, including services that depend on the spatial configuration of the landscape rather than on individual land use classes (but see for examples Geoghegan et al., 1997; Egoh et al., 2008; Willemen et al., 2008; Nelson et al., 2009).

In many rural areas, the regulation and cultural services provide a significant share of the per hectare economic value of land cover units. For example, in many wetlands, water purification, recreation and biodiversity conservation are among the most valuable services provided by the ecosystem (Wilson and Carpenter, 1999). The value of the carbon sequestration and hydrological services may exceed the value of timber harvesting in a range of tropical forest ecosystems (e.g. Creedy and Wurzbacher, 2001; Hein et al., 2008). Hence, land use maps tend not to provide a full picture of the benefits provided by the different units of the maps. An ecosystem services approach to mapping benefits provided by spatial units may provide more comprehensive information relevant to land use planning. A spatial ecosystem services approach requires a GIS linking land cover and land management to ecosystem service supply.

Hence, GIS modelling of ecosystem services can be a valuable tool for supporting land use planning. Provided that a comprehensive modelling of ecosystem services supply as a function of land cover and management can be realised, such a tool can assist in quantifying the costs and benefits of land use change as a function of spatial policies, as well as in optimising the spatial configuration of land uses within a certain area (e.g. Voinov et al., 1999; Willemen et al., 2008).

There are various challenges in the spatial quantification and modelling of ecosystem services, and in making economic values of these services spatially explicit. A first challenge is to delineate services that are not strongly linked to a particular land cover unit. For instance, in the case of the provision of opportunities for recreation, the supply of the service depends on the characteristics of the landscape or ecosystem (visual attractiveness, biodiversity, etc.), as well as on the presence of tourism facilities (access roads, hiking paths, hotels and camp sites, etc.). The performance of the service can be quantified in terms of, for instance, visitor-days, but it is not straightforward to allocate these visitor-days over the landscape or ecosystem involved in order to identify the amount of recreation and tourism opportunities generated per spatial unit.

A second challenge is to account for interactions occurring between ecosystem services at the scale of the landscape. The supply of an ecosystem service may be neutral, conflict with, or provide synergies with regards to the supply of other services. For example, the conservation of biodiversity may support recreation by increasing the attractiveness of an area for visitors, but recreation may cause disturbance and reduce the value of the biodiversity conservation service (e.g. Willemen et al., 2009). Hence, the value of an ecosystem service generated by a land use unit is, in part, determined by land use in neighbouring land use units. Changes in the supply of a service are likely to have repercussions for ecosystem service supply in nearby land cover units.

In general, spatial analysis and modelling of ecosystem services is data intensive and computationally complex, particularly with regards to linking regulation and cultural services to the spatial configuration of the landscape. Further development of spatial ecosystem service modelling techniques will enhance the possibilities of analysing the impacts of spatial policies on ecosystem services supply and, hence, for the analysis of economic efficiency, sustainability and equity implications of spatial policies.

7.5 CONCLUSIONS

The framework for ecosystem assessment presented in this book comprises the modelling of drivers for ecosystem change, quantifying ecosystem states and processes, and analysing societal impacts in terms of changes in ecosystem services supply. It can be used to analyse the economic efficiency, sustainability and equity impacts of ecosystem change and ecosystem management options. The framework itself is generic; applying it involves the development of specific ecosystem models as illustrated by the case studies presented in Chapters 4 to 6. Application of the framework requires ecological–economic modelling, for which a dynamic systems modelling approach is best suited. Dynamic systems modelling has been applied to reveal the ecological and/or economic impacts of ecosystem management options in a range of cases, see Costanza and Ruth (1998) and Eriksson and Hammer (2006). The added value of the framework is that it facilitates a structured, comprehensive approach to analysing ecosystem change, and that it links ecosystem change to the three criteria of efficiency, sustainability and equity.

Developing ecosystem models based on the framework presented involves modelling the causal chain 'drivers–state–ecosystem services supply' as well as analysing the costs of management options and the benefits of changes in ecosystem services supply. Drivers can be both exogenous (e.g. management options) or endogenous (ecological processes). Applying a dynamic systems approach involves specifying the ecosystem in terms of stock and flow variables, and modelling their interactions. The link between the ecosystem state and the supply of ecosystem services needs to be assessed for each ecosystem service separately. For provisioning services, this requires the linking of flows (e.g. timber harvest) to the specific stock involved (e.g. standing timber stock). Most regulating and cultural services depend upon a combination of components (e.g. the hydrological service depends on topography, vegetation cover, soil characteristics, etc.). Incorporating ecological realism into the models requires consideration of potential feedbacks between ecosystem components and ecosystem service use. Thresholds for rapid and/or irreversible ecosystem change can both be imposed on the ecosystem model, if the threshold levels are known, or they can be endogenous in the model, resulting from the modelling of interactions between drivers and ecosystem components.

Data constraints and uncertainty are inherent characteristics of the approach. Ecosystems are complex, involving a large number of ecological processes operating across a range of scales (e.g. Limburg et al., 2002; Holling and Gunderson, 2002), and incorporation of only one or several drivers and processes means that a simplification is made. Calibration of the model can be done on the basis of time series of ecosystem change as a function of pressures and drivers, and sensitivity analysis should be an integral part of the modelling in order to verify model structure and outcomes. Data requirements limit the application of the approach to those ecosystems for which sufficient data are or can be made available. Normally, applying the approach requires time series of several years of data on pressures, state variables and ecosystem service supply.

Provided that sufficient data and resources for model development are available, the approach has a high potential to support ecosystem management. In particular, applying the approach can point out in a quantitative rather than an intuitive manner the economic implications of ecosystem change for different stakeholders, as well as the sustainability of ecosystem management options. The approach is also useful in supporting stakeholder participation processes for designing environmental management strategies, by providing a quantitative basis for stakeholder deliberations.

As illustrated in the three case studies presented in this book, complex dynamics have major implications for the economic efficiency, sustainability and equity of ecosystem management options, and need to be understood and considered prior to decision making. For instance, investments in ecosystem rehabilitation may not lead to any economic benefit if rehabilitation does not take place beyond the level of thresholds guiding shifts in ecosystem state. And, in the case of irreversible ecosystem responses, trial-and-error approaches to ecosystem management may be highly ineffective from both an efficiency, sustainability and an equity perspective.

Given the current rapid degradation of ecosystems worldwide, enhanced, informed decision making on remaining ecological resources is crucial. One of the preconditions for better managing ecosystems is a thorough understanding and alignment among stakeholders on the ecological, economic and social impacts of ecosystem change. The quantitative approach described in this book can be used to elucidate the societal implications of ecosystem management options. However, the analysis and identification of approaches that provide the best balance between economic efficiency, sustainability and equity is not sufficient for a transition to enhanced ecosystem management. Because many of the services provided by ecosystems are public goods, regulation of ecosystem use by key stakeholders and/or a government is required in order to ensure efficient and sustainable ecosystem management. The creation of markets for ecosystem services is an additional, important mechanism for enhancing ecosystem management, applicable to those services for which efficient markets including benefit sharing arrangements can be developed.

References

Adriansen, H.K. (2006), Continuity and change in pastoral livelihoods of Senegalese Fulani. *Agricultural Human Values*, **23**, 215–29.

Adriansen, H.K. and T.T. Nielsen (2002), Going where the grass is greener: on the study of pastoral mobility in Ferlo, Senegal. *Human Ecology*, **30**(2), 215–26.

AEAT (2005), Service contract for carrying out cost–benefit analysis of air quality related issues, in particular in the Clean Air For Europe (CAFE) programme. Methodology for the cost–benefit analysis for CAFE: Volume 1 to 3. AEA Technology Environment, Oxon, UK.

Agder, W.N., M. Kelly, N.H. Ninh and N.C. Thanh (1997), Property rights and the social incidence of mangrove conversion in Vietnam. CSERGE Working Paper GEC 97-21. Centre for Social and Economic Research on the Global Environment, University of East Anglia and University College London, UK.

AIChE (2000), *Total Cost Assessment Methodology*, American Institute of Chemical Engineers, Center for Waste Reduction Technologies.

Alder, J. and Sumaila, U.R. (2004), Western Africa: a fish basket of Europe past and present. *Journal of Environment and Development*, **13**, 156–78.

Aldred, J. (1994), Existence value, welfare and altruism. *Environmental Values*, **3**, 381–402.

Alvarez, L.H.R and L. Shepp (1998), Optimal harvesting of stochastically fluctuating populations. *Journal of Mathematical Biology*, **37**, 155–77.

Anderson, J.R. and J.L. Dillon (1992), Risk analysis in dryland farming systems. Farm Systems Management Series 2, FAO, Rome, 1992.

Andre, D. (1998), 17 ans de suivi de la vegetation ligneuse et herbacée dans le périmètre expérimental de Widou-Thiengoly, Fero, Nord Sénégal. Projet d'autopromotion pastorale dans le Ferlo (PAPF), Saint-Louis, Sénégal.

Ansink, E., L. Hein and K.P. Hasund (2008), To value functions or services? An analysis of ecosystem valuation approaches. *Environmental Values*, **17**(4), 489–503.

Arrow, K.J. (1963), *Social Choice and Individual Values* (2nd edition), New Haven, CT: Yale University Press.

Arrow, K., R. Solow, P.R. Portneye et al. (1993), Report of the NOAA Panel on Contingent Valuation, Federal Reg. 58, Washington, DC.

Arrow, K., B. Bolin, R. Costanza et al. (1995), Economic growth, carrying capacity and the environment. *Science*, **268**, 520–21.

Arrow, K.J., W. Cline, K.G. Mäler et al. (1996), Intertemporal equity, discounting and economic efficiency, in J.P. Bruce, H. Lee and E.F. Haites (eds), *Climate Change 1995: Economic and Social Dimensions of Climate Change, Second Assessment of the Intergovernmental Panel on Climate Change*, Cambridge, UK and New York, NY, USA: Cambridge University Press, pp. 135–44.

Asheim, G.B. (1986), Hartwick's rule in open economies. *The Canadian Journal of Economics*, **19**, 395–402.

Atkinson, G. and D.W. Pearce (1993), Measuring sustainable development. The Globe 13, UK GER Office Swindon.

Augustine, D.J., L.E. Frelich and P.A. Jordan (1998), Evidence for two alternate stable states in an ungulate grazing system. *Ecological Applications*, **8**, 1260–69.

Bachmann, R.W., M.V. Hoyer and D.E. Canfield (1999), The restoration of Lake Apopka in relation to alternative stable states. *Hydrobiologia*, **394**, 219–32.

Bal, D., H.M. Beije, M. Felinger et al. (2001), Handbook Nature Target Types (In Dutch: *Handboek Natuurdoeltypes*). Report Expertise Centre LNV 2001/020.

Balmford, A., A. Bruner, P. Cooper et al. (2002), Economic reasons for conserving wild nature. *Science*, **297**, 950–53.

Barbault R. and S. Sastrapradja (1995), Generation, maintenance and loss of biodiversity, in V.H. Heywood and R.T. Watson (eds), *Global Biodiversity Assessment*, Cambridge, UK: United Nations Environment Programme, pp. 193–276.

Barbier, E.B. (2000), Valuing the environment as input: review of applications to mangrove-fishery linkages. *Ecological Economics*, **35**, 47–61.

Barbier, E.B. and A. Markandya (1990), The conditions for achieving environmentally sustainable development. *European Economic Review*, **34**, 659–69.

Barbier, E.B., J. Burgess and C. Folke (1994), *Paradise Lost? The Ecological Economics of Biodiversity*, London: Earthscan.

Barkai, A. and C. McQuaid (1988), Predator–prey role reversal in a marine benthic ecosystem. *Science*, **242**, 62–4.

Barro, R.J. and X. Sala-i-Martin (1995), *Economic Growth*, New York: McGraw-Hill Inc.

Batabyal, A.A., B. Biswas and B.E. Godfrey (2001), On the choice between the stocking rate and time in range management, *Environmental and Resource Economics*, **20**, 211–23.

Bauman, W. (2003), Communicating sustainable tourism in National Parks. Forest and Nature Conservation Policy Group, Thesis, Wageningen University.

Baumol, W.J. (1986), *Superfairness: Applications and Theory*, Cambridge, MA: MIT Press.

Bayer, W. and A. Waters-Bayer (1998), Forage Husbandry. Technical Centre for Agricultural and Rural Cooperation, CTA, Wageningen, the Netherlands.

Becker, C.D. and E. Ostrom (1995), Human ecology and resource sustainability: the importance of institutional diversity. *Annual Review of Ecology and Systematics*, **26**, 113–33.

Beckerman, W. (1994), 'Sustainable development': is it a useful concept? *Environmental Values*, **3**, 191–209.

Berck, P. (1995), Empricial consequences of the Hotelling principle, in D.W. Bromley (ed.), *The Handbook of Environmental Economics*, Oxford, UK: Blackwell.

Berkes, F., and C. Folke (1998), *Linking Social and Ecological Systems. Management Practices and Social Mechanisms for Building Resilience*, Cambridge, UK: Cambridge University Press.

Bishop, R.C. (1978), Endangered species, irreversibility, and uncertainty: the economics of a safe minimum standard. *American Journal of Agricultural Economics*, **60**, 10–18.

Bishop, J., S. Kapila, F. Hicks, P. Mitchell and F. Vorhies (2008), *Building Biodiversity Business*. London, UK, and Gland, Switzerland: Shell International Limited and the International Union for Conservation of Nature, 164 pp.

Blalock, H.M. (1987), *Social Statistics* (2nd edition), London: McGraw-Hill.

Bockstael, N., R. Costanza, I. Strand et al. (1995), Ecological economic modelling and valuation of ecosystems. *Ecological Economics*, **14**, 143–59.

Boers, P.C.M., W. van Raaphorst and D.T. van der Molen (1998), Phosphorus retention in sediments. *Water Science and Technology*, **37**, 31–9.

Borum, J. and K. Sand-Jensen (1996), Is total primary production in shallow coastal marine waters stimulated by nutrient loading? *Oikos*, **76**, 406–10.

Bosch, J.M. and J.D. Hewitt (1982), A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. *Journal of Hydrology*, **55**, 3–23.

Boudet, G. (1975), *Manuel sur les Pâturages Tropicaux et les Cultures Fourragères*, Paris, France: Editions du Ministère de la Coopération.

Boulding, K.E. (1966), The economics of the coming spaceship Earth, in H. Jarrett (ed.), *Environmental Quality in a Growing Economy*, Baltimore, Maryland: Resources for the Future/John Hopkins Press, pp. 3–15.

Bouma, F. and S.W.F. van der Ploeg (1975), Functions of nature, an economic–ecological assessment (in Dutch: *Functies van de natuur, een economisch-oecologische analyse*). IVM–VU Publication No. 46.

Boyd, J. (2007), Nonmarket benefits of nature: what should be counted in green GDP? *Ecological Economics*, **61**, 716–23.

Brazee, R.J. (2001), The Faustmann formula: fundamental to forest economics 150 years after publication. *Forest Science*, **47**, 441–2.

Breman, H. and N. De Ridder (1991), Manuel sur les pâturages des pays sahéliens, Wageningen, the Netherlands: Technical Centre for Agricultural and Rural Cooperation, CTA, 485 pp.

Brink, C.J. (2003), Modelling cost-effectiveness of interrelated emission reduction strategies: the case of agriculture in Europe. PhD Thesis Wageningen University.

Briske, D.D., S.D. Fuhlendorf and F.E. Smeins (2003), Vegetation dynamics on rangelands: a critique of the current paradigms. *Journal of Applied Ecology*, **40**, 601–14.

Brock, W.A. and D. Starrett (2003), Managing systems with non-convex positive feedback. *Environmental and Resource Economics*, **26**(4), 575–602.

Brock, W.A., K.G. Maler and C. Perrings (2002), Resilience and sustainability: the economic analysis of non-linear dynamic systems, in L.H. Gunderson and C.S. Holling (eds), *Panarchy: Understanding Transformations in Human and Natural Systems*, Washington, DC: Island Press, pp. 261–92.

Brouwer, R., I.H. Langford, I.J. Bateman and R.K. Turner (1997), A meta-analysis of wetland contingent valuation studies. CSERGE Working Paper, University of East Anglia, Norwich, UK.

Brown, G. (1996), Biodiversity valuation and decision making, paper presented at the OECD conference on incentive measures for biodiversity conservation and sustainable use. Cairns, Australia, March 1996.

Bulte, E.H. and G.C. Van Kooten (1999), Metapopulation dynamics and stochastic bioeconomic modelling, *Ecological Economics*, **30**, 293–9.

Bulte, E., L. Lipper, R. Stringer and D. Zilberman (2008), Payments for ecosystem services and poverty reduction: concepts, issues, and empirical perspectives. *Environment and Development Economics*, **13**, 245–54.

Cammeraat, L.H., A.C. Imeson and L. Hein (2002), Investigation on environmental characteristics to underpin the selection of desertification indicators in the Guadelentin Basin, in N.A. Geeson, C.J. Brandt and J.B. Thornes (eds), *Mediterranean Desertification: A Mosaic of Processes and Responses*, Chichester, UK: John Wiley and Sons Ltd, pp. 187–202.

Campbell, B.M. (1996), *The Miombo Woodlands in Transition: Woodlands and Welfare in Africa*, Bogor, Indonesia: Center for International Forestry Research (CIFOR).

Carpenter, S.R. and K.L. Cottingham (1997), Resilience and restoration of lakes. *Conservation Ecology* [online] 1: 2. http://www.consecol.org/vol1/iss1/art21.

Carpenter, R.S., D. Ludwig and W.A. Brock (1999), Management of eutrophication for lakes subject to potentially irreversible change. *Ecological Applications*, **9**, 751–71.

Carpenter, S., B. Walker, J. M. Anderies and N. Abel (2001), From metaphor to measurement: resilience of what to what? *Ecosystems*, **4**, 765–81.

Carson, R.T. (1998), Valuation of tropical rainforests: philosophical and practical issues in the use of contingent valuation. *Ecological Economics*, **24**, 15–29.

Carter, N. (2001), *The Politics of the Environment – Ideas, Activism, Policy*. Cambridge, UK: Cambridge University Press.

Chape, S., J. Harrison, M. Spalding and I. Lysenko (2005), Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. *Philosophical Transactions of the Royal Society B*, **360**, 443–55.

Chiang, A.C. (1992), *Elements of Dynamic Optimization*, Singapore: McGraw-Hill Inc.

Chomitz, K.M., E. Brenes and L. Constantino (1998), *Financing Environmental Services: The Costa Rican Experience and its Implications*, Washington, DC: World Bank ESSD.

Cicchetti, C.V. and A.M. Freeman (1971), Option demand and consumer surplus, further comment. *Quarterly Journal of Economics*, **85**, 528–39.

Ciriacy-Wantrup, S. (1968), *Resource Conservation: Economics and Politics*, 2nd edition, Berkeley, USA: University of California Press.

Clark, C.W. (1976), *Mathematical Bioeconomics: The Optimal Management of Renewable Resources*, New York: Wiley-Interscience pp. 18–21 and 42–44.

Clark, C.W., D.D. Jones and C.S. Holling (1979), Lessons for ecological policy design. a case study of ecosystem management. *Ecological Modelling*, **7**, 22–53.

Clements, F.E. (1916) Plant succession: an analysis of the development of vegetation. Publication 242, Washington, DC: Carnegie Institute.

Cleveland, C.J. and M. Ruth (1997), When, where, and by how much do biophysical limits constrain the economic process? A survey of Nicholas Georgescu-Roegen's contribution to ecological economics. *Ecological Economics*, **22**, 203–23.
Clevering, O.A. (1998), An investigation into the effects of nitrogen on growth and morphology of stable and die-back populations of *Phragmites australis*. *Aquatic Botany*, **60**, 11–25.
Cline, W.R. (1992), *The Economics of Global Warming*, Washington, DC: Institute for International Economics.
Cole, L.C. (1954), Some features of random cycles. *Journal of Wildlife Management*, **18**, 107–9.
Common, M. and C. Perrings (1992), Towards an ecological economics of sustainability. *Ecological Economics*, **6**, 7–34.
Costanza, R. and H.E. Daly (1987), Toward an ecological economics. *Ecological Modelling*, **38**, 1–7.
Costanza, R. and H.E. Daly (1992), Natural capital and sustainable development. *Conservation Biology*, **6**, 37–46.
Costanza, R. and M. Ruth (1998), Using dynamic modelling to scope environmental problems and build consensus. *Environmental Management*, **22**(2), 183–95.
Costanza, R., L. Wainger, C. Folke and K.G. Mäler (1993), Modelling complex ecological economic systems: toward an evolutionary, dynamic understanding of people and nature. *BioScience*, **43**, 545–5.
Costanza, R., R. d'Arge, R.S. de Groot et al. (1997), The value of the world's ecosystem services and natural capital. *Nature*, **387**, 253–60.
Costanza, R., B.S. Low, E. Ostrom and J. Wilson (eds) (2001), *Institutions, Ecosystems, and Sustainability*, Boca Raton, Florida: Lewis Publishers.
Costanza, R., L.J. Graumlich and W. Steffen (eds) (2007), Sustainability or collapse? An integrated history and future of people on Earth. Dahlem Workshop Reports, MIT Press.
Creedy, J. and A.D. Wurzbacher (2001), The economic value of a forested catchment with timber, water and carbon sequestration benefits. *Ecological Economics*, **38**, 71–83.
Creighton, J., J. Delli Priscoli and C. Dunning (eds) (1998), Public involvement techniques: a reader of ten years of experience at the Institute of Water Resources. IWR. Report nr. 82-R-I. Alexandria. USA. (http://www.iwr.usace.army.mil/iwr/pdf/pifirst.pdf).
Croke, J., P. Hairsinea and P. Fogartya (2001), Soil recovery from track construction and harvesting changes in surface infiltration, erosion and delivery rates with time. *Forest Ecology and Management*, **143**, 3–12.
Cropper, M. and D. Liabson (1999), The implications of hyperbolic discounting for project evaluation, in P.R. Portney and J.P. Weyant (eds),

Discounting and Intergenerational Equity, Washington, DC: Resources for the Future, pp. 163–72.
Cummings, R.G. and G.W. Harrison (1995), The measurement and decomposition of nonuse values: a critical review. *Environmental and Resource Economics*, **5**, 225–47.
Curran, L.M., S. Trigg and A. McDonald et al. (2004), Lowland forest loss in protected areas of Indonesian Borneo. *Science*, **303**, 1000–1003.
Daly, H.E. (1990), Towards some operational principles of sustainable development. *Ecological Economics*, **2**, 1–6.
Daily, G.C., S. Polasky and J. Goldstein et al. (2009), Ecosystem services in decision making: time to deliver. *Frontiers in Ecology and the Environment*, **7**, 21–8.
Dasgupta, P.S. and G.M. Heal (1979), *Economic Theory and Exhaustible Resources*, Cambridge, UK: Cambridge University Press.
Dasgupta, P. (1993), Natural resources in an age of substitutability, in A.V. Kneese and J.L. Sweeney (eds), *Handbook of Natural Resource and Energy Economics*, Vol 3. Amserdam: Elsevier.
Dasgupta, P. (2006), Commentary: the Stern review's economics of climate change. *National Institute Economic Review*, **199**, 4–7.
De Bruin, R., H. Oostindie and J. Meindertsma (2001), *Strategical Economic Analysis of De Wieden* (in Dutch: *Strategische bedrijfsanalyse in De Wieden*). Wageningen, the Netherlands: Streekwijzer.
De Groot, R.S. (1992), *Functions of Nature, Evaluation of Nature in Environmental Planning, Management and Decision Making*, Groningen, the Netherlands: Wolters-Noordhoff.
De Groot, R.S., M.A. Wilson and R.M.J. Bouman (2002), A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*, **12**, 393–408.
De Mazancourt, C., M. Loreau and L. Abbadie (1999), Grazing optimisation and nutrient cycling: potential impact of large herbivores in a savanna ecosystem. *Ecological Applications*, **9**, 784–97.
De Leeuw, P.N. and J.C. Tothill (1990), The concept of rangeland carrying capacity in Sub-Saharan Africa: myth or reality. Pastoral Development Network Paper 29. London: Overseas Development Institute.
Deacon, R.T., D.S. Brookshire, A.C. Fisher et al. (1998), Research trends and opportunities in environmental and natural resource economics. *Environmental and Resource Economics*, **11**, 383–97.
Defra (2008), *Sustainable Indicators in your Pocket*, London: Department for Food, Environment and Rural Affairs (Defra) Publications.
Diamond, J. (2005), *Collapse: How Societies Choose to Fail or Succeed*, New York, USA: Viking Press.
Diamond, P.A. and J.A. Hausman (1994), Contingent valuation: is some

number better than no number? *Journal of Economic Perspectives*, **8**, 45–66.
Dillon, J.L. and H.T. Burley (1961), A note on the economics of grazing and its experimental investigation. *Australian Journal of Agricultural Economics*, **5**, 123–32.
Dillon, P.J. and F.H. Rigler (1974), The phosphorus–chlorophyll relationship in lakes. *Limnology Oceanography*, **19**, 767–73.
Direction de la prévision et de la statistique, Ministère de l'Economie, des Finances et du Plan, Sénégal (1997), Enquête Sénégalaise auprès des Ménages. Mars 1994–Avril 1995.
Dixon, J.A. and M.M. Hufschmidt (eds) (1986), *Economic Valuation Techniques for the Environment: A Case Study Workbook*, Baltimore, USA: Johns Hopkins University Press.
Drake, J. (1990), Communities as assembled structures: do rules govern pattern? *Trends in Ecological Evolution*, **5**, 159–64.
Drepper, F.R. and B.Å. Månsson (1993), Intertemporal valuation in an unpredictable environment. *Ecological Economics*, **7**, 43–67.
Dublin, H.T., A.R. Sinclair and J. Glade (1990), Elephants and fires as causes of multiple steady states in the Serengeti-Mara woodlands. *Journal of Animal Ecology*, **59**, 1147–64.
Duelli, P. and M.K. Obrist (1998), In search of the best correlates for local organismal biodiversity in cultivated areas. *Biodiversity Conservation*, **7**, 297–309.
Egoh, B., B. Reyers, M. Rouget et al. (2008), Mapping ecosystem services for planning and management. *Agriculture Ecosystems and Environment*, **127**, 135–40.
Ehrlich, P. and A. Ehrlich (1981), *Extinction: The Causes and Consequences of the Disappearance of Species*, New York: Random House.
Ellis, J.E. and D.M. Swift (1988), Stability of African pastoral ecosystems: alternate paradigms and implications for development. *Journal of Range Management*, **41**, 450–59.
Eriksson, S. and M. Hammer (2006), The challenge of combining timber production and biodiversity conservation for longterm ecosystem functioning: a case study of Swedish boreal forestry. *Forest Ecology and Management*, **237**, 208–17.
European Communities (2009), Conference Proceedings Beyond GDP, Measuring progress, true wealth, and the well-being of nations, 19–20 November 2007, Brussels, available at http://www.beyond-gdp.eu/proceedings.html.
European Environment Agency (2009), Indicators about Europe's environment, available at http://themes.eea.europa.eu/indicators (accessed 15 July 2009).

FAO (1992), Mixed and pure forest plantations in the tropics and subtropics. FAO Forestry Paper 103, 1992. FAO, Rome.

FAO (2001), Farming systems and Poverty: improving farmers' livelihoods systems in a changing world. FAO, Rome.

Faucheux, S., D. Pearce and J. Proops (eds) (1996), *Models of Sustainable Development*, Aldershot, UK and Brookfield, US: Edward Elgar.

Faustmann, M. (1849), On the determination of the value which forestland and immature stands pose for forestry. *Journal for Forest Economics*, **1**, 7–44.

Fernandez-Gimenez, M. and B. Allen-Diaz (1999), Testing a non-equilibrium model of rangeland vegetation dynamics in Mongolia. *Journal of Applied Ecology*, **36**, 871–85.

Folmer, H., N. Hanley and F. Missfeldt (1998), Game-theoretic modelling of environmental and ressource problems: an introduction, in N. Hanley and H. Folmer (eds), *Game Theory and the Environment*, Cheltenham, UK: Edward Elgar.

Forkenbrock, D.J. (1999), External costs of intercity truck freight transportation. *Transportation Research A*, **33** (7/8), 505–26.

Freeman, A.M. (1993), *The Measurement of Environmental Values and Resources: Theory and Methods*, Washington, DC: Resources for the Future.

Friedel, M.H. (1991), Range condition assessment and the concept of thresholds: a viewpoint. *Journal of Range Management*, **44**, 422–6.

Fujita, M., P. Krugman and A.J. Venables (1999), *The Spatial Economy*, Cambridge, MA: The MIT Press.

Fynn, R.W.S. and T.G. O'Connor (2000), Effect of stocking rate and rainfall on rangeland dynamics and cattle performance in a semi-arid savanna, South Africa. *Journal of Applied Ecology*, **37**, 491–507.

Geoghegan, J., L. Wainger and N. Bockstael (1997), Spatial landscape indices in a hedonic framework: an ecological economics analysis using GIS. *Ecological Economics*, **23**, 251–64.

Georgescu-Roegen, N. (1979), Energy analysis and economic valuation. *Southern Economic Journal*, **45**, 1023–58.

Gibbs H.K., S. Brown, J.O. Niles and J.A. Foley (2007), Monitoring and estimating tropical forest carbon stocks: making REDD a reality. *Environmental Resource Letters*, **2**, 045023.

Gordon, H.S. (1954), The economic theory of a common property resource. *Journal of Political Economy*, **62**, 124–42.

Gouveia N. and T. Fletcher (2000), Time series analysis of air pollution and mortality: effects by cause, age and socioeconomic status. *Journal of Epidemiology and Community Health*, **54**, 750–55.

Grasso, M. (1998), Ecological–economic modelling for optimal mangrove trade off between forestry and fishery production: comparing a dynamic and a simulation approach. *Ecological Modelling*, **112**, 131–50.

Green, C., S. Tunstall, J. Garner and A.M. Ketteridge (1994), Benefit transfer: rivers and coasts. Paper prepared for the CEGB meeting on benefit transfer, Flood Hazard Research Centre, Middlesex University, UK, H.M. Treasury, Publication No. 231.

Greenaway, D., S.J. Leybourne, G.V. Reed and J. Whalley (1993), *Applied General Equilibrium Modelling: Applications, Limitations and Future Development*, London: HMSO.

Gregory, R., S. Lichtenstein and P. Slovic (1993), Valuing environmental resources: a constructive approach. *Journal of Risk and Uncertainty*, **7**, 177–97.

Groeneveld, R.A. (2004), Biodiversity conservation in agricultural landscapes: a spatially explicit economic analysis. PhD Thesis Wageningen University No. 3580, Wageningen University, the Netherlands.

Guerin, H., D. Friot, Nd. Mbaye and D. Richard (1993), Alimentation des ruminants domestiques sur pâturages naturels Saháliens et Sahelo-Soudaniens. Rapport No. 39, Institut Sénégalais de recherches agricoles, Dakar, Sénégal.

Guisan, A. and W. Thuiller (2005), Predicting species distribution: offering more than simple habitat models. *Ecology Letters*, **8**, 993–1009.

Gunderson, L.H., C.S. Holling and S.S. Light (1995), *Barriers and Bridges to the Renewal of Ecosystems and Institutions*, New York: Columbia University Press.

Hailu, A., W.L. Adamowicz and P.C. Boxall (2000), Complements, substitutes, budget constraints and valuation: application of a multi-program environmental valuation method. *Environmental and Resource Economics*, **16**, 51–68.

Hajkowicz, S. and K. Collins (2007), A review of multiple criteria analysis for water resource planning and management. *Water Resources Management*, **21**, 1553–66.

Hamilton, K. (2000), Genuine saving as a sustainability indicator. Environment Department Paper No. 77, World Bank, Washington, DC.

Hammond, A., A. Adriaanse, E. Rodenburg, D. Bryant and R. Woodward (1991), *Environmental Indicators: A Systematic Approach to Measuring and Reporting on Environmental Policy Performance in the Context of Sustainable Development*, Washington, DC: World Resources Institute.

Hanemann, W.M. (1995), Contingent valuation and economics, in K. Willis and J. Corkindale (eds), *Environmental Valuation: New Perspectives*, Wallingford, UK: CAB International, pp. 79–117.

Hanley, N. (1999), Cost–benefit analysis of environmental policy and management, in J.C.J.M. van de Bergh (ed.), *Handbook of Environmental Resource Economics*, Cheltenham, UK and Northampton, MA, USA: Edward Elgar, pp. 824–36.

Hanley, N. and C.L. Spash (1993), Cost–Benefit Analysis and the Environment, Aldershot, UK and Brookfield, US: Edward Elgar.

Hannah, L., J.L. Carr and A. Lankeran (1995), Human disturbance and natural habitat: level analysis of a global data set. *Biodiversity and Conservation*, **4**, 128–55.

Hardaker, J.B. (2000), Some issues in dealing with risk in agriculture. Working paper, Agricultural and Resource Economics Group, University of New England, Armidale, Australia.

Hargrove, E.C. (1989), *Foundations of Environmental Ethics*, Englewood Cliffs, NJ, USA: Prentice-Hall.

Harris, G. (1999), This is not the end of limnology (or of science): the world may be a lot simpler than we think. *Freshwater Biology*, **42**, 689–706.

Hartman, R. (1976), The harvesting decision when a standing forest has value. *Economic Inquiry*, **16**, 52–8.

Hartwick, J.M. (1977), Intergenerational equity and the investing of rents from exhaustable resources. *American Economic Review*, **67**, 972–4.

Hayashi, Y. and H. Morisugi, (2000), International comparison of background concept and methodology of transport project appraisal, *Transport Policy*, **7**(1), 73–88.

Hein, L. (2006a), Assessing the costs of eutrophication control in a shallow lake ecosystem subject to two steady states. *Ecological Economics*, **59**, 429–39.

Hein, L. (2006b), The impact of grazing and rainfall variability on the productivity of semi-arid rangelands. *Journal of Arid Environment*, **64**, 488–504.

Hein, L. (2009), Valuation of the pollination service: a review across scales. *The Open Ecology Journal*, **2**(9), 74–82.

Hein, L., K. van Koppen, R.S. de Groot and E.C. Van Ierland (2006), Spatial scales, stakeholders and the valuation of ecosystem services. *Ecological Economics*, **57**, 209–28.

Hein, L. and E.C. Van Ierland (2006), Efficient and sustainable management of complex forest ecosystems. *Ecological Modelling*, **190**, 351–66.

Hein, L. and N. De Ridder (2006), Desertification in the Sahel; a reinterpretation. *Global Change Biology*, **12**, 1–8.

Hein, L. and H.P. Weikard (2008), Optimal long-term stocking rates for livestock grazing in a Sahelian rangeland. *African Journal for Agricultural and Resource Economics*, **2**(2), 126–50.

Hein, L., R. de Groot and K. Soma (2008), Analyzing the economic impacts of land use change: a framework and a case study for the

Miombo woodlands, Zambia. *Journal of Land Use Science*, **3**(4), 231–49.
Hein, L., M.J. Metzger and R. Leemans (2009), The local impacts of climate change in the Ferlo, Western Sahel. *Climatic Change*, **93**, 465–83.
Heino, J., T. Muotka, H. Mykra et al. (2003), Defining macroinvertebrate assemblage types of headwater streams: implications for bioassessment and conservation. *Ecological Applications*, **13**, 842–52.
Held, I.M., T.D. Delworth, J. Lu, K.L. Findell and T.R. Knutson (2005), Simulation of Sahel drought in the 20th and 21st centuries. *Proceedings of the National Academy of Sciences of the United States of America*, **102**, 17891–6.
Hicks, J.R. (1939), The foundations of welfare economics. *Economic Journal*, **49**(196), 696–712.
Hicks, J.R. (1941), The rehabilitation of consumer's surplus. *Review of Economic Studies*, **8**, 108–16.
Hildreth, R.J. and M.E. Riewe (1963), Grazing production curves II: determining the economic optimum stocking rate. *Agronomy Journal*, **55**, 367–9.
Holling, C.S. (1986), The resilience of terrestrial ecosystems: local surprise and global change, in W.C. Clark and R.E. Munn (eds), *Sustainable Development of the Biosphere*, Cambridge, UK: Cambridge University Press.
Holling, C.S. (1992), Cross-scale morphology, geometry and dynamics of ecosystems. *Ecological Monographs*, **62**, 447–502.
Holling, C.S., and L.H. Gunderson (2002), Resilience and adaptive cycles, in L.H. Gunderson and C.S. Holling (eds), *Panarchy, Understanding Transformations in Human and Natural Systems*, Washington, DC: Island Press, pp. 25–62.
Holling, C.S., L.H. Gunderson and G.D. Peterson (2002), Sustainability and panarchies, in L.H. Gunderson and C.S. Holling (eds), *Panarchy, Understanding Transformations in Human and Natural Systems*, Washington, DC: Island Press, pp. 63–103.
Holtzman, J.S. and N.P. Kulibaba (1994), Livestock marketing in pastoral Africa: policies to increase competetiveness, efficiency and flexibility, in I. Scoones (ed.), *Living with Uncertainty: New Directions in Pastoral Development in Africa*, London: Intermediate Technology Publications.
Hosper, H. (1997), Clearing lakes; an ecosystem approach to the restoration and management of shallow lakes in the Netherlands. PhD thesis Wageningen University, Wageningen, the Netherlands.
Hosper, S.H., M.L. Meijer, P.A. Walker et al. (1992), *Handbook*

Biomanipulation: Evaluation of the Possibilities of Biomanipulation for the Rehabilitation of Lakes (In Dutch: *Handleiding actief biologisch beheer : beoordeling van de mogelijkheden van visstandbeheer bij het herstel van meren en plassen*). Lelystad, the Netherlands: RIZA.

Hotelling, H. (1931), The economics of exhaustable resources. *Journal of Political Economy*, **39**, 137–75.

Howarth, R.B. and R.B. Norgaard (1990), Intergenerational resource rights, efficiency, and social optimality. *Land Economics*, **66**, 1–11.

Howarth, R.B and R.B. Norgaard (1993), Intergenerational transfers and the social discount rate. *Environmental and Natural Resource Economics*, **3**, 337–58.

Hueting, R. (1980), *New Scarcity and Economic Growth*. Amsterdam/New York: Agon-Elsevier.

Hueting, R. (1991), The use of the discount rate in a cost–benefit analysis for different uses of a humid tropical forest area. *Ecological Economics*, **3**, 43–57.

Hueting, R., L. Reijnders, B. de Boer, J. Lambooy and H. Jansen (1998), The concept of environmental function and its valuation. *Ecological Economics*, **25**, 31–5.

Hufschmidt, M.M., D.E. James, A.D. Meister, B.T. Bower and J.A. Dixon (1983), Environment, natural systems and development: an economic valuation guide. Baltimore, US: The Johns Hopkins University Press.

Huggett, R.J. (1993), *Modelling the Human Impact on Nature: Systems Analysis of Environmental Problems*, Oxford, UK: Oxford University Press.

Illius, A.W. and T.G. O'Connor (1999), On the relevance of non-equilibrium concepts to arid and semi-arid grazing systems. *Ecological Applications*, **9**, 798–813.

Imeson, A.C. and L.H. Cammeraat (2002), Environmentally sensitive areas in the Medalus target area study sites, in N.A. Geeson, C.J. Brandt and J.B. Thornes (eds), *Mediterranean Desertification: A Mosaic of Processes and Responses*, Chichester, UK: John Wiley and Sons Ltd, pp. 177–86.

IMF (1999), Press Release No. 99/29, Annex: Programme Summary Senegal Enhanced Structural Adjustment Facility. International Monetary Fund, Washington, DC.

IMF Commodity Prices (2003), http://www.imf.org/external/np/res/commod/index.asp.

Infield, M. (2001), Cultural values: a forgotten strategy for building community support for protected areas in Africa. *Conservation Biology*, **15**, 800–802.

IPCC (2007), Climate Change 2007: The Physical Science Basis. IPCC Fourth Assessment Report. Intergovernmental Panel on Climate Change, Cambridge University Press.

IUCN/UNEP/WWF (1980), World Conservation Strategy. IUCN, Gland, Switzerland.

Jackson, J.B.C., M.X. Kirby, W.H. Berger et al. (2001), Historical overfishing and the recent collapse of coastal ecosystems. *Science*, **293**, 629–37.

Janssen, M.A., J.M. Anderies and B.H. Walker (2004), Robust strategies for managing rangelands with mulitple stable attractors. *Journal of Environmental Economics and Management*, **47**, 140–62.

Jeppesen, E., J.P. Jensen, K. Kristensen et al. (1990), Fish manipulation as a lake restoration tool in shallow, eutrophic, temperate lakes 2: threshold levels, long-term stability and conclusions. *Hydrobiologia*, **200/201**, 219–27.

Johansson, P. (1999), Theory of economic valuation of environmental goods and services, in J.C.J.M. van de Bergh (ed.), *Handbook of Environmental Resource Economics*. Cheltenham, UK and Northampton, MA, USA: Edward Elgar, pp. 747–54.

Jollands, N., J. Lermit and M. Patterson (2003), The usefulness of aggregate indicators in policy making and evaluation: a discussion with application to eco-efficiency indicators in New Zealand. Palmerston North, New Zealand: AgResearch Limited.

Kahneman, D., J.L. Knetsch and R.H. Thaler (1986), Fairness as a constraint on profit seeking entitlements in the market. *American Economic Review*, **76** (4), 728–41.

Karp, L. and A. Pope III (1984), Range management under uncertainty. *American Journal of Agricultural Economics*, **66**, 437–46.

Khan, F.I., T. Husain and R. Hejazi (2004), An overview and analysis of site remediation technologies. *Journal of Environmental Management*, **71**, 95–122.

Khanna, N. and D. Chapman (1996), Time preference, abatement costs, and international climate policy: an appraisal of IPCC95. *Contemporary Economic Policy*, **14**, 56–66.

Kirk, J.T.O. (1994), *Light and photosynthesis in Aquatic Ecosystems*, Cambridge, UK: Cambridge University Press.

Kishor, N.M. and L.F. Constantino (1993), Forest management and competing land uses: an economic analysis for Costa Rica. Laten dissemination note no. 7, World Bank, Washington DC.

Klein, A.M., B. Vaissière, J.H. Cane et al. (2007), Importance of crop pollinators in changing landscapes for world crops. *Proceedings of the Royal Society of Biological Sciences*, **274**, 303–13.

Klinge, M. (1999), Fish management plan Northwest Overijssel (in Dutch: *Visstandbeheersplan Noordwest Overijssel*). Witteveen en Bos, Rotterdam, the Netherlands.

Klinge, M., M.P. Grimm and H. Hosper (1995), Eutrophication and ecological rehabilitation of Dutch lakes: presentation of a new conceptual framework. *Water Science and Technology*, **31**, 207–18.

Klug, S. (1982), Inventaire et suivi de la vegetation dans la parcelle d'elevage a Vindou-Tiengoly (Ferlo-Senegal). GTZ, Hamburg, Germany.

Knowlton, N. (1992), Thresholds and multiple stable states in coral reef community dynamics. *American Zoologist*, **32**, 674–82.

Kolstad, C.D. (2000), *Environmental Economics*, New York/Oxford: Oxford University Press.

Konow, J. (1996), A positive theory of economic fairness. *Journal of Economic Behaviour and Organization*, **31** (1), 13–35.

Konow, J. (2001), Fair and square: the four sides of distributive justice. *Journal of Economic Behaviour & Organization*, **46**, 137–64.

Koop, G. and L. Tole (2001), Deforestation, distribution and development. *Global Environmental Change*, **11**, 193–202.

Larsen, D.P., D.W. Schult, and K. Malueg (1981), Summer internal phosphorus supplies in Shagawa lake, Minnesota. *Limnology and Oceanography*, **26**, 740–53.

Laycock, W.A. (1991), Stable states and thresholds of range condition on North American rangelands: a viewpoint. *Journal of Range Management*, **44**, 427–33.

Leach, M., R. Mearns and I. Scoones (1999), Environmental entitlement: dynamics and institutions in community-based natural resource management. *World Development*, **27**(2), 225–47.

LeDoux, C.B. and N.K. Huyler (2000), Cost comparisons for three harvesting systems operating in northern hardwood stands. Research Paper NE-715. Northeastern Research Station, Forest Service, U.S. Department of Agriculture. Newtown Square, PA, USA.

Leemans, R. (2000), Modelling of global land use: connections, causal chains and integration. Inaugural lecture, Department of Plant Production Systems, Wageningen University, Wageningen, the Netherlands.

Le Houérou, H.N. (1984), Rain-use efficiency: a unifying concept in arid land ecology. *Journal of Arid Environments*, **7**, 213–47.

Le Houérou, H.N., R.L. Bingham and W. Skerbek (1988), Relationship between the variability of primary production and the variability of annual precipitation in world arid lands. *Journal of Arid Environments*, **15**, 1–18.

Levin, S.A. (1992), The problem of pattern and scale in ecology. *Ecology*, **73**, 1943–67.

Levin, S.A., S. Barrett, S. Aniyar et al. (1998), Resilience in natural and socio-economic systems. *Environment and Development Economics*, **3**, 222–35.

Likens, G.E. (1992), *The Ecosystem Approach: Its Use and Abuse*, Oldendorf/Luhe, Germany: Ecology Institute.

Limburg, K.E., R.V. O'Neill, R. Costanza and S. Farber (2002), Complex systems and valuation. *Ecological Economics*, **41**, 409–20.

Lind, R.C. (1982), *Discounting for Time and Risk in Energy Policy*, Resources for the Future, Washington, DC.

Lindebo, E. (1999), A review of fishing capacity and overcapacity. SJFI Working Paper no. 14/1999. Revised and extended version of 'A Review of the Fishing Capacity Concept' prepared for the XIth Annual Conference of EAFE. Dublin, 7–10 April 1999.

Little, I.M.D. and J.A. Mirrlees (1974), Project appraisal and planning for developing countries. Heinemann, London.

Little, I.M.D. and M.F. Scott (1976), *Using Shadow Prices*, London: Heinemann.

Ludwig, D., D.D. Jones and C.S. Holling (1978), Qualitative analysis of insect outbreak systems: the spruce-budworm and forest. *Journal of Animal Ecology*, **47**, 315–32.

Ludwig, J.A., M.B. Coughenour, A.C. Liedloff and R. Dyer (2001), Modelling the resilience of Australian savanna systems to grazing impacts. *Environment International*, **27**, 167–72.

Luijten, J.C., J.W. Jones et al. (2000), Dynamic modelling of strategic water availability in the Cabuyal River, Colombia: the impact of land cover change on the hydrological balance. *Advances in Environmental Monitoring and Modelling*, **1**, 36–60.

Mageau, M.T., R. Costanza and R.E. Ulanowicz (1998), Quantifying the trends expected in developing ecosystems. *Ecological Modelling*, **112**, 1–22.

Mäler, K.G. (2000), Development, ecological resources and their management: a study of complex dynamic systems. *European Economic Review*, **44**, 645–65.

Mäler, K.G., A. Xepapadeas and A. de Zeeuw (2003), The economics of shallow lakes. *Environmental and Resource Economics*, **26**, 603–24.

Mayeres, I., S. Proost, D. Vanderdruyssen et al. (2001), *The External Costs of Transportation*, Leuven: Centre for Economic Studies.

McArthur, D. and J.L. Dillon (1971), Risk, utility and stocking rate. *Australian Journal of Agricultural Economics*, **15**, 20–35.

McPeak, J.G. (2003), Analyzing and addressing localized degradation in the commons. *Land Economics*, **79**, 515–36.

Meijer, M.L. (2000), Biomanipulation in the Netherlands; 15 years of experience. PhD thesis Wageningen University.

Midmore, P. and L. Harrison-Mayfield (1996), Rural economic modelling: an input–output approach, Wallingford, Oxon, UK: CAB International.
Miehe, S. (1992), L'evolution de la strate herbacée au cours de la nouvelle sécheresse 1990–1992. Rapport d'une mission à court terme. GTZ, Eschborn, Germany.
Miehe, S. (1997), Inventaire et suivi de la végétation dans le périmètre expérimental à Widou-Thiengoly dans le cadre du projet Sénégalo-Allemand d'autopomotion pastorale dans le Ferlo (PAPF). Rapport 1997, GTZ, Eschborn, Germany.
Millennium Ecosystem Assessment (2003), Ecosystems and Human Well-being: A Framework for Assessment. Report of the conceptual framework working group of the Millennium Ecosystem Assessment. Island Press, Washington, DC.
Millennium Ecosystem Assessment (2005), Ecosystems and Human Well-being: Global Assessment Report Vol. 1 to 3. Island Press, Washington, DC.
Ministère de l'Hydraulique, Senegal (1987), Plannification des resources en eau – Bilan-diagnostique. Rapport SEN/87/006, Ministère de l'Hydraulique, Dakar, Senegal.
Ministry of Forestry of Indonesia (2006), Indonesia Case Study: Management Effectiveness Assessment of National Parks using WWF's RAPPAM Methodology. Directorate General of Forest Protection and Nature Conservation, Ministry of Forestry, Jakarta.
Mooney, H.A., J. Lubchenco, R. Dirzo and O.E. Sala (1995), Biodiversity and ecosystem functioning: ecosystem analyses, in V.H. Heywood and R.T. Watson (eds), *Global Biodiversity Assessment*, UNEP.
Morgan, R.P.C. (1995), *Soil Erosion and Conservation* (2nd edition), Harlow, UK: Longman Group Limited.
Mortimore, M.J. and W.M. Adams (2001), Farmer adaptation and 'crisis' in the Sahel. *Global Environmental Change*, **11**, 49–57.
Munasinghe, M. (1993), Environmental Economics and Sustainable Development. World Bank Environment Paper, No. 3. World Bank, Washington, DC.
Munasinghe, M.J. and MacNeely (1994), *Protected Area Economics and Policy: Linking Conservation and Sustainable Development*, Washington, DC: World Bank.
Munasinghe, M. and A. Schwab (1993), *Environmental Economics and Natural Resource Management in Developing Countries*, Washington, DC: World Bank.
Muradian, R. (2001), Ecological thresholds: a survey. *Ecological Economics*, **38**, 7–24.

Murray, A.G. and J.S. Parslow (1999), Modelling of nutrient impacts in Port Phillip Bay: a semi-enclosed marine Australian ecosystem. *Marine and Freshwater Research*, **50**, 597–611.

Myers, R.A. and B. Worm (2003), Rapid worldwide depletion of predatory fish communities. *Nature*, **6937**, 280–82.

Naevdal, E. (2001). Optimal regulation of eutrophying lakes, fjords and rivers in the presence of threshold effects. *American Journal of Agricultural Economics*, **83**(4), 972–84.

Natuurmonumenten (2000), Nature vision De Wieden (in Dutch: *Natuurvisie De Wieden*). Vereniging Natuurmonumenten, 's-Graveland, the Netherlands.

Natuurmonumenten (2003), Annual report Natuurmonumenten 2002. Vereniging Natuurmonumenten, 's-Graveland, the Netherlands.

Navrud, S. (2003), State-of-the-art on economic valuation of noise ECE/WHO 'Pan-European Program on Transport, Health and Environment'. Workshop on Economic Valuation of Health Effects due to Transport, June 12–13 2003, Stockholm.

Ndour, H. and A. Wané (1998), *Financial Intermediation and Poverty in Senegal*, Dakar, Senegal: EAGER Research.

Nearing, M.A., G.R. Foster, L.J. Jane and S.C. Finker (1989), A process-based soil erosion model for USDA Water Erosion Prediction Project technology. *Transactions of American Society of Agricultural Engineers*, **32**, 1587–93.

Nelson, E., G. Mendoza, J. Regetz et al. (2009), Modelling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment*, **7**, 4–11.

Netherlands Association for Dragonfly Studies (2002), *The Dutch Dragonflies (Odonata)* (In Dutch: *De Nederlandse libellen (Odonata)*), Leiden, the Netherlands: KNNV Publishers & European Invertebrate Survey.

Nicholson, S.E., C.J. Tucker and M.B. Ba (1998), Desertification, drought and surface vegetation: an example from the West-African Sahel. *Bulletin of the American Meteorological Society*, **79**, 815–29.

Nijkamp, P. and J. Spronk (1979), Analysis of production and location decisions by means of multi-criteria analysis. *Engineering and Process Economics*, **4**, 285–302.

Noordhuis, R., D.T. van der Molen and M.S. van den Berg (2002), Response of herbivorous waterbirds to the return of Chara in Lake Veluwemeer, the Netherlands. *Aquatic Botany*, **72**, 349–67.

Nordhaus, W.D. (1999), Discounting and public policies that affect the distant future, in P.R. Portney and J.P. Weyant (eds), *Discounting and*

Intergenerational Equity, Washington, DC: Resources for the Future, pp. 145–62.

Nordhaus, W. (2007), A review of the Stern review of the economics of climate change. *Journal of Economic Literature*, **45**(3), 686–702.

North, D.C. (1990), *Institutions, Institutional Change and Economic Performance*, Cambridge, UK: Cambridge University Press.

Nozick, R. (1974), *Anarchy, State, and Utopia*, New York: Basic Books.

Nunes, P.A.L.D. and J.C.J.M. van den Bergh (2001), Economic valuation of biodiversity: sense or nonsense? *Ecological Economics*, **39**, 203–22.

Nyström, M., C. Folke and F. Moberg (2000), Coral reef disturbance and resilience in a human-dominated environment. *Trends in Ecology and Evolution*, **15**, 413–17.

O'Connor, T.G., L.M. Haines and H.A. Snyman (2001), Influence of precipitation and species composition on phytomass of a semi-arid African grassland. *Journal of Ecology*, **89**(5), 850–60.

Odum, E.P. (1971), *Fundamentals of Ecology*, Philadelphia, USA: Saunders College Publishing.

Odum, E.P. and H.T. Odum (1972), Natural areas as necessary components of man's total environment, in Transactions of the 37th North American Wildlife and Natural Resources Conference 37, March 12–15, 1972. Wildlife Management Institute, Washington, DC, pp. 178–89.

OECD (1979), *Facing the Future: Mastering the Probable and Managing the Unpredictable*, Paris: InterFutures Study Team, Organisation for Economic Co-operation and Development.

OECD (1995), *The Economic Appraisal of Environmental Projects and Policies: A Practical Guide*, Paris: OECD.

OECD (2003), *OECD Environmental Indicators; Development, Measurement and Use*, Paris: OECD.

Office of the President, Republic of Guyana (2008), Creating Incentives to Avoid Deforestation. Office of the President, Georgetown, Guyana.

O'Neill, J. (2001), Representing people, representing nature, representing the world. Environment and planning. Part C: *Government and Policy*, **19**, 483–500.

O'Neill, R.V. and A.W. King (1998), Hommage to St Michael: or why are there so many books on scale? in D.L. Peterson and V.T. Parker (eds), *Ecological Scale: Theory and Applications*, New York: Columbia University Press, pp. 3–15.

Opschoor, H. and F. van der Ploeg (1990), Sustainability and quality: main goals of environmental policy (In Dutch: *Duurzaamheid en kwaliteit: hoofddoelstellingen van milieubeleid*), in *Commission Long-term Environmental Policy, 1990: The Environment: Concepts for the Twenty-*

first century (In Dutch: *Het milieu – denkbeelden voor de 21e eeuw*), the Netherlands: Zeist, pp. 81–128.

O'Riordan, T., C. Cooper, A. Jordan et al. (1998), Institutional frameworks for political action, in *Human Choice and Climate Change, Vol. 1: The Societal Framework*, Columbus, OH: Battelle Press, pp. 345–439.

Ostro, B. and L. Chestnut (1998), Assessing the health benefits of reducing particulate matter air pollution in the United States. *Environmental Research*, **76**, 94–106.

Pagiola, S. (2008), Payments for environmental services in Costa Rica. *Ecological Economics*, **65**, 712–24.

Pagiola, S., A. Arcenas and G. Platais (2005), Can payments for environmental services help reduce poverty? An exploration of the issues and the evidence to date from Latin America. *World Development*, **33**, 237–53.

Pain, D.J., A.A. Cunningham, P.F. Donald et al. (2003), Causes and effects of temporospatial declines of Gyps vultures in Asia. *Conservation Biology*, **17**, 661–71.

Palmer, A.R. (2000), Description of the grazing capacity information system (GCIS). Report Range and Forage Institute. Range and Forage Institute, Grahamstown.

Pauly, D., V. Christensen, J. Dalsgaard, R. Froese, F. Torres Jr. (1998), Fishing down marine food webs. *Science*, **279**, 860–63.

PBL (2008), Natuurbalans (the 'State of Nature'). Bilthoven, the Netherlands: PBL, 177 pp.

Pearce, D.W. (1995), *Capturing Global Environmental Value*, London: Earthscan.

Pearce, D.W. (2007), Do we really care about biodiversity? *Environmental and Resource Economics*, **37**, 313–33.

Pearce, D.W. and A. Howarth (2000), Technical report on methodology: cost–benefit analysis and policy responses. RIVM report 481505020.

Pearce, D.W. and D. Moran (1994), *The Economic Value of Biodiversity*, London: Earthscan in association with the IUCN.

Pearce, D.W. and C.G.T. Pearce (2001), The value of forest ecosystems. A report to the Secretariat of the Convention on Biological Diversity, Montreal, Canada.

Pearce, D.W. and R.K. Turner (1990), *Economics of Natural Resources and the Environment*, Hertfordshire, UK: Harvester Wheatsheaf.

Pearce D.W., G. Atkinson and S. Mourato (2005), *Cost–benefit Analysis and the Environment: Recent Developments*, Paris: Organisation for Economic Co-operation and Development.

Pearce, D.W., A. Markandya and E.B. Barbier (1989), *Blueprint for a Green Economy*, London: Earthscan.

Penning de Vries, F.W.T. and M.A. Djitèye (1982), La productivité des pâturages sahéliens, une étude des sols, des végétations et de l'exploitation de cette ressources naturelle. Centre for Agricultural Publishing and Documentation, Wageningen.

Penttinen, M.J. (2000), Timber Harvesting with Variable Prices and Costs. Working Paper 00045, International Institute for Applied Systems Analysis, Laxenburg, Austria.

Perman, R., Y. Ma, J. McGilvray and M. Common (1999), *Natural Resource and Environmental Economics*, 2nd edition, London/New York: Longman.

Perrings, C. (1998), Resilience in the dynamics of economy–environment systems. *Environmental and Resource Economics*, **11**, 503–20.

Perrings, C. (2000), Modelling sustainable ecological economic development. Paper presented to the 10th annual conference of the European Association of Environmental and Resource Economists, Crete, 2000.

Perrings, C. (1997), Stress, shock and the sustainability of resource use in semi-arid environments, in C. Perrings, *Economics of Ecological Resources*, Cheltenham, UK and Lyme, USA: Edward Elgar.

Perrings, C. and D.W. Pearce (1994), Threshold effects and incentives for the conservationof biodiversity. *Environmental and Resource Economics*, **4**, 13–28.

Perrings, C. and B.H. Walker (1997), Biodiversity, resilience and the control of ecological–economic systems: the case of fire-driven rangelands. *Ecological Economics*, **22**, 73–83.

Perrow, M.R., M.L. Meijer, P. Dawidowicz and H. Coops (1997), Biomanipulation in shallow lakes: state of the art. *Hydrobiologia*, **342/343**, 355–65.

Peterson G.D., S.R. Carpenter and W.A Brock (2003), Uncertainty and the management of multistate ecosystems: an apparently rational route to collapse. *Ecology*, **84**, 1403–11.

Petts, J. (ed.) (1999), *Handbook of Environmental Impact Assessment*, Vol 1 and 2, Oxford, UK: Blackwell.

Pezzey, J.C.V. and M.A. Toman (2002), The economics of sustainability: a review of articles. Discussion Paper 02–03. Resources for the Future, Washington, DC.

Pielou, E.C. (1969), *An Introduction to Mathematical Ecology*, New York: Wiley.

Pieterse, N.M., W. Bleuten and S.E. Jorgensen (2002), Contribution of diffuse sources to nitrogen and phosphorus loads in lowland river tributaries. *Journal of Hydrology*, **271**, 231–25.

Pimm, S.L. (1984), The complexity and stability of ecosystems. *Nature*, **307**, 321–6.

Pirot, J.-Y., P.J. Meynell and D. Elder (2000), *Ecosystem Management: Lessons From Around the World*, Gland, Switzerland: World Conservation Union.

Posey, D.A. (1999), *Cultural and Spiritual Values of Biodiversity. A Complementary Contribution to the Global Biodiversity Assessment*, London: Intermediate Technology Publications.

Put, M., J. Verhagen, E. Veldhizen and P. Jellema (2004), Climate change in dryland West-Africa? The empirical evidence of rainfall variability and trends, in A.J. Dietz, R. Ruben and A. Verhagen (eds), *The Impact of Climate Change on Drylands*, Dordrecht, the Netherlands: Kluwer Academic Publishers, pp. 27–32.

Rabl, A. and J.V. Spadaro. (1999), Damages and costs of air pollution: an analysis of uncertainties. *Environment International*, **25**(1), 29–46.

Randall, A. and M.C. Farmer (1995), Benefits, costs, and the safe minimum standard of conservation, in D.W. Bromley (ed.), *The Handbook of Environmental Economics*, Oxford, UK: Blackwell, pp. 26–44.

Rastetter, E.B. (1996), Validating models of ecosystem response to global change. *BioScience*, **46**, 190–98.

Rawls, J. (1971), *A Theory of Justice*, Cambridge, MA: Belknap Press of Harvard University Press.

Reed, W.J. (1974), A stochastic model for the economic management of a renewable resource. *Mathematical Biosciences*, **22**, 313–37.

Reed, W.J. (1988), Optimal harvesting of a fishery subject to random catastrophic collapse. *IMA Journal of Mathematics Applied in Medicine and Biology*, **5**, 215–35.

Renn, O. (2006), Participatory processes for designing environmental policies. *Land Use Policy*, **2**, 34–43.

Renn, O., T. Webler, H. Rakel, P. Dienel and B. Johnson (1993), Public participation in decision making: a three step procedure. *Policy Sciences*, **26**, 189–214.

Renn, O., T. Webler and P. Wiedeman (eds) (1995), *Fairness and Competence in Citizen Participation. Evaluating New Models for Environmental Discourse*, Dordrecht/Boston: Kluwer.

Ricketts, T.H., G.C. Daily, P.R. Ehrlich and C.D. Michener (2004), Economic value of tropical forest to coffee production. *Proceedings of the National Academy of Sciences of the United States of America*, **101**, 12579–82.

RIVM/UNEP (1997), The future of the global environment: a model based analysis supporting UNEP's first global environment outlook. J.A. Bakkes and J.W. van Woerden (eds), RIVM 402001007 and UNEP/DEIA/TR.97-1.

RIZA (1997), Biomanipulation in the Netherlands (In Dutch: *Actief*

Biologisch beheer in Nederland). RIZA, report 97.084. RIZA, Lelystad, the Netherlands.
Rockström, J., W. Steffen, K. Noone et al. (2009), A safe operating space for humanity. *Nature*, **461**, 472–5.
Rodriguez, A. and R.G. Taylor (1988), Stochastic modelling of short-term cattle operations. *American Journal of Agricultural Economics*, **70**, 121–32.
Roemer, J.E. (1996), *Theories of Distributive Justice*, Cambridge, MA: Harvard University Press.
Romero, J.A., H. Brix and F.A. Comín (1999), Interactive effects of N and P growth, nutrient allocation and NH_4 uptake kinetics by *Phragmites australis*. *Aquatic Botany*, **64**, 369–380.
Rosa, H., S. Kandel and L. Dimas (2004), Compensation for environmental services and rural communities: lessons from the Americas, *International Forestry Review*, **6**, 187–94.
Rotmans, J. and M.B.A. van Asselt (2001), Uncertainty in integrated assessment modelling towards a pluralistic approach. *Environmental Monitoring and Assessment*, **69**, 101–30.
Rounsevell, M.D.A., F. Ewert, I. Reginster, R. Leemans and T.D. Carter (2005), Future scenarios of European agricultural land use. II. Projecting changes in cropland and grassland. *Agriculture Ecosystems and Environment*, **107**, 117–35.
Rounsevell, M.D.A., I. Reginster, M.B. Araújo et al. (2006), A coherent set of future land use change scenarios for Europe. *Agriculture, Ecosystems & Environment*, **114**(1), 57–68.
Samuelson, P. (1976), Economics of forestry in an evolving society. *Economic Inquiry*, **14**, 466–92.
Schaefer, M.D. (1957), Some considerations of population dynamics and economics in relation to the management of marine fisheries. *Journal of Fisheries Research, Board of Canada*, **14**, 669–81.
Scheffer, M. (1998), *Ecology of Shallow Lakes*, Dordrecht, the Netherlands: Kluwer Academic Publishers.
Scheffer, M. and S.R. Carpenter (2003), Catastrophic regime shifts in ecosystems: linking theory to observation. *Trends in Ecology and Evolution*, **18**, 648–56.
Scheffer, M., S.H. Hosper, M.L. Meijer and B. Moss (1993), Alternative equilibria in shallow lakes. *Trends in Ecology and Evolution*, **8**, 275–9.
Scheffer, M., M. van den Berg, A. Beukelaar et al. (1994), Vegetated areas with clear water in turbid shallow lakes. *Aquatic Botany*, **6**, 193–6.
Scheffer, M., W. Brock and F. Westley (2000), Socio-economic mechanisms preventing optimum use of ecosystem services: an interdisciplinary theoretical analysis. *Ecosystems*, **3**, 451–71.

Scheffer, M., S. Carpenter, J.A. Foley, C. Folke and B. Walker (2001), Catastrophic shifts in ecosystems. *Nature*, **413**, 591–6.

Schmeer, K. (1999), Guidelines for conducting a stakeholder analysis. November 1999. Partnerships for Health Reform, Abt Associates Inc, Bethesda, MD, USA.

Schmieman, E.C. (2001), Acidification and trophospheric ozone in Europe; towards a dynamic economic analysis. PhD thesis Wageningen University, 2001.

Scholes, R.J. and B.H. Walker (1993), *An African Savanna: Synthesis of the Nylsvley Study*. Cambridge, UK: Cambridge University Press.

Schoorl, J.M., A. Veldkamp and J. Bouma. (2002), Modelling water and soil redistribution in a dynamic landscape context. *Soil Science, Society of America Journal*, **66**, 1610–19.

Scoones, I. (1994), New directions in pastoral development in Africa, in I. Scoones (ed.), *Living with Uncertainty*, London: Intermediate Technology Publications.

Sen, A. (1970), *Collective Choice and Social Welfare*, San Francisco: Holden Day.

Simmonds, E.J. (2007), Comparison of two periods of North Sea herring stock management: success, failure, and monetary value. *Journal of Marine Science*, **64**, 686–92.

Sinclair A.R.E. and J.M. Fryxell (1985), The Sahel of Africa: ecology of a disaster. *Canadian Journal of Zoology*, 987–94.

Smith, C.T., A.T. Lowe, M.F. Skinner et al. (2000), Response of radiata pine forests to residues management and fertilisation across a fertility gradient in New Zealand. *Forest Ecology and Management*, **138**, 203–23.

Smith, R.L. (1990), *Ecology and Field Biology*, New York: Harper and Row.

Snyman, H.A. (1998), Dynamics and sustainable utilization of the rangeland ecosystem in arid and semi-arid climates of Southern Africa. *Journal of Arid Environments*, **9**, 645–66.

Solow, R.M. (1956), A contribution to the theory of economic growth. *Quarterly Journal of Economics*, **70**, 65–94.

Solow, R.M. (1993), An almost practical step toward sustainability. *Resources Policy*, **19**, 162–72.

Solow, R.M. (1999), Foreword, in P.R. Portney and J.P. Weyant (eds), *Discounting and Intergenerational Equity*, Washington, DC: Resources for the Future.

Soma, K. and A. Vatn (2009), Local democracy implications for coastal zone management: A case study in southern Norway. *Land Use Policy*, **26**, 755–62.

Sommer, H., N. Künzli, R. Seethaler et al. (2000), Economic Evaluation of Health Impacts Due to Road Traffic-Related Air Pollution: An Impact Assessment Project of Austria, France, and Switzerland, in OECD et al., Ancillary Benefits and Costs of Greenhouse Gas Mitigation, Proceedings of an Expert Workshop, Paris.

Søndergaard, M., P. Kristensen and E. Jeppesen (1993), Eight years of internal phosphorus loading and changes in sediment phosphorus profile of Lake Sobygaard, Denmark. *Hydrobiologia*, **253**, 345–56.

Spash, C.L. (2008), How much is that ecosystem in the window? The one with the bio-diverse trail. *Environmental Values*, **17**, 259–84.

Spash, C.L. and N. Hanley (1995), Preferences, information and biodiversity preservation. *Ecological Economics*, **12**, 191–208.

Squire, L. and H.G. van der Tak (1975), *Economic Analysis of Projects*, Baltimore, MD: Johns Hopkins University Press.

Steele, J.H. (1998), Regime shifts in marine ecosystems. *Ecological Applications*, S33–S36.

Steele, J.H. and E.W. Henderson (1984), Modelling long-term fluctuations in fish stocks. *Science*, **224**, 985–7.

Stern, N. (2008), The economics of climate change. *American Economic Review: Papers & Proceedings*, **98**, 2–37.

Streeten, P. (1980), Basic needs and human rights. *World Development*, **8**, 107–11.

Strijker, D., F.J. Sijtsma and D. Wiersma (2000), Evaluation of nature conservation; an application to the Dutch Ecological Network. *Environmental and Resource Economics*, **16**, 363–78.

Sullivan, S. and R. Rohde (2002), On non-equilibrium in arid and semi-arid grazing systems. *Journal of Biogeography*, **29**, 1595–618.

Sutter, J.W. (1987), Cattle and inequality: herd size differences and pastoral production among the Fulani of North Eastern Senegal. *Africa*, **57**, 196–218.

Svedang, H., E. Neuman and H. Wiskstrom (1996), Maturation patterns in female European eel: age and size at the silver eel stage. *Journal of Fish Biology*, **48**, 342–51.

Swanson, T. (1997), *Global Action for Biodiversity*. London: Earthscan.

Tacconi, L. (2000), *Biodiversity and Ecological Economics, Participation, Values, and Resource Management*. London: Earthscan.

Tahvonen, O. (1991), On the dynamics of renewable resource harvesting and pollution control. *Environmental and Resource Economics*, **1**, 97–118.

Tahvonen, O. (2004), Timber production versus old-growth preservation with endogenous prices and forest age-classes. *Canadian Journal of Forest Research*, **34**(6), 1296–310.

Talberth, J., C. Cobb and N. Slattery (2007), The Genuine Progress Indicator 2006, A Tool for Sustainable Development. Redefining Progress, Oakland, USA.

Tansley, A.G. (1935), The use and abuse of vegetational terms and concepts. *Ecology*, **16**, 284–307.

Ten Brink, B.J.E. and T. Stekelenburg (2002), Biodiversity: how much is left? The Natural Capital Index framework (NCI). RIVM report 402001014. Bilthoven.

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

Thébaud, B., H. Grell and S. Miehe (1995), Recognising the effectiveness of traditional grazing practices: lessons from a controlled grazing experiment in northern Senegal. Drylands Programme, International Institute for Environment and Development, London. Paper No. 55.

Tietenberg, T.H. (2000), *Environmental and Natural Resource Economics* (5th edition), Reading, MA, USA: Addison Wesley Longman, Inc.

Tilman, D. and J.A. Downing (1994), Biodiversity and stability in grasslands. *Nature*, **367**, 363–5.

Timms, R.M. and B. Moss (1984), Prevention of growth of potentially dense pytoplankton populations by zooplankton grazing in presence of zooplanktivorous fish in a shallow wetland ecosystem. *Limnology and Oceanography*, **29**, 472–86.

Toth, F.L. (1995), Practice and progress in integrated assessments of climate change, a workshop overview. *Energy Policy*, **23**, 253–66.

Turner, M.D. (1999), Spatial and temporal scaling of grazing impact on the species composition and productivity of Sahelian annual grasslands. *Journal of Arid Environments*, **41**, 277–97.

Turner, M.D. and T.O. Williams (2002), Livestock market dynamics and local vulnerabilities in the Sahel. *World Development*, **30**, 683–705.

Turner, R.K., C.J.M. van den Bergh, T. Soderqvist et al. (2000), Ecological–economic analysis of wetlands: scientific integration for management and policy. *Ecological Economics*, **35**, 7–23.

Turner, R.K., J. Paavola, P. Coopera et al. (2003), Valuing nature: lessons learned and future research directions. *Ecological Economics*, **46**, 493–510.

Uri, N.D. and J.A. Lewis (1998), The dynamics of soil erosion in US agriculture. *Science of the Total Environment*, **218**, 45–58.

UN (1992), Convention Text, Convention on Biological Diversity, United Nations Conference on Environment and Development, Rio de Janeiro, 1992.

UN (2003), World Population Prospects; The 2002 Revision. Population

Division, Department of Economics and Social Affairs, United Nations, New York.
UNEP (1995), *Global Biodiversity Assessment*, in V.H. Heywood and R.T. Watson (eds), Cambridge, UK: Cambridge University Press.
UNEP (2005), Discussion Paper Multilateral Environmental Agreements and Pro-Poor Markets for Ecosystem Services. UNEP, Nairobi, 19 pp.
Van Berkum, J.A. (2000), *Waterplants and De Wieden: Rehabilitation of the Water Quality* (In Dutch: *Waterplant en Wieden: herstel van helder water*), Zwolle, the Netherlands: Waterboard Groot Salland.
Van den Bergh, J.C.J.M. (1996), *Ecological Economics and Sustainable Development: Theories, Methods and Applications*, Cheltenham, UK and Brookfield, USA: Edward Elgar.
Van den Bergh, J.C.J.M., G.M.M. Gelauff and V.R. Okker (1988), The Freia-Kompas model for the Netherlands. *Economic Modelling*, **5**, 170–235.
Van den Hove, S. (2000), Participatory approaches to environmental policy-making: the European Commission Climate Policy Process as a case study. *Ecological Economics*, **33**, 57–572.
Van der Maarel, E. and P.L. Dauvellier (1978), Towards a global ecological model (GEM) for the spatial planning of the Netherlands (In Dutch: *Naar een globaal ecologisch model (GEM) voor de ruimtelijke ordening van Nederland*), Ministry of Housing, Spatial Planning and Environment, The Hague, the Netherlands.
Van der Molen, D.T., R. Portielje, P.C.M. Boers and L. Lijklema (1998), Changes in sediment phosphorus as a result of eutrophication and oligotrophication in Lake Veluwe, the Netherlands. *Water Research*, **32**, 3281–8.
Van der Sluijs, J.P. (1997), Anchoring amid uncertainty: on the management of uncertainties in risk assessment of anthropogenic climate change. PhD thesis, Utrecht University.
Van der Veeren, R.J.H.M. (2002), Economic analyses of nutrient abatement policies in the Rhine basin. PhD Thesis, Free University Amsterdam, the Netherlands.
Van der Veeren, R.J.H.M. and C.M. Lorenz (2002), Integrated economic–ecological analysis and evaluation of management strategies on nutrient abatement in the Rhine Basin. *Journal of Environmental Management*, **66**, 361–76.
Van Dijk, A. (2003), Personal communication with A. van Dijk, Secretary of the Inland Fishermen organisation Northwest Overijssel.
Van Konijnenburg, P.G. (1996), Recreation in De Wieden (in Dutch: *Recreatieondertzoek De Wieden*). Natuurmonumenten, 's-Graveland, the Netherlands.

Van Kooten, G.C. and E.H. Bulte (2000), *The Economics of Nature: Managing Biological Assets*, Malden, USA: Blackwell Publishers Inc.

Van Kooten, G.C., C. Binkley and G. Delcourt (1995), Effect of carbon taxes and subsidies on optimal forest rotation age and supply of carbon services. *American Journal of Agricultural Economics*, **46**, 273–85.

Van Koppen, C.S.A. (2000), Resource, Arcadia, Lifeworld. Nature concepts in environmental sociology. *Sociologia Ruralis*, **40**, 300–318.

Van Nes, E.H., M. Scheffer, M.S. van den Berg and H. Coops (2002), Dominance of charophytes in eutrophic shallow lakes – when should we expect it to be in an alternative stable state? *Aquatic Botany*, **72**, 275–96.

Van Wee, B., R. Van den Brink and H. Nijland (2003), Environmental impacts of high-speed rail links in cost–benefit analyses: a case study of the Dutch Zuider Zee line, *Transportation Research Part D*, **8**, 299–314.

Varian, H.R. (1993), *Intermediate Micro-economics*, 3rd edition, New York: W.W. Norton & Company.

Vatn, A. (2005), *Institutions and the Environment*, Cheltenham, UK and Northampton, MA, USA: Edward Elgar.

Vermeulen, S. and I. Koziell (2002), *Integrating Global and Local Values. A Review of Biodiversity Assessment*, London: IIED.

Vetter, S. (2005), Rangelands at equilibrium and non-equilibrium: recent developments in the debate. *Journal of Arid Environments*, **62**, 321–41.

Victor, P.A. (1994), Natural capital, substitution, and indicators of sustainable development. *Journal of Public Economics*, **86**, 341–60.

Voinov, A., R. Costanza, L. Wainger et al. (1999), Patuxent landscape model: integrated ecological economic modelling of a watershed. *Environmental Modelling and Software*, **14**, 473–91.

Voinov, A., H. Voinov and R. Costanza (1999), Surface water flow in landscape models, 2. Patuxent watershed case study. *Ecological Modelling*, **119**(2–3), 211–30.

Vollenweider, R.A. (1968), Scientific fundamentals of the eutrophication of lakes and flowing waters, with special reference to nitrogen and phosphorous as factors in eutrophication. OECD Technical Report DAS/CSI/68, 27. OECD, Paris.

Von Thünen, J.H. (1863), Der isolierte Staat in Beziehung auf Landwirtschaft und Nationalökonomie. Dritter Teil. Wissenschaftliche Buchgesellschaft, Darmstadt, Germany, 1966.

VW (1998), Fourth National Policy Document on Water Management; Government Decision. Ministry of Transport, Public Works and Water Management, the Hague, the Netherlands.

Walker, B.H. (1981), Is succession a viable concept in African savanna ecosystems? in D.C. West, H. Shugart and D.B. Botkin (eds) *Forest Succession: Concepts and Application*, New York, USA: Springer-Verlag.

Walker, B.H. (1992), Biological diversity and ecological redundancy. *Conservation Biology*, **6**, 18–23.

Walker, B.H. (1993), Rangeland ecology: understanding and managing change. *Ambio*, **22**, 80–87.

Walker, B.H. (1995), Conserving biological diversity through ecosystem resilience. *Conservation Biology*, **9**, 747–52.

Walker, B.H. and N. Abel (2002), Resilient rangelands: adaptation in complex systems, in L.H. Gunderson and C.S. Holling (eds), *Panarchy, Understanding Transformations in Human and Natural Systems*, Washington, DC: Island Press, pp. 293–314.

Walker, B.H. and I. Noy-Meir (1982), Aspects of the stability and resilience of savanna ecosystems, in B.J. Huntley and B.H. Walker (eds), *Ecology of Tropical Savannas*, Berlin: Springer, pp. 577–90.

Walter, H. (1971), *Ecology of Tropical and Sub-tropical Vegetation*, Edinburgh: Oliver and Boyd.

Walters, D.V. (1968), The most profitable sheep stocking rate. *Farm Management*, **4**, 13–22.

Walzer, M. (1983), *Spheres of Justice: A Defense of Pluralism and Equality*, New York: Basic Books.

Wathern, P., S.N. Young, I.W. Brown and D.A. Roberts (1986), Ecological evaluation tecniques. *Landscape Planning*, **12**, 403–20.

Waterboard Reest and Wieden (2003), Database with water quality samples period 1970–2003. Waterboard Reest en Wieden, Meppel, the Netherlands.

Waterschap Groot Salland (2000), Watershed analysis Northwest Overijssel (in Dutch: *Stroomgebiedsanalyse Noordwest Overijssel*). Waterschap Groot Salland, Zwolle, the Netherlands.

Watt, A.S. (1947), Pattern and process in the plant community. *Journal of Ecology*, **35**, 1–22.

WCED (World Commission on the Environment and Development) (1987), *Our Common Future*, Oxford, UK: Oxford University Press.

Weaver, J.E. and F.E. Clements (1938), *Plant Ecology* (2nd edition), New York: McGraw-Hill.

Weikard, H.P. (2002), The existence value does not exist and non-use values are useless. Paper presented at the Annual meeting of the European Public Choice Society, Belgirate, Italy, 2002.

Weikard, H.P. (2003), On the quasi-option value of biodiversity and conservation, in J. Wesseler, H.P. Weikard and R. Weaver (eds), *Risk and Uncertainty in Environmental and Resource Economics*, Cheltenham, UK and Northampton, MA, USA: Edward Elgar, pp. 23–37.

Weisbrod, B.A. (1964), Collective consumption services of individual consumption goods. *Quarterly Journal of Economics*, **78**, 471–7.
Weitzman, M.L. (2007), A review of the Stern review of the economics of climate change. *Journal of Economic Literature*, **45**(3), 703–24.
West, O. (1971), Fire, man and wildlife as interacting factors limiting the development of climax vegetation in Rhodesia, in *Procedings of the Tall Timbers Fire Ecology Conference*, pp. 121–45.
Westoby, M., B.H. Walker and I. Noy-Meir (1989), Opportunistics management for rangelands not at equilibrium. *Journal of Rangeland Management*, **42**, 266–74.
Wiens, J. (1984), On understanding a non-equilibrium world: myth and reality in community patterns and processes, in D.R. Strong, D. Simberloff, L.G. Abele and A.B. Thistle (eds), *Ecological Communities: Conceptual Issues and the Evidence*, Princeton, USA: Princeton University Press, pp. 439–57.
Willemen, L., P.H. Verburg, L. Hein and M.E.F. van Mensvoort (2008), Spatial characterization of landscape functions. *Landscape and Urban Planning*, **88**(2), 34–43.
Willemen, L., L. Hein, M.E.F. van Mensvoort and P.H. Verburg (2009), Space for people, plants, and livestock? Quantifying interactions among multiple landscape functions in a Dutch rural region. *Ecological Indicators*, **10**(1), 62–73.
Williamson, J.R. and W.A. Nielsen (2003), The effect of soil compaction, profile disturbance, and fertilizer application on the growth of eucalypt seedlings in two glasshouse studies. *Soil and Tillage Research*, **71**, 95–107.
Willig, R. (1976), Consumer's surplus without apology. *American Economic Review*, **66**, 589–97.
Willis, K.G. and G.D. Garrod (1995), Transferability of benefit estimates, in K.G. Willis and J.T. Corkindale (eds), *Environmental Valuation: New Perspectives*, Wallingford, UK: CAB International.
Wilson, M.A. and S.R. Carpenter (1999), Economic valuation of freshwater ecosystem services in the United States 1971–1997. *Ecological Applications*, **9**, 772–83.
Wissel, C. (1984), A universal law of the characteristic return time near thresholds. *Oecologia*, **109**, 101–7.
Wik, M., T.A. Kebede, O. Bergland and S.T. Holden (2004), On the measurement of risk aversion from experimental data. *Applied Economics*, **36**(2), 2443–51.
Wolff, W.J. (2000), The south-eastern North Sea; losses of vertebrate fauna during the past 2000 years. *Biological Conservation*, **95**, 209–17.
Wulff, F., Bonsdorff, E., I.-M. Gren and A. Stigebrandt (2001), Giving

advice on cost effective measures for a cleaner Baltic Sea: a challenge for science. *Ambi*, **30**, 254–59.
Wunder, S. (2007), The efficiency of payments for environmental services in tropical conservation. *Conservation Biology*, **21**(1), 48–58.
Young, H.P. (1994), *Equity in Theory and Practice*, Princeton, NJ: Princeton University Press.

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