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An ecological-economic modelling approach

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Preface

This study examines the implications of complex ecosystem dynamics for the economic efficient and sustainable management of ecosystems. It integrates ecological and economic methodologies in order to identify optimal ecosystem management strategies. My interest in the combination of ecology and economics started during my MSc, when I followed a range of courses in both disciplines. In the 10 years that have passed since my MSc graduation, I have worked in various positions: as applied researcher at Utrecht University, as freelance consultant, and as environmental expert at the FAO Investment Centre in Rome. In these jobs, I was frequently exposed to both the ecological and economic sides of environmental and natural resource management. Increasingly, it has become clear to me that, in order to find optimal solutions to environmental and natural resource management issues, ecological and economic aspects need to be considered in an integrated manner. This involves the joint consideration of the ecological and economic dimensions of the environmental system in all steps of the analysis, from problem identification to the formulation of enhanced resource management strategies.

The linking of ecological and economic aspects in integrated analysis is no easy task, in particular because the two disciplines have developed specific, and not always easily compatible, sets of concepts and methodologies. In addition, integrated analysis of environmental systems generally requires substantial amounts of data, covering both biophysical and economic variables. In my previous jobs, there was usually not enough time and/or data to conduct an in-depth assessment of the environmental system, and recommendations for environmental and natural resource management strategies needed to be based upon limited analysis, and/or a 'best-professional judgement' of various experts. This approach is adequate for many situations, but, in order to enhance my own understanding of natural resource management issues, I became more and more interested in also conducting an in-depth study that integrates ecology and economics in studying a relevant resource management issue.

Therefore, I was very keen to start with this PhD study at Wageningen University. Although the funding for my appointment came from two other projects, these other projects left me enough time to work on my PhD study. In terms of results, I feel that the study demonstrates that the integrated economic and ecological analysis of natural resource management questions is very useful, and that it can assist in the selection of enhanced resource management strategies. The research also shows that a 'best professional judgement' approach does not always provide the best basis for setting up a natural resource management strategy. One of the projects I worked on several years ago concerned the preparation of a project proposal to support the further development of the livestock sector in northern Senegal. Based upon the available information, the project proposal comprised a mix of measures aimed at enhancing the numbers and productivity of livestock in the area – such as the drilling of new wells and enhanced veterinary aid. The project team, including myself as natural resources management expert, assumed that this project would lead to making better use of the local natural resource base – and there were no studies indicating otherwise. As part of my PhD study, I had the time to look at the case of grazing in northern Senegal in more detail. Based upon a more in-depth analysis, my study shows that the current grazing pressure in the area is already significantly higher than the economic optimum grazing pressure – and that increasing livestock numbers is contra-productive from an economic perspective. Government actions should be aimed at reducing livestock numbers instead of increasing them (see chapter 7). To me, this illustrates how important it is to understand your natural resource management system before you plan on changing it based upon simplified assumptions regarding the current state of the system, or its response to management.

Conducting a PhD study is, to a large extent, a solitary affair that involves thinking, modelling and writing in the company of your computer and scientific literature. Nevertheless, the support and feedback from supervisors, colleagues, friends and family is indispensable and I would like to very much thank the people that supported me in the past 3 years. First and foremost, I want to thank my two promoters, Leen Hordijk and Ekko van Ierland, who spent a considerable amount of effort in guiding me through the process of writing a PhD thesis. Both have been closely involved in this project from the start, which has helped me to maintain the quality of the research. Furthermore, I want to thank Dolf de Groot and Rik Leemans, who have made an important contribution to this study by reviewing different chapters and by organising the financial basis to conduct the research. Several people have contributed to or reviewed specific chapters, in particular Hans-Peter Weikard (environmental economics), Jeroen de Klein (aquatic ecology), Kris van Koppen (environmental policy) and Sip van Wieren (grazing ecology).

Many thanks are also due to the colleagues of the Environmental Systems Analysis Group at Wageningen University for providing a very pleasant and productive working atmosphere. Finally, I want to especially thank my partner Katrine for her support, and her interest in my research. Our daughter Kari, who is 9 months old at the time this book is printed, provided all the entertainment required to deal with the challenges of the final part of the PhD research, and at the same time gave me inspiration to continue studying options to enhance the use of natural resources.

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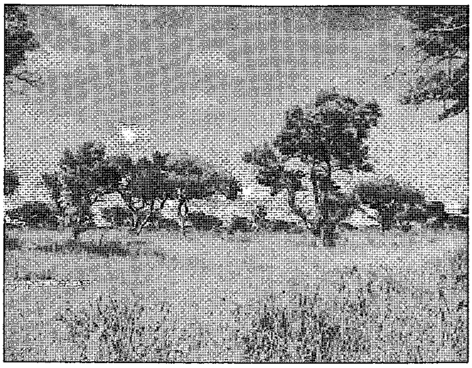
1. Introduction



1.1 Background

The setting

The coming decades will show a further increase in global population levels, from the current 6 billion to some 9 billion in 2050 (medium population scenario, UN, 2003). In addition, per capita production needs to grow in order to allow for an increase in living conditions, particularly in developing countries. These two factors will result in further pressure on the world's ecosystems. Already, in many parts of the world, human pressure has led to large scale conversion or degradation of ecosystems (Balmford et al., 2002). However, ecosystems supply a range of goods and services to mankind, including such essential services as the regulation of climatic and biochemical processes. In addition, ecosystems have a non-use value that warrants their conservation and their handing over to future generations. Hence, reconciling economic development with the sustainable management of the world's ecosystems is one of today's main challenges. One of the requirements for this reconciliation is decision making that takes into account both the *efficiency* and *sustainability* aspects of ecosystem management.

Efficient management is often interpreted in terms of maximising the present value of the net current and discounted future benefits derived from a system. 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. This 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 fisheries, 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 services supplied by the ecosystem should be considered (cf. Turner et al., 2003).

The Hotelling rule compares the intertemporal aspects of resource use on the basis of the social discount rate. However, this approach has been criticised as ethically questionable because of the large weight it attaches to the welfare of current generations as compared to the welfare of future generations. Partly in response to this shortcoming, the concept of *sustainability* has been introduced (IUCN/UNEP/WWF, 1980). The well-known 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). Subsequent to the publication of the Brundtland report, a range of different interpretations of the sustainability concept has been developed, including different types of weak and strong sustainability (see e.g. Carter, 2001). In general, weak sustainability allows for the substitution between natural, human and man-made capital, although in some interpretations of weak sustainability it is explicitly recognised that life supporting ecosystem services need to be maintained (e.g. Toman et al., 1995). The main principle of strong sustainability is that the total natural capital stock should not be further reduced (e.g. Pearce et al., 1989; and Barbier and Markandya, 1990). Hence, an important issue in interpreting sustainable development is to what extent natural capital can be replaced by man-made capital (Carter, 2001; Pezzey and Toman, 2002).

Problem statement

In the last decades, a wide range of ecological-economic models has been developed to analyse or predict the efficiency and/or sustainability of ecosystem management. Particular attention has been given to the analysis of modifications in ecosystems in response to pressures generated by the economic system (e.g. Braat and Van Lierop, 1987; Van Ierland, 1993), as well as to the economic consequences of changes in ecosystems (Costanza et al., 1993; Balmford et al., 2002). A methodological synthesis on integrating economic and ecological systems in a modelling approach can be found in, for example, Wang et al. (2001). A key element in assessing the efficiency and sustainability of ecosystem management is the analysis of the *dynamics* of the ecosystem, *i.e.* the development of the ecosystem, and its capacity to supply ecosystem services, over time.

The early economic models dealing with ecosystems assumed that ecosystem changes proceed in a gradual and reversible manner (e.g. Gordon, 1954; Hildreth and Riewe, 1963; Munro, 1982). However, starting in the mid 1970s, ecological studies increasingly pointed out that ecosystems often change in much more complex ways (Holling, 1973; Ludwig et al., 1978; Westoby et al., 1989; Scheffer et al., 2001). Complex ecosystem dynamics comprise irreversible, non-linear and/or stochastic responses of the ecosystem to human and/or ecological drivers. Irreversible ecosystem changes can not, or only to a limited extent be undone through natural processes. Non-linear responses involve, for instance, steady states and thresholds. Steady states are relatively stable configurations of the ecosystem, whereas thresholds involve sudden and strong changes in the ecosystem state in response to a relatively minor disturbance. Stochasticity occurs when ecosystem changes are triggered by stochastic events such as fire or heavy rainfall.

Complex dynamics are of major importance for the understanding of the dynamics of a range of ecosystems. These include 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; Nyström et al., 2000) and coastal estuaries (Murray and Parslow, 1999). For instance, shallow freshwater lakes can, under certain conditions, be in either of two states: a clear water state with waterplants and a fish community dominated by piscivorous fish; or a turbid water state with high phytoplankton concentrations, and dominated by benthivorous fish (Timms and Moss, 1984). The two states represent alternative equilibriums that exist over a certain range of nutrient conditions. Bifurcations from one state to the next take place once a critical nutrient level is passed, or as a function of a disturbance or human modification of the system (Scheffer, 1998).

The implications of these complex dynamics for efficient and sustainable ecosystem management have increasingly been recognised in environmental economics (Deacon et al., 1998; Turner et al., 2003). Ciriacy-Wantrup (1968) already proposed a 'safe minimum standard of conservation' as a means of incorporating uncertainty and irreversibility in the appraisal of natural resource utilization. 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. Perrings and Pearce (1994) demonstrate that it is crucial to account for non-linearities and threshold effects in the economic analyses of ecosystems that experience such dynamics, and provide a general, conceptual model for doing so. Arrow et al. (1995) analyse the implications of discontinuous and irreversible ecosystem change for economic systems, and state that ecosystem management needs to ensure the resilience of ecosystems. In addition, a large number of case studies has been conducted that analyse the implications of complex dynamics for the efficient management of specific ecosystems. 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. Carpenter et al. (1999) and Wu and Skelton-Growth (2002) examine economic efficient management of lakes respectively small watersheds subject to multiple states and thresholds. These, and other, studies show that complex dynamics have a major impact on the responses of the ecosystem to management. Consequently, complex dynamics have major implications for the efficient and sustainable management of natural resources (cf. Perrings, 1998).

Although substantial progress has been made in incorporating complex dynamics in ecological-economic models, there is still a need to further examine the implications of complex dynamics for the efficient and sustainable management of ecosystems (Van den Bergh, 1996; Perrings, 1998; Hanley, 1999). Many models consider economy-ecosystem interactions at an aggregated level, and do not explicitly account for ecosystem complexities such as irreversible responses, multiple states, thresholds, and/or stochasticity (Deacon et al., 1998). In addition, where monodisciplinary approaches towards ecological-economic modelling have been followed, the models are frequently based upon equations and/or parameters that are not commonly measured or analysed in the other relevant disciplines (Van den Bergh, 1996; Perrings, 2000; Westley et al., 2002). Hence, there is a need to further examine how ecological and economic variables can be integrated in ecological-economic models, and to assess the implications of complex ecosystem dynamics for the efficient and sustainable management of ecosystems.

Complex dynamics appear in a wide range of ecosystems that provide a livelihood to hundreds of millions of people (including fishermen and rangeland pastoralists). Hence, understanding the implications of complex dynamics is also of major importance for formulating natural resource management strategies that maintain the income earning opportunities for large numbers of people. The problem definition of this thesis is therefore: 'how can efficient and sustainable management be effected for ecosystems subject to complex dynamics'.

1.2 Objective, research questions and scope

Objective. The objective of the study is:

'to analyse the implications of complex ecosystem dynamics for the efficient and sustainable management of ecosystems'.

The study does not focus on 'natural' ecosystems, but specifically considers ecosystems that are managed and modified by people. The term ecosystem, rather than environmental system, is used because ecosystems are more homogeneous in terms of structure and processes, which makes it a better entry point to study dynamic economy-ecosystem interactions. The study is based upon an ecological-economic modelling approach that is strongly disaggregated, which allows for the integration of the economic system and the ecosystem at the level of economic and ecosystem components. Specific attention is paid to the ecological aspects of ecological-economic modelling, and in particular to the incorporation of ecological complexities in the models.

In order to analyse the implications of complex dynamics for the efficient and sustainable management of ecosystems, I have formulated the following research questions.

Research questions:

1. **How can the efficiency and sustainability of ecosystem management options be analysed ?**

This thesis commences with the development of an ecological-economic framework for the assessment of the economic efficiency and sustainability of management options for ecosystems subject to complex dynamics. The framework allows for a dynamic assessment of the development of the state of the ecosystem, and the flow of ecosystem services, as a function of human management and ecological processes. Subsequently, it is examined how the framework can be translated into an ecological-economic model. Three types of economy-ecosystem interactions are distinguished in the framework, relating to the harvest and use of ecosystem services, pollution and ecosystem interventions. Two critical aspects in the application of the framework are (i) the analysis of the dynamic supply of ecosystem services as a function of human management and ecological processes; and (ii) the valuation of ecosystem services (see e.g. Van den Bergh, 1996, Perrings, 1998; and Turner et al., 2003). These two aspects have been translated into two separate research questions.

2. **How do complex dynamics influence the response of the ecosystem to management measures ?**

Application of the framework requires modelling of the dynamics of the ecosystem as a function of human management and ecological processes. This relates in particular to changes in the state and the resilience of the ecosystem, and to changes in the ecosystem's capacity to supply ecosystem services following the implementation of ecosystem management measures (Levin, 1992; Common and Perrings, 1992;

Perrings, 1998). The response of many ecosystems to human management is strongly influenced by complex dynamics (e.g. Scheffer et al., 2001). This thesis examines the implications of three types of complex dynamics, specifically (i) irreversible ecosystem responses; (ii) multiple states and thresholds; and (iii) stochasticity and lag effects (see table 1.1).

3. How can ecosystem services valuation be applied to analyse ecosystem management options ?

The valuation of ecosystem services has been an important field of research in environmental and ecological economics (Sterner and Van den Bergh, 1998; Deacon et al., 1998). Both the advantages and the limitations of the various valuation methodologies have become increasingly clear (see e.g. Turner et al., 2003), and the further development of valuation methodologies is not among the objectives of this thesis. However, the ecological-economic modelling approach pursued in the thesis requires the valuation of ecosystem services. In order to assess the feasibility of the overall approach, the applicability of ecosystem valuation is examined on the basis of a literature review and a case study in which the economic value of four ecosystem services supplied by the De Wieden wetland is calculated.

4. How can the management of complex ecosystems be optimised, from an efficiency and sustainability perspective ?

Once the interactions between the ecosystem and the economic system have been assessed, and the ecological-economic model is constructed, the efficiency and sustainability of the various management options can be analysed. Efficient ecosystem management options can be identified through a simulation, and/or an algebraic optimisation approach (Chiang, 1992; Van den Bergh, 1996; Grasso, 1998). Both approaches are tested in this thesis. Assessment of the sustainability of management options requires analysis of the long-term development of the ecosystem as a function of human management and ecological processes (WCED, 1987; Pearce et al., 1989). Efficient and sustainable management options are compared, on the basis of the case studies, in the concluding chapter of this thesis.

In order to examine these four research questions, I analyse the dynamics, and the management options, in three ecosystems. First, in chapter 3, I examine a hypothetical hillside forest ecosystem that is driven by wood harvesting and erosion. This study examines the implications of irreversible ecosystem behaviour for the efficient and sustainable management of the forest ecosystem.

Next, the De Wieden wetland in the Netherlands is studied. This wetland supplies four main ecosystem services, and it is subject to complex dynamics. Chapter 4 presents an analysis and valuation of the four ecosystem services supplied by De Wieden. In chapter 5, an ecological-economic model for De Wieden is developed and the most efficient level of eutrophication control in the four main lakes of the wetland is calculated. The model accounts for the presence of two steady states and a threshold in the response of De Wieden to eutrophication control measures.

Subsequently, I examine the "Ferlo" semi-arid rangeland in Northern Senegal. In the Ferlo, livestock keeping is the main source of income for the local population. Livestock depends upon the plant biomass available for grazing, which, in turn, depends upon the rainfall conditions, and the impact of grazing on the plant cover. In chapter 6, I examine the impacts of stochastic rainfall and grazing pressure on the productivity of the rangeland. In chapter 7, an ecological-economic model is developed, and the efficient livestock stocking rate is calculated.

The De Wieden and the Ferlo ecosystems have been selected for the case studies because (i) they provide a number of important ecosystem services; (ii) general models that describe the dynamics of these types of ecosystems are available (e.g. Walker, 1993; Scheffer, 1998); and (iii) for these systems, I was able to obtain long-term data series on their respective dynamics. The main characteristics of the case studies are summarised in table 1.1.

Table 1.1. Case studies of the thesis

Chapter	Study-area	Ecological complexity analysed	Examined management option
3	Hypothetical hillside forest	Irreversibility	Assessment of the efficient and sustainable harvest rates
4, 5	De Wieden wetland (the Netherlands)	Multiple states and thresholds	Assessment of the efficient eutrophication control level
6, 7	The Ferlo rangeland (Senegal)	Stochasticity and lag effects	Assessment of the efficient long-term livestock stocking rate

General approach and scope

This thesis follows a welfare neo-classical approach to the valuation of ecosystem services and the analysis of efficient ecosystem management (e.g. pollution control) levels. Efficient ecosystem management is defined as management that maximises a utility function, with the utility function reflecting the benefits of the ecosystem services and the costs of maintaining and managing the ecosystem (cf. Chiang, 1992). Sustainable ecosystem management is 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 (based upon WCED, 1987 and Barbier and Markandya, 1990). This corresponds to a 'strong' sustainability interpretation (Carter, 2001). The concepts of efficient and sustainability are further examined in chapter 2. The thesis identifies ecosystem management options from the perspective of the social planner, without consideration of the policies that need to be put in place to ensure that individual decision makers act in accordance with the efficient and/or sustainable management approach.

In the thesis, only monetary value indications are used for the valuation of ecosystem services in order to facilitate the comparison of the total net value of the different services under different forms of management (but see e.g. Martinez-Alier et al., 1999 for a discussion on multicriteria evaluation in ecological economics). Mainstream economic approaches are complemented with the consideration of the multiple services that ecosystems supply to society, and of the specific ecological mechanisms that guide the response of ecosystems to human management. In this way, the thesis attempts to enhance the ecological accuracy of ecological-economic modelling.

The study does not consider the equity aspects related to ecosystem management. Hence, optimal as used in this thesis refers only to efficient and sustainable management. Equity aspects are relevant where there are different stakeholder groups in ecosystem management, in particular if these have conflicting interest. This may result from interests in different services supplied by the system (e.g. wood cutting versus nature conservation). Assessing the values of ecosystem services for different stakeholders may form a basis for the consideration of these equity aspects, but a more in-depth analysis of this topic is outside the scope of the thesis. However, *intergenerational* equity is partly accounted for through the analysis of the sustainability of ecosystem management.

The study focuses on the temporal dimensions of ecosystem management – related to its focus on complex dynamics. Hence, there is only limited study of the spatial aspects of ecosystem services. In particular, the thesis contains an assessment of the spatial scales at which the four main ecosystem services of the De Wieden wetland are supplied to the relevant stakeholders. More detailed assessment of the spatial characteristics of ecosystem services requires the use of spatial models in a GIS, and is outside the scope of the current study (see e.g. Low et al., 1999 and Voinov et al., 1999 for more information on this topic). Nevertheless, the basic principles of the modelling approach can also be applied in a spatially explicit manner, through the spatial definition and modelling of the relevant economic and ecological processes.

Uncertainty in ecosystem behaviour has, to a modest extent, been accounted for in this study. In particular, the study contains a brief literature review on the implications of uncertainty for efficient and sustainable ecosystem management (chapter 2). Furthermore, in the De Wieden wetland, the statistical significance of the empirical equations has been tested, and a sensitivity analysis has been conducted for the ecological threshold in the system, which is the main source of uncertainty in the constructed ecological-economic model. For the Ferlo rangeland, stochasticity in rainfall has explicitly been modelled, but a lack of data limited the analysis of uncertainty in the different variables of the model.

1.3 Methodology

The general methodology is briefly discussed below for the four research questions. More detailed descriptions of the applied methodologies can be found in the next chapters of the thesis.

1. How can the efficiency and sustainability of ecosystem management options be analysed ?

Analysis of the efficiency and sustainability of ecosystem management has received ample attention in the last decades, in particular in economics and ecology. Two major types of models that have been widely used in natural resource economics are the Faustmann models - used to determine efficient harvest intervals for forests (Faustmann, 1849; Brazee, 2001); and the Schaefer models – used to determine efficient fish harvest rates (Schaefer, 1957). In systems ecology, there has been much interest in the human impacts on ecosystems, for example through pollution and resource use, since around the late 1960s (Ehrlich et al., 1970; Odum, 1971). Further interest in the sustainability aspects of human ecosystem management was triggered by the report 'The limits to growth' (Meadows et al., 1972). Subsequently, a wide range of environmental economic models have been developed to analyse the potential impacts of human management on ecosystems and the supply of natural resources (for instance Dasgupta and Heal, 1979 and Van Ierland, 1993). In the more recent field of ecological economics, an interdisciplinary approach to economics and ecology is pursued (Costanza, 1989). Analysis of the efficiency and sustainability of ecosystem use are among the main research topics in ecological economics (Costanza and King, 1999).

The thesis starts with a review of relevant literature in the fields of environmental and ecological economics and systems ecology, covering the topics economy-ecosystem interactions, ecosystem dynamics, ecosystem valuation, and ecological-economic modelling. Based on this review, a conceptual framework for the assessment of the efficiency and sustainability of ecosystem management options is developed (chapter 2). Based on the literature review, a dynamic systems modelling approach is selected for this thesis. This essentially comprises linking ecosystems and the economic system at the level of components, accounting for the states and flows of the relevant components (Van Ierland, 1993; Van den Bergh, 1996). It is subsequently explored how this modelling approach can be used to analyse the efficiency and sustainability of ecosystem management options, both conceptually (chapter 3) and in the case studies of De Wieden and the Ferlo (chapters 4 to 7).

2. How do complex dynamics influence the response of the ecosystem to management measures ?

Ecosystem dynamics are determined by ecological processes and human management, which take place over a range of temporal and spatial scales (Clark et al., 1979; Holling et al., 2002). In the last decades, the traditional, Clementsian model of ecosystem dynamics has been complemented with a range of new insights. It is now widely recognised that ecosystems may be subject to various non-linear, complex dynamics, including multiple states, thresholds, irreversibilities, hysteresis, interacting drivers, delayed responses and stochasticity (May, 1973; Scheffer et al., 2000; Holling and Gunderson, 2002). A key element in the modelling of ecosystem dynamics is the identification of the main steering variables and state parameters that can be used to characterise the development of the ecosystem (Holling et al., 2002). Recent applications of this systems approach to ecosystem modelling are available for, among others, hillside forests (West et al., 1981), freshwater lakes (Scheffer, 1998; Carpenter et al., 1999; Mäler, 2000), and rangelands (De Mazancourt et al., 1999; Ludwig et al., 2001). Changes in the state of the ecosystem are generally reflected in the capacity of the ecosystem to supply services (Limburg et al. 2002).

This thesis includes a brief review of ecosystem dynamics in chapter 2, with particular reference to the three types of dynamics studied in this thesis: irreversible responses (chapter 3), thresholds and multiple states (chapter 5), and stochasticity and lag effects (chapters 6 and 7). The study includes three case studies. Irreversible responses are examined for a hypothetical forest system. Thresholds and multiple states are examined for the shallow lakes of the De Wieden wetland in the Netherlands, and stochasticity and lag effects are examined for the Ferlo rangeland in Northern Senegal. The dynamics of forests and shallow lake ecosystems are modelled according to existing models (e.g. West et al., 1981; Hosper, 1997; Scheffer, 1998; Cammeraat et al., 2002). The dynamics of rangelands, in particular the impact of grazing on rangeland systems, are still highly debated in ecological literature (Walker, 1993; Illius and O'Connor, 1999; Sullivan and Rohde, 2002). Therefore, these dynamics are specifically examined in this thesis, based upon long-term grazing data collected in Northern Senegal (as reported in Andre, 1998; Miehe, 1992, 1997).

3. How can ecosystem services valuation be applied to analyse ecosystem management options ?

Ecosystem services have been studied since the early 1970s (Helliwell, 1969; Odum and Odum, 1972; Hueting, 1980; De Groot, 1992; Daily, 1997; Millennium Ecosystem Assessment, 2003). These studies showed that ecosystems provide a wide range of services including (i) the provision of a variety of products such as food and raw materials; (ii) the regulation of climatic, biochemical and biological processes; and (iii) the provision of a suitable environment for cultural, recreational or spiritual activities (Millennium Ecosystem Assessment, 2003). *Valuation* of ecosystem services comprises expressing the effect of a marginal change in ecosystem services provision in a monetary metric (Turner et al., 2003). This allows comparison of the marginal benefits provided by ecosystems to those of other forms of land use. A range of valuation methods have been developed to quantify the different types of economic value that ecosystem services may posses (e.g. Dixon and Hufschmidt, 1986; Pearce and Turner, 1990; Hanley and Spash, 1993; Pearce and Moran 1994; and Willis and Garrod, 1995). Ecosystem services valuation has been conducted for a wide range of ecosystems, see e.g. Costanza et al. (1997a) and Pearce and Pearce (2001).

This study does not aim to further develop existing valuation methodologies, but to test the applicability of ecosystem services valuation in the context of the proposed framework. The study comprises a literature review on valuation (chapter 2) as well as a case study (chapter 4). The case study includes a concise valuation of the four main ecosystem services of the De Wieden wetland: nature conservation, recreation, reed cutting and fisheries. Different methods have been used to express these four services in a monetary proxy, including a travel cost method for the valuation of the recreational service. In the other chapters that examine the value of ecosystem services (chapters 3 and 7), value estimates have been derived from literature.

4. How can the management of complex ecosystems be optimised, from an efficiency and sustainability perspective ?

In terms of *efficiency*, optimisation can be based upon simulation modelling or upon an algebraic or numerical optimisation approach (Chiang, 1992; Van den Bergh, 1996). In the simulation (or programming) approach, the development of the ecosystem is simulated for a range of values of the decision variables. Based on the outcomes of the model, the optimal solutions within the tested range can be identified (e.g. Grasso, 1998). In the algebraic or numerical optimisation approach, optimal solutions are found through the formulation of the Hamiltonian and solving the relevant conditions in an algebraic or numerical manner (Van den Bergh, 1996). These optimisation approaches are 'static' if they provide a single optimal magnitude for every decision variable. Dynamic optimisation involves the identification of optimal magnitudes of decision variables at each point of time in a given time interval (Chiang, 1992). This thesis is confined to static optimisation approaches. Analysis of the *sustainability* of ecosystem management requires analysing the long-term changes in the ecosystem. This relates in particular to the development of selected state indicators for the ecosystem, its resilience, and/or the supply of services by the ecosystem (Barbier and Markandya, 1990; Common and Perrings, 1992; Bossel, 1999).

Both the simulation and the algebraic optimisation approach are applied in the thesis. The simulation approach is pursued for the hillside ecosystem (in chapter 3) and the De Wieden wetland (chapter 5). An algebraic optimisation is conducted for the Sahelian rangeland (chapter 7), whereas the first order conditions have been formulated for the hillside forest ecosystem modelled in chapter 3. In chapter 8, the application of both approaches for the analysis of complex ecosystems is compared. Sustainability is analysed through the modelling of relevant state indicators, in particular for the forest ecosystem. The sustainability of ecosystem management in the De Wieden wetland and the Ferlo has not been explicitly modelled because it was difficult to establish a benchmark condition for sustainable management of these systems, as elaborated in chapter 8. Chapter 8 also evaluates the relevance of the results of this thesis in terms of the reconciliation of the efficiency and the sustainability of ecosystem management.

1.4 Scientific and management relevance of the study

Scientific relevance

Incorporation of ecological complexities and feedback mechanisms into ecological-economic models has been identified as one of the research priorities in the field of environmental and ecological economics (Van den Bergh, 1996; Perrings, 1998; Deacon et al., 1998; Hanley, 1999). This thesis examines how complex ecosystem dynamics can be integrated in ecological-economic models and how they influence the possibilities to manage ecosystems in an efficient and/or sustainable way. Contrary to most existing ecological-economic models (Van den Bergh, 1996; Deacon et al., 1998), a strongly disaggregated approach is followed for the integration of the economic system and ecosystems. In this thesis, the interactions between the economic system and the ecosystem are analysed at the level of ecosystem components, and there is specific attention for non-linear dynamics, stochasticity and ecological feedback mechanisms. This allows examining the management implications of three types of complex dynamics: irreversibility, multiple states and thresholds, and stochasticity and lag effects. The thesis presents a general framework and modelling approach, as well as an application of the framework to one hypothetical, and two real-world ecosystems. For the case studies, it is demonstrated how complex dynamics determine the efficiency and sustainability of ecosystem management options. Application of the approach to real-world ecosystems allows examining how ecological-economic modelling can be pursued in a context where not all data are available, and how concrete management recommendations can be formulated on the basis of disaggregated ecological-economic models of specific ecosystems.

Additional scientific issues that are addressed in this thesis relate to the inclusion of the topsoil component in Faustmann models used to analyse efficient rotation periods in forestry (chapter 3), the spatial scales of ecosystem services (chapter 4), the cost-effectiveness of reductions in nutrient loading versus biomanipulation of shallow lakes (chapter 5), the impacts of grazing on semi-arid rangelands (chapter 6), and the implications of including both stochasticity and dynamic feedbacks in optimisation models (chapter 7).

Relevance for ecosystem management

Environmental decision making frequently requires complex trade-offs, for instance between the short-term depletion of natural resources, and a more sustainable, long-term use of the benefits provided by an ecosystem. Ideally, such decision making is supported by environmental cost-benefit analysis that allows the identification of the optimal ecosystem management strategy, including optimal levels of resource extraction, pollution control, and other interventions in the ecosystem. However, in reality, these optimal strategies are often difficult to define (see e.g. Turner et al., 2003). One of the reasons for this is that the response of the ecosystem to management, and subsequent changes in the supply of ecosystem services, may be hard to predict. In particular, complex ecosystem dynamics can have major implications for the responses of ecosystems to management (Carpenter et al., 1999; Scheffer et al., 2001).

Therefore, accounting for complex dynamics can substantially increase the accuracy of ecological-economic models and environmental cost-benefit analysis (Costanza and Gottlieb, 1998). In particular, it is anticipated that further examination of the role of complex dynamics enhances the possibilities to identify efficient and sustainable ecosystem management strategies with ecological-economic models. This is, of course, of particular relevance to ecosystems subject to complex dynamics, including fish stocks, freshwater lakes, certain woodlands, rangelands, coral reefs and coastal zones (Steele and Henderson, 1984; Timms and Moss, 1984; Dublin et al., 1990; Friedel, 1991; Murray and Parslow, 1999; Nyström et al., 2000; Scheffer et al., 2001).

1.5 Structure of the thesis

The structure of the study is as follows. Chapter 2 provides a literature review, covering the topics economy-ecosystem interactions, ecosystem dynamics, valuation of ecosystem services, and ecological-economic modelling. Based on this literature review, the chapter provides a conceptual framework and modelling approach that can be used to assess the efficiency and sustainability of ecosystem management options. The chapter also examines how the approach can be applied to identify optimal ecosystem management options, in terms of both efficiency and sustainability.

In Chapter 3, the conceptual framework for the assessment of ecosystem management options is tested. The chapter deals with a hypothetical hillside forest ecosystem that provides two ecosystem services: wood supply and erosion control. Two models have been constructed, representing ecosystems that respond in a reversible and an irreversible way to the overharvesting of wood. The models are based upon potentially realistic values for all state and flow parameters, derived from literature. The outcomes provide insights in the implications of pursuing efficient versus sustainable management under reversible and irreversible ecosystem behaviour.

Chapter 4 comprises a monetary valuation of the four most important ecosystem services provided by the De Wieden wetland, the Netherlands. These services are nature conservation, recreation, reed cutting and fisheries. The chapter presents the approximate economic value of the four services, and it is examined at which spatial scales these services are supplied to the economic system. Subsequently, it is assessed how this affects the interests of stakeholders in the management of the ecosystem.

Chapter 5 deals with eutrophication control in the four most important shallow lakes in the De Wieden wetland. An ecological-economic model is constructed that includes the costs of eutrophication control measures, the response of the ecosystem to these measures, and the potential implications for the supply of ecosystem services. The model specifically deals with the presence of two multiple states and a threshold in the state of the lakes. It is examined how the presence of two steady states influences the costs and benefits of eutrophication control measures, and the most efficient eutrophication control strategy for the wetland is identified.

Chapter 6 presents the results of a case study conducted in the Ferlo rangeland, Northern Senegal. There is a long-standing controversy on the impacts of high grazing pressures versus climate variability on rangelands (Sullivan and Rohde, 2002; Briske et al., 2003), and this chapter examines the combined impact of these two drivers on the productivity of the rangeland. The study is based upon existing data resulting from a long-term grazing experiment. It is shown how high grazing pressures influence the species composition and the productivity of the rangeland, and how this affects the capacity of the rangeland to support livestock grazing during a drought.

Chapter 7 examines how, on the basis of an improved understanding of rangeland dynamics, grazing pressures can be optimised. The chapter presents a formal model, and a case study in which this model is applied to the Ferlo rangeland in Northern Senegal. The proposed optimisation model accounts for the complex dynamics of the ecosystem as well as stochasticity in rainfall. In the case study, it is also examined what the implications are of changes in livestock prices during and following droughts.

Chapter 8 presents a discussion and the main conclusions of the thesis. Specifically, the implications of the outcomes of the research for the four research questions are examined. This includes an evaluation of (i) the general applicability of the framework proposed in chapter 2; (ii) the implications of the examined complex dynamics for ecosystem management; (iii) the applicability of ecosystem services valuation in the context of the proposed framework; and (iv) the general insights obtained in optimising the management of ecosystems subject to complex dynamics. The chapter also provides a number of recommendations for ecosystem management, and for further research.

2. A framework for assessing the efficiency and sustainability of ecosystem management options

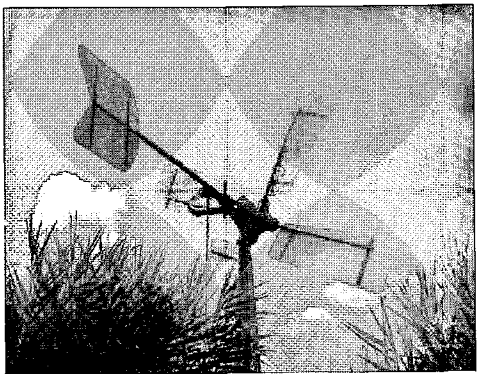


Photo: Mark Grutters

2.1 Introduction

In recent decades, the interactions between the economic system and ecosystems have been studied extensively in environmental and ecological economics (Deacon et al., 1998; Turner et al., 2003). Key interactions include the modification of ecosystems as a function of human management and ecological processes, and the supply of services by the ecosystem. A range of economic models have been developed to analyse the potential impacts of human management on ecosystems and the supply of natural resources (for instance Dasgupta and Heal, 1979; Van Ierland, 1993; and Parks et al., 1998). Increasingly, ecosystem complexities have been considered in these models (e.g. Perrings, 1997; Bulte and Van Kooten, 1999; Carpenter et al., 1999). Nevertheless, as motivated in chapter 1, there is a need to further pursue the incorporation of ecosystem complexities in ecological-economic models (Perrings, 1998; Deacon et al., 1998, Hanley, 1999).

In this chapter, I develop a general, conceptual framework that can be used to assess the economic efficiency and sustainability of management options of complex, dynamic ecosystems. *Efficient* management of ecosystems can be defined as management that maximises a utility function, including the benefits of all services supplied by the ecosystem, and the costs involved in providing or accessing the services (based upon Chiang, 1992). For the purpose of this thesis, *sustainable* ecosystem management is 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 (based upon WCED, 1987; Pearce et al., 1989 and Barbier and Markandya, 1990).

The framework can be used to establish the conceptual outline of an ecological-economic model, and it provides the basis for the studies undertaken in the subsequent chapters of this thesis. It includes the main interactions between the economic system and the ecosystem, and the different types of ecosystem management options. It is dynamic and capable of accounting for complex ecosystem dynamics. The framework is based upon a review of relevant environmental and ecological economics, and ecological literature, covering the following topics (i) ecosystem services; (ii) ecosystem dynamics; (iii) valuation of ecosystem services; and (iv) ecological-economic modelling.

This chapter is organised as follows. Section 2.2 presents the outcomes of the literature review, including the main findings regarding the four topics identified above. Section 2.3 presents the conceptual framework. Section 2.4 demonstrates how the framework can be applied to select the most efficient and/or sustainable ecosystem management options. Finally, section 2.5 presents a brief discussion of the main outcomes of this chapter, and the implications for the remaining part of the thesis.

2.2 Theoretical background

2.2.1 The concept of ecosystem services

Introduction

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, I use 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.

In the early 1970s, the concept of ecosystem *function* was proposed to facilitate the analysis of the benefits that ecosystems provide to society (Bouma and Van der Ploeg, 1975; Van der Maarel and Dauvellier, 1978; 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 are the goods or services provided by the ecosystem to society (Costanza et al., 1997a; Millennium Ecosystem Assessment, 2003). 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).

Types of ecosystem services

In this thesis, I distinguish three different categories of ecosystem services: (i) production services; (ii) regulation services; and (iii) cultural services, based upon De Groot et al. (2002) and Millennium Ecosystem Assessment (2003). These categories are described below, and table 2.1 presents an overview of the ecosystem services in each category.

(i) Production 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 and agricultural systems and, in the calculation of the value of the service, the relevant production and harvest costs have to be considered.

(ii) Regulation services result from the capacity of ecosystems to regulate climate, hydrological and bio-chemical cycles, earth surface processes, and a variety of biological processes. These services often have an important spatial aspect; e.g. the flood control service of an upper watershed forest is only relevant in the flood zone downstream of the forest. Contrary to De Groot et al. (2002), but in line with De Groot (1992), 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.

(iii) Cultural services relate to the 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). These services have also been named 'information services' (De Groot et al., 2002).

Contrary to Millennium Ecosystem Assessment (2003), I do not distinguish the category 'supporting services', which represents the ecological processes that underlie the functioning of the ecosystem. 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 ecosystems, and it is unclear on which basis supporting services should be included in, or excluded from a valuation study.

Table 2.1. List of ecosystem services (based on Van der Maarel and Dauvellier, 1978; Ehrlich and Ehrlich, 1981; Costanza et al., 1997a; De Groot et al., 2002; Millennium Ecosystem Assessment, 2003).

Category	Examples of goods and services provided
Production 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
Regulation 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	- 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

2.2.2 Ecosystem dynamics

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 consists 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 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 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: (i) exploitation (colonisation of disturbed areas); (ii) conservation (slow accumulation and storage of material and energy); (iii) release (creative destruction); and (iv) 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 dying of a substantial proportion of the trees. This is followed by the dying of the budworms and the regrowth of the trees, and the cycle repeats itself (Ludwig et al., 1978; Clark et al., 1979).

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 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 ("keystone species") are responsible for the maintenance of the functioning of the ecosystem and that these species can take over each others' 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).

Complexities in ecosystem dynamics

Characteristic of the new theories for 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. In this thesis, the following types of complex dynamics are considered: (i) irreversibilities; (ii) multiple states and thresholds; and (iii) stochasticity and lag-effects.

(i) 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, or can be constrained by the amount of time needed to regain the system, as may be the case with 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, 2001). 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 location of the threshold, or the overall rate of change of the system following a disturbance (e.g. Scheffer and Carpenter, 2003).

(ii) 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 (e.g. Scheffer, 1998) or species deletion 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 (Scheffer et al., 2002). Often, the shift between multiple states occurs suddenly and comprises the existence of threshold effects (Wissel, 1994; 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 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 allowing the phytoplankton to grow to even higher concentrations. 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).

(iii) 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. For instance, in rangelands, the impact of soil degradation resulting in reduced seedling establishment may become apparent only after a fire (Friedel, 1991).

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 (e.g. Costanza et al., 1993).

2.2.3 Valuation of ecosystem services

The theoretical context of valuation

As mentioned in Chapter 1, this thesis takes a neo-classical welfare economic approach to the valuation of ecosystem services. According to welfare economics, the welfare generated by an ecosystem service, or the economic value of this service, is the (weighted) sum of the utility gained by all individuals as a result of the provision of the ecosystem service. Utility is gained by the person consuming the ecosystem service (e.g. by eating a piece of fruit or walking in a national park). Utility may also be gained, or lost, by the person or institute offering the ecosystem service (the person collecting and selling the fruit, or the ecosystem manager maintaining the recreational facilities of a park). For private ecosystem services, and assuming perfect market conditions, price reflects the marginal economic value of the service. Two central concepts in understanding the utility that consumers and producers gain from a transaction are the consumer and the producer surplus. These two central concepts are briefly described below.

(i) 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). Hence, the market price plus the consumer surplus equals the utility of a specific good for a certain consumer. Note that the utility gained by the consumer from an actual transaction also depends upon a number of other factors, such as transaction costs. Estimation of the consumer surplus generally requires the construction of a demand curve (see e.g. Perman et al., 1999). Hicks (1941) found an inconsistency in the ordinary, or Marshallian consumer's 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's 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: (i) if the income elasticity of demand for the good in question is low; and (ii) 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, and may be particularly complicated (Johanson, 1999).

(ii) The producer surplus. The producer surplus indicates the amount of welfare a producer gains at a certain production level and at certain price. The estimation of the producer surplus generally requires the construction of a supply curve (see e.g. Perman et al., 1999). In the short term, a producer's fixed costs can be considered foregone. Hence, in micro-economics, the individual producer surplus is defined as total revenues minus variable costs (Varian, 1993). In the valuation of ecosystem services, the producer's surplus needs to be considered if there are costs related to "producing" the ecosystem good or service (Freeman, 1993; Hueting et al., 1998). In general, in the case of private ecosystem good or services, these costs relate to the costs of harvesting or producing the ecosystem good or service (Hueting et al., 1998). For public ecosystem goods, supply curves reflect 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 producers surplus is difficult to establish (Hueting et al., 1998). The supply curve will in many cases show a relatively steep increase at higher quantities of ecosystem service supplied – e.g. the costs of providing marginal cleaner water increase as purity becomes higher (Hueting, 1980).

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; and Millennium Ecosystem Assessment, 2003). Based upon Pearce and Turner (1990) and Millennium Ecosystem Assessment (2003), I distinguish the following four types: (i) direct use value; (ii) indirect use value; (iii) option value; and (iv) non-use value.

(i) 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 production services, and some cultural services (such as recreation) have direct use value.

(ii) Indirect use value stems from the indirect utilization of ecosystems, in particular through the positive externalities that ecosystems provide (Munasinghe and Schwab, 1993). This reflects the type of benefits that regulation services provide to society.

(iii) 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).

(iv) None-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 recognize 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.

These four value types all need to be considered in the assessment of the total value of the services supplied by an ecosystem. In principle, the values are additive (Pearce and Turner, 1990). Insofar as commensurable value indicators have been used, they may be summed in order to obtain the total value of the services supplied by the ecosystem.

Valuation methods

Following neo-classical welfare economics, valuation requires analysis and aggregation of the consumer and producer surpluses (Freeman, 1993). In the last 3 decades, a range of economic valuation methods for ecosystem services has been developed. They differ for private and public goods.

(i) 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 practise this is often difficult, as (i) it is not always known how people will respond to large increases or decreases in the price of the good, and (ii) it may be difficult to assess when consumers will start looking for substitute goods or services. In case of substantial 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 case the private good is not traded in the market, because it is bartered or used for auto-consumption, shadow prices need to be constructed on the basis of: (i) the costs of substitutes; or (ii) the derived benefit of the good (Munasinghe and Schwab, 1993).

(ii) Valuation of public goods. For public goods or services, the marginal willingness to pay can not be estimated from 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 indirect and the direct approach (Pearce and Turner, 1990). Pearce and Howarth (2000) call them revealed and expressed preference methods, respectively. The *indirect* 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 indirect 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 place. 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 the making of 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: (i) defensive expenditure (a water filter); (ii) the purchase of environmental surrogates (bottled water); and (iii) relocation (OECD, 1995; Pearce and Howarth, 2000). The travel cost method and the hedonic pricing method are other indirect approaches using behavioural linkages (Van Kooten and Bulte, 2000).

With <u>direct</u> approaches, various types of questionnaires are used to reveal the willingness-to-pay of consumers for a certain ecosystem service. The most important direct approaches are the Contingent Valuation Method (CVM) and related methods. In the last decades, CVM studies have been widely applied (see e.g. 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 U.S. - 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 overestimates economic values because respondents do not actually have to pay the amount they express to be willing to pay for a service (see e.g. Diamond and Hausman, 1994; Cummings and Harrison, 1995; Hanemann, 1995 and Carson, 1998).

In response to the difficulties encountered in quantifying the non-use values of ecosystems in monetary terms, some authors have proposed to quantify this value in *ecological* terms only (e.g. Strijker et al., 2000; Costanza and Folke, 1997). A large number of ecological valuation methods have been developed; Wathern et al. (1986) mention that over 100 of these techniques have been described in 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 (Margules and Usher, 1981). Comparison of the ecological and monetary value indicators can be left up to the person reading the valuation study (as in Strijker et al., 2000), or can be done through Multi Criteria Analysis (Nijkamp and Spronk, 1979; Costanza and Folke, 1997).

An overview of the different valuation methods, and the value types they can be used for, is presented in table 2.2. 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. (1997a) and Pearce and Pearce (2001) provide indications 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; and Brouwer et al., 1997).

Table 2.2. Valuation methods and their applicability to different value types (Based upon Pearce and Turner, 1990; Hanley and Spash, 1993; Munasinghe, 1993; Cummings and Harrison, 1995).

Valuation method	**Suitable for**	**Value category**			
		direct use value	indirect use value	option value	non-use value
Economic valuation methods					
Direct methods:					
a) market valuation	Ecosystem goods and services traded on the market	x	x	x	
b) CVM	The use of CVM is limited to goods and services that are easily to comprehend for respondents – excluding most regulation services	x		x	x
Indirect methods:					
a) hedonic pricing	Applicable where environmental amenities are reflected in the prices of specific goods, in particular property.	x		x	
b) travel cost method	Can be used to value the recreation service.	x			
c) averting behaviour method	Applicable to services that relate to the purification services of some ecosystems.	x	x		
d) damage function approach	Applicable where loss of ecosystem services will cause economic damage, e.g. through an increased flood risk.		x	x	
Ecological valuation methods					
a) ecological valuation	Only for the part of the existence value related to the nature conservation service				x

2.2.4 Ecological-economic modelling

Introduction

The linking of economic and bio-physical variables has, in ecological-economic modelling, been pursued mainly through the integration of economics with two main other disciplines: thermodynamics and ecology (Van den Bergh, 1996; Costanza et al., 1997b). With respect to thermodynamics, various studies have incorporated the First Law and the Second Law of thermodynamics into ecological-economic models (e.g. Georgescu-Roegen, 1979). The First Law states that energy and matter can be transformed, but not created or destroyed; the Second Law that entropy, in a closed system, is increasing. In integrating economics with insights from thermodynamics, a seminal paper was 'The economics of the coming spaceship Earth', which stated that Earth should be considered as an almost closed system, which puts constraints on its capacity to accommodate economic activity (Boulding, 1966).

For ecological-economic assessments or modelling studies conducted at the scale of the ecosystem, the consideration of ecological insights in the studies is more relevant. Therefore, in line with the focus of this study, the implications of thermodynamics are not further analysed here. This section provides a brief review of four model types used for the integrated modelling of economy-ecosystem interactions, followed by an analysis of how ecological complexities can be included in these models.

Brief review of selected model types

Ecological-economic models can be used to maximise (or minimise) some function of one or more decision variables of an ecosystem in its economic context. 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 thesis, only integrated models are considered. There are a number of integrated approaches to ecological-economic modelling, including (i) generalised input-output models; (ii) neo-classical growth models; (iii) dynamic system models; and (iv) 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.

(i) 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 utility and 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). In other words, 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.

(ii) 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 e.g. Barro and Sala-i-Martin, 1995). Integration of environmental issues is comparable to the integration of environmental issues in general equilibrium models.

(iii) 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 (adapted from Van den Bergh, 1996). This may comprise a range of theoretical, statistical or methodological constructs, dependent upon the requirements and limitations 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). Hence, it is potentially more suitable to examine the implications of ecosystem complexities for the efficient management of ecosystems. In the context of dynamic systems models, two approaches can be followed to determine the value of the decision variables that provides maximum utility: (i) a simulation (programming) approach and; (ii) 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. In the algebraic optimisation approach, optimal solutions are found in a numerical or algebraic manner, through the preparation of the Hamiltonian and solving the first and second order conditions (Chiang, 1992), as further discussed below.

(iv) 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. They are often built 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 e.g. 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. Nevertheless, in comparison, less attention has been paid to developing methodologies for analysing the economic implications of spatial variations in the supply of ecosystem services (Deacon et al., 1998). In addition, data requirements are high for a spatially explicit approach. Initial conditions, processes, and implications of decision variables need to specified for each distinguished spatial unit. Both the lack of available methodologies for assessing the spatial aspects of resource use and values, and the high data requirements act as constraints to the application of spatially explicit optimisation models.

Hence, review of the four potentially suitable model types indicates that input-output and neo-classical growth models tend to be applied at relatively high aggregation levels, which constrains their applicability for the integrated ecological-economic modelling of ecosystem dynamics. Dynamic systems models, on the other hand, offer better possibilities to model economy-ecosystem interactions at a lower aggregation level, for example at the scale of ecosystem components, and to include various types of dynamics and interactions in connecting the different state and flow variables. Spatially explicit models may also be applied at low aggregation levels, but their application is constrained by high data requirements and limited availability of methodologies to assess the spatial aspects of resource use and values.

Dealing with complexities in ecological-economic models

The initial ecological-economic models (also called bio-economic models) were based on the assumption that the harvesting of renewable resources is constrained by the population dynamics 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. A well-known example of this type of models are the fisheries models constructed by Clarke (e.g. Clark, 1976). 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:

$$dS/dt = f(s_t, S_t)$$

The growth function of the stock may take the form of a logistic growth curve, or a set of Lotka-Volterra equations. A main variable determining the growth of the stock is the size of the stock, reflecting that the reproduction rate of a species generally depends upon the numbers of the species present in relation to the carrying capacity of its environment (e.g. Pielou, 1969). Importantly, this 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.

In its simplest form, one variable may suffice to represent the ecosystem state, but more sophisticated models are likely to require a set of, interconnected, state variables that represent different components of the ecological-economic system (Van den Bergh, 1996). This also enhances the possibility to include the various ecosystem complexities in the models such as (i) irreversibility; (ii) multiple states and thresholds; and (iii) stochasticity. These three aspects are briefly discussed below.

(i) Modelling irreversibility. 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. This may be the case where, for instance, sustained, high pollution loads lead to local extinction of particular species (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 model for dealing with irreversibilities and derive that maintenance of the resilience of ecosystems is an important factor for economic efficient management.

(ii) Modelling multiple states and thresholds. The presence of multiple states and thresholds is dependent upon feedback mechanisms in the ecosystem. Ecosystem states may be stable or unstable configurations, involving predominantly negative or positive feedback mechanisms, respectively. This is illustrated in figure 2.1, where the valleys represent stable states, and the top of the hill in the top two profiles represents an unsteady stable state. In general, systems with alternative equilibriums are sensitive to the initial condition of the ecosystem, which is referred to as the 'path-dependency' of such systems (e.g. Scheffer et al., 2001). A sufficiently severe perturbation (e.g. fire eliminating part of the biomass) can induce a shift to another stable state. This is expressed in figure 2.1 by a movement of the ball from the left to the right-hand valley. Hence, ecological-economic modelling of multiple states and thresholds requires definition of the initial conditions of the ecosystem, modelling of the ecological processes acting as positive or negative feedback, modelling of the impact of decision variables and possible exogenous disturbances on the system, and quantification of the economic implications of changes in ecosystem state (Perrings and Pearce, 1994; Carpenter et al., 1999; Wu and Skelton-Growth, 2002; and Mäler et al., 2003).

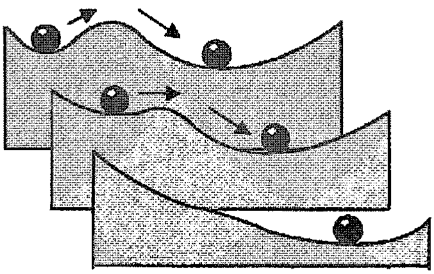


Figure 2.1 Ball and cup heuristic of system stability. Valleys represent stability domains, balls represent the system, and arrows represent disturbances. Resilience is determined by both the slopes and the widths in the stability landscapes. Adapted from Scheffer et al. (1993) and Gunderson (2000).

(iii) Modelling stochasticity. 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.

2.3 Framework for assessing the efficiency and sustainability of ecosystem management

2.3.1 Interactions between the economic system and ecosystems

Three main categories of interactions between the economic system and ecosystems can be distinguished (De Groot, 1992; Van Ierland, 1993; and WRI, 2000). These are: (i) the supply of ecosystem services; (ii) pressures exerted on the ecosystem through pollution; and (iii) direct interventions in the ecosystem – comprising ecosystem management measures, such as reforestation and spatial zoning, as well as physical disturbance of the ecosystem, for instance through land use conversion. These three interactions are presented in figure 2.2. A fourth type of interaction is not further considered in this thesis: the influence of ecosystems on the economic system through environmental hazards and disasters (De Groot, 1992). The reason is that this thesis focuses on the assessment of specific ecosystem management options. Nevertheless, this type of interaction is partly included, i.e. to the extent that regulation services offset particular environmental hazards (e.g. the regulation service 'control of landslides' reduces the risk of landslides). The characteristics of the three types of interactions are described below.

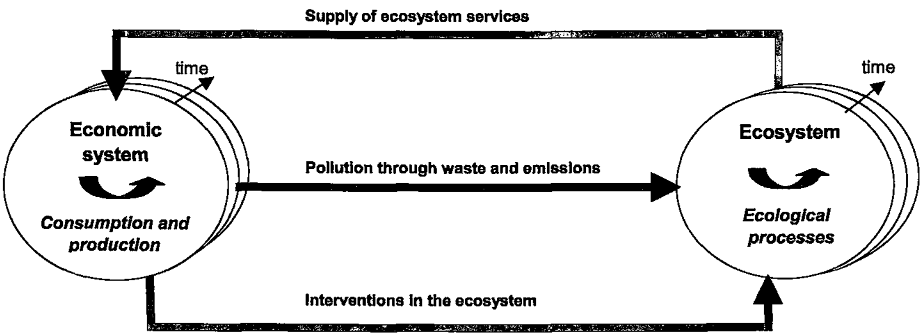


Figure 2.2. Interactions between the economic system and ecosystems. The arrows indicate the direction of the physical flows, or the flows of the services, involved. All interactions have an impact on both the economic system and the ecosystem.

(i) The supply of ecosystem services. The use of ecosystem goods and services is a function of both the developments in the economic system (the demand for ecosystem services) and the ecosystem (the capacity of the ecosystem to supply services). Ecosystem services may be 'consumed' directly, or may be used as input factor in the production process (e.g. Van Ierland, 1993). As described above, three basic types of ecosystem services can be distinguished: production, regulation and cultural services. These service types have a different impact on the ecosystem. Production services are extractive, and their impact depends upon the harvest level in relation to the carrying capacity of the ecosystem. Regulation 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 of the use of this service depends upon the numbers of visitors and their activities in relation to the carrying capacity of the ecosystem for these activities.

(ii) Pollution. Production and consumption processes may release emissions and waste in the environment. The possibility to dispose unwanted substances provides a benefit for the economic system but may have an adverse impact on the functioning of the ecosystem as well as the supply of ecosystem services. The pollution experienced by a particular ecosystem depends upon the total emissions and amounts of waste released, 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 (Van Ierland, 1993; RIVM/UNEP, 1997).

(iii) Ecosystem interventions. This comprises a broad category containing the direct human interventions in the ecosystem. It includes both measures with a positive impact on the ecosystem, such as reforestation, as well as human disturbance of the ecosystem, for instance through the construction of a road. 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 of ecosystems includes physical interventions in the ecosystem that do not involve the extraction of resources or the release of pollutants, such as land use conversion and infrastructure construction. Such activities have adverse impacts on (nearby) ecosystems and can reduce the ecosystem's supply of services.

Commonly, the three interactions are expressed in monetary terms when their impact on the economy is studied, and biological or physical units when their interactions with the ecosystem are studied. Figure 2.2 neglects the spatial aspects of the interactions. All interactions can take place at varying spatial scale levels, depending on the specific service, type of pollution, or intervention involved.

Note that figure 2.2 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. All three interactions can result in a *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 supply of ecosystem services (e.g. reduction of the hydrological service of a forest as a result of timber harvesting). 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 (RIVM/UNEP, 2000).

2.3.2 Development of the conceptual framework

The framework

Building upon figure 2.2, a framework is developed that can be used to assess the efficiency and sustainability of ecosystem management options, while accounting for the complex dynamics of ecosystems. The framework is presented in figure 2.3. It includes three types of management measures, related to the use or harvest of ecosystem services, the control of pollution levels, and direct interventions in the ecosystem. Each measure corresponds to a decision variable. These measures, in combination with the relevant 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 of the ecosystem services, the benefits stemming from the release of pollutants, and the costs of management and intervention measures.

The framework can be used as the conceptual outline of an ecological-economic model or assessment. Ecological-economic models developed according to the framework can be used to assess the efficiency and sustainability of the current management of an ecosystem, as well as to identify more efficient and/or sustainable management options. The optimisation of ecosystem management is further analysed in the next section of this chapter.

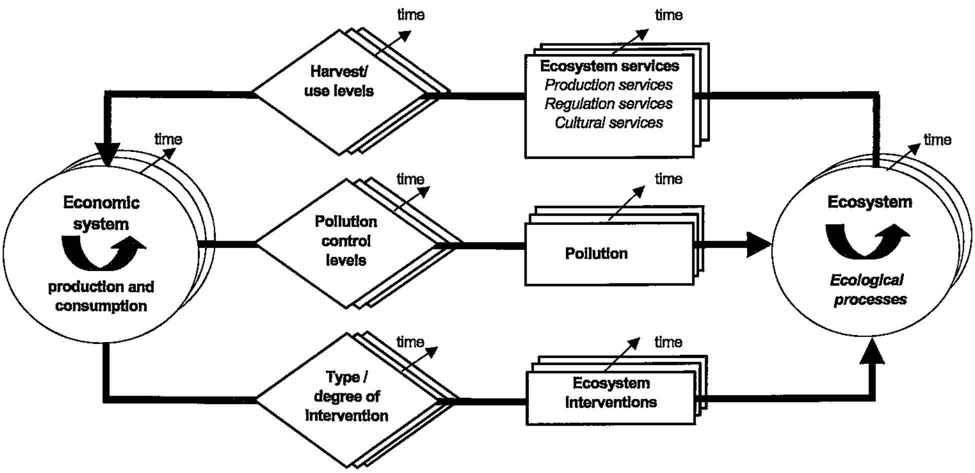


Figure 2.3. Conceptual framework for the ecological-economic modelling of the efficiency and sustainability of management options for a dynamic ecosystem. The circles represent the ecosystem respectively the economic system, the square boxes are labels for the flows between the two systems, and the diamonds represent the decision variables.

The framework is particularly suitable to assess the *temporal* scales related to ecosystem management, in line with the focus of the thesis on complex ecosystem dynamics. The framework, as presented in figure 2.3, does not distinguish between different *spatial* scales of ecosystem management. However, the framework can be adjusted in order to become more spatially explicit by defining the relevant interactions at different spatial scales. In line with the focus of this thesis on complexities and temporal scales, this is not further explored.

Application of the framework involves four main steps: (i) identification of the relevant economy-ecosystem interactions and the potential management options; (ii) modelling the dynamics of the ecological-economic system; (iii) analysis of the costs of management options and valuation of the ecosystem services; and (iv) analysis of the economic efficiency and sustainability of management options. These four steps are described below.

(i) Identification of the relevant economy-ecosystem interactions and the potential management options. This step first involves the definition of the ecological-economic system to be studied, in terms of its spatial and temporal boundaries. Next, it involves the identification of the interactions between the ecosystem and the economic system, including the current management of the system and the various services supplied by the ecosystem. Subsequently, the potential options to enhance the management should be identified. This may include, for instance, changing the harvest levels of ecosystem services or the application of pollution control technologies. Together, these three elements determine the outline of the ecological-economic model to be constructed in steps 2 and 3.

(ii) Modelling the dynamics of the ecological-economic system. This steps involves the modelling, in physical terms, of the impacts of management options on the ecosystem, and the impact of changes in ecosystem state on the system's capacity to supply ecosystem services. 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 accurately represent the (complex) dynamics of the system. A 'dynamic systems' approach to ecological-economic modelling offers the best approach to construct a model at the required aggregation level (see section 2.2.4).

(iii) Analysis of the costs of management options and valuation of the ecosystem services. In this 3rd step, the physical flows need to be expressed in a monetary measure. This involves both examining the costs of the management options, for example through the establishment of a pollution abatement cost curve, and the valuation of changes in the supply of ecosystem services following changes in management. Appropriate valuation methods differ per type of ecosystem service, and per value type. Valuation of non-use values is particularly complex, as elaborated in section 2.2.3.

(iv) Analysis of the economic efficiency and sustainability 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 utility given a certain utility 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).

Hence, the framework provides the conceptual lay-out of an ecological-economic model that can be used to assess the efficiency and sustainability of ecosystem management options. Below, the potential sources of uncertainty in the application of the framework are analysed. Subsequently, in the next section, it is examined how the model can be used for the identification of efficient and/or sustainable management options.

Potential sources of uncertainty in the application of the framework

Translation of the framework in an ecological-economic model requires the modelling of a dynamic set of economy-ecosystem relations. Such modelling may be subject to significant levels of uncertainty. The main sources of uncertainty stem from (i) the input data used to describe the system; (ii) the equations and the structure of the model; and (iii) the valuation of the ecosystem services. 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 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). 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. The principal approach developed to deal with uncertainty in model structure is model validation. Toth (1995) proposed three routes for model validation: (i) check against historical records; (ii) adopt models and codes from other modelling groups for conceptual verification; and (iii) model inter-comparisons (see Toth, 1995 or Van der Sluijs, 1997 for more information). Regarding *valuation*, uncertainty relates to both the application of the valuation methods as such, and to uncertainty people hold regarding their future preferences. The amount of uncertainty in valuation methodologies depends upon the valuation method used. In market valuation approaches, a major issue is the shape of the demand and supply curves for ecosystem services (Hueting, 1980). In revealed preference methods, uncertainty may be involved in the linking of the service to be values to an indicator with a market value (Pearce and Turner, 1990), whereas CVM methods face uncertainty regarding the interpretation of the respondents' stated willingness-to-pay for ecosystem services. Regarding future preferences, the principal way of including uncertainty in valuation studies is to introduce the concept of 'option value', see section 2.2.

2.4 Analysing the efficiency and sustainability of ecosystem management

2.4.1 Introduction

This section examines how the economic efficiency and sustainability of ecosystem management can be analysed. First, a brief review of the efficiency aspects of ecosystem management is presented. This comprises a review of the conditions required for a static, respectively an intertemporally efficient allocation of natural and man-made capital. It is also analysed how ecosystem management can be optimised in terms of efficient resource extraction, efficient pollution control, and the efficient implementation of ecosystem interventions, and in terms of a combination of these three options. Second, the different interpretations of sustainability are reviewed, and it is examined under which conditions ecosystem management can be considered sustainable. Third, a brief analysis of the potential discrepancy between efficient and sustainable ecosystem management is presented. In line with the focus of the thesis, this section only considers efficiency and sustainability aspects of the management of renewable resources.

2.4.2 Efficient ecosystem management

Introduction

Following the well-known Pareto 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 (see e.g. Freeman, 1993). Both Kaldor and Hicks developed an alternative 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 (Freeman, 1993). Following the three criteria mentioned above, there are, in general, a range of allocations of sets of production factors that satisfy these criteria and that can be called efficient. Note that the socially optimal allocation of resources depends upon the social welfare function that specifies how the welfare of different individuals in society can be compared[1].

[1] As explained in section 1.2, the thesis does not address equity issues, and this topic is not further explored here. Where relevant, it is assumed in this thesis that the utility gained from the supply of ecosystem services has equal weights for all individuals involved.

In the case of ecosystem management, the manager is often confronted with *intertemporal* allocation questions, for instance, should a particular resource be harvested now or at some moment in the future. The formulation of an 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 (Perman et al., 1999). Howarth and Norgaard (1990) showed that effects of initial allocations on equity and efficiency readily translate from a static to an intergenerational context. It follows that, also in the intertemporal case, there are many efficient allocations. An intergenerational social welfare function is required in order to specify socially optimum allocations. Following standard neo-classical approaches, future and present utilities can be compared through the use of the utility discount rate Freeman (1993). Another important factor in the analysis of intertemporal efficiency is technological progress. Technological progress may lead to a more efficient use of natural resources, allowing, under a number of conditions, to maintain utility levels even at a decreasing natural capital stock (e.g. Dasgupta, 1993). However, in line with the focus of the thesis on complex ecosystem dynamics, the implications of technological progress are not further examined.

Following the general theoretical concepts briefly described above, the efficiency of different types of ecosystem management can be examined. According to the framework presented in figure 2.3, there are three principal types of ecosystem management: (i) changing the use level of ecosystem services; (ii) the control of pollution influxes; and (iii) direct interventions in the ecosystem. Below, it is analysed how the efficiency of these three types of measures can be assessed, and which conditions need to be met to achieve efficient management. Subsequently, it is briefly discussed how the management of ecosystems subject to more than one type of interaction can be optimised.

Efficient extraction of renewable resources

Efficient resource extraction has been studied since 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. This section further focuses on fisheries models, with forestry type of models ('Faustmann models') further analysed in Chapter 3.

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 (e.g. Tietenberg, 2000). 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).

For an ecosystem where harvest costs increase with decreasing stocks, a mathematical approach can be used to calculate efficient harvest levels as a function of the costs and benefits of harvesting the resource. The objective function can be expressed as:

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

Where $B(s_t)$ are the benefits and $C(s_t, S_t)$ are the harvesting costs. The benefits depend upon the harvest level s_t only, whereas the harvest costs also depend upon the size of the stock S_t. The discount factor is e^{-rt}; a positive discount rate is assumed. Let $\theta(S)$ be the natural growth of the population, β the constant growth factor, and S_{max} the carrying capacity. In case of a simple logistic growth model, the objective function is subject to:

$dS/dt = \theta(S) - s_t$; and

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

Following dynamic control theory, the objective function can be maximised using the Hamiltonian (see e.g. Chiang, 1992 and Perman et al., 1999). The current value Hamiltonian, noted as H, can be expressed as:

$$H_t = B(s_t) - C(s_t, S_t) + \lambda \cdot (\theta(S) - s_t)$$

with λ being the time-dependent co-state variable, interpretable as the shadow price of one unit of stock of the resource. The necessary conditions for a maximum are:

(i) $\partial Ht/\partial s_t = 0 = dB/ds_t - \partial C/\partial s_t - \lambda$

(ii) $d\lambda/dt = r \cdot \lambda - \lambda \cdot d\theta/dS_t + \partial C/\partial S_t$

(iii) $P(t_0) = P_0$ *(initial condition)*

(iv) $H(t_T) = 0$ *(transversality condition)*

The first condition essentially shows that at the point of maximum utility 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 this 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.

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 e.g. 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 2.4. 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).

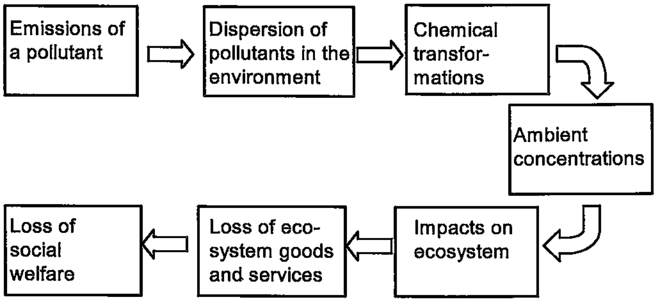


Figure 2.4. Schematic overview of a marginal damage function (adapted from Farmer et al., 2001)

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

$$J = \int_{0}^{\infty} e^{-rt} \{B(P_t) - C(p_t)\}\, dt$$

Where the utility is 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). 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 e.g. Carpenter et al., 1999):

$$dP/dt \equiv P'_t = p_t - \phi P_t + f(P_t)$$

The segment ϕP_t expresses the removal of pollutants from the system. ϕ is determined by the decomposition and loss rates of the pollutant. Decomposition rates will be small for a persistent, inorganic pollutant, and can be much higher for organic pollutants. The loss rate indicates how much of the pollutant is transported out of the ecosystem, for instance through the flushing of the water in a lake. Unless the decomposition rate of the pollutant is affected by the state of the ecosystem, a linear function may be adequate to model the removal (see e.g. Carpenter et al., 1999). The function $f(P_t)$ indicates the behaviour of the pollutant in the ecosystem. This may be a rather complex function, for instance in case the pollutant is buffered in a particular component of the ecosystem, such as lake sediments. Note that 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).

The objective function can be maximised using the current value Hamiltonian (see Chiang, 1992). The Hamiltonian, noted as H, can be expressed as:

$$Ht = B(P_t) - C(p_t) + \lambda(p_t - \phi P_t + f(P_t))$$

with λ being the time dependent co-state variable that can be interpreted as the shadow price of one unit of pollution. The necessary conditions for a maximum are:

$$\text{(i) } \partial Ht/\partial pt = 0 = B \cdot \partial P_t/\partial p_t - dC/dp_t + \lambda - \lambda \cdot \phi \cdot \partial P_t/\partial p_t$$
$$\text{(ii) } d\lambda/dt = r \cdot \lambda - \partial L/\partial p_t$$
$$\text{(iii) } P(t_0) = P_0 \text{ (initial condition)}$$
$$\text{(iv) } H(t_T) = 0 \text{ (transversality condition)}$$

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).

Efficient levels of intervention in the ecosystem

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, or not. The 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 (Hanley and Spash, 1993).

In case 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 (see e.g. Hanley and Spash, 1993). A loss of environmental quality may cause a loss of ecosystem services, and bring costs for compensation payments to stakeholders impacted by that loss (Hueting, 1980). For concave benefit and convex cost functions, the marginal benefits of implementing the measure equal the marginal costs of adverse environmental quality at the point of maximum efficiency (Tietenberg, 2000).

Assessment of the efficiency of ecosystem management under multiple decision variables

Above, the efficiency conditions for three types of ecosystem management options have been examined in isolation. However, for many ecosystems, management needs to consider the *combined* implementation of a number of different ecosystem management options. For instance, the manager may need to manage harvest levels of ecosystem services and the inflow of pollution, at the same time. In this case, the analysis of the efficiency of ecosystem management options needs to consider multiple decision variables. These variables need to be included in the ecological-economic model, and incorporated in the efficiency conditions. The interaction between the decision variables is determined by the structure and processes of the modelled ecosystem. For instance, pollution may affect particular ecosystem components, which may lead to a reduction in the supply of ecosystem services. Depending upon the interactions involved, the objective function of a model guided by multiple decision variables may be highly complex (e.g. Azadivar, 1999). It may incorporate the relevant elements of the objective functions of the single decision variable cases specified in the previous paragraphs. In the case of multiple decision variables, there may be more than one efficient solution to the optimisation problem (Braat and Van Lierop, 1987; Van den Bergh, 1996; Azadivar, 1999).

2.4.3 Sustainability of ecosystem management

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 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 ?

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, and it is still under discussion. 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: (i) harvest rates of renewable resources (e.g., fish, trees) not exceed regeneration rates; (ii) use rates of non-renewable resources (e.g., coal, gas, oil) not exceed rates of development of renewable substitutes; and (iii) rates of pollution not exceed the assimilative capacities of the environment.

On the other hand, several other authors strongly criticise such strong interpretations of the concept of sustainability. For instance, Beckerman (1994) assumes 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) argues 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. He also states that raw material deposits are sufficient to cover many more centuries of consumption, with hydrocarbons being the least widely available with reserves of 'only' several hundreds of years. 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).

Other authors have taken a more intermediate position, proposing 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; and 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 human-made and natural capital is allowed.

Based upon the assumed rates of substitutability, Carter (2001) classifies the different definitions of sustainability into four main categories: (i) very weak; (ii) weak; (iii) strong; and (iv) 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-cut 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. In this perspective, sustainable management is interpreted as management that maintains the resilience of ecosystems (Common and Perrings, 1992; Levin et al., 1998).

In this thesis, following the Brundtland definition, sustainable management of ecosystems 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'. Technical progress is not further considered in this thesis, and it is assumed that this definition implies the maintenance of the full set of natural capital, and its capacity to supply ecosystem services, over time. This interpretation corresponds to the strong sustainability criterion (following Carter, 2001). Strong sustainability poses more restrictions on ecosystem management than weak sustainability, and, hence, the selection of the strong sustainability criterion allows for a more pronounced comparison of the implications of pursuing efficient or sustainable ecosystem management options. Assessing the sustainability will normally involve the modelling of 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.

2.4.4 Efficient versus sustainable ecosystem management

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 lead to an 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 efficiency and sustainability of ecosystem management is the discount rate used to compare present and future flows of benefits derived from the ecosystem. The discount rate can be derived following two approaches, on the basis of (i) the consumption discount rate; and (ii) the social opportunity costs of capital (Pearce and Turner, 1990).

(i) the consumption discount rate. Present and future flows of benefits can be compared on the basis of the consumption discount rate. 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):

$$r = \eta \cdot c + \rho$$

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 ($\eta>0$). 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 it concerns 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; there is a preference for immediate rather than future consumption. However, the ethical basis for this assumption has been questioned (Howarth and Norgaard. 1990). For instance, Arrow et al. (1996) argue that there is no ethical basis to give less weight to the utility of future generations as compared to the current generation.

(ii) the social opportunity costs of capital. The social opportunity costs of capital represent the rate of return on capital. The basic idea behind this approach is that capital investment is productive and that, therefore, capital availability at present is more valuable than the availability of an equal amount of capital in the future. Following this approach, the returns that society could obtain from investment represents the discount factor that should be used in CBA (Common, 1988; Pearce and Turner, 1990).

In a simple economy with no taxes or inflation, and perfect capital markets, the consumption discount rate equals the social opportunity costs of capital equals the market interest rate (Lind, 1982; Varian, 1993). In reality, this is usually not the case. In particular, due to taxation and inflation, the market interest rate is higher than the consumption discount rate (Freeman, 1993; Hanley and Spash, 1993). In addition, the consumption discount rate and the social opportunity costs of capital tend to differ, and a choice needs to be made regarding the discount rate to be used (Pearce and Turner, 1990).

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 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, whereas Nordhaus (1994) argued that a 6% discount rate is most consistent with historical savings data. These rates 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. Through discounting, a much larger weight is attached to the net benefits accruing to current generations as compared to the benefits for future generations. In many circumstances, 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).

In discussing the efficiency and sustainability of ecosystem management, it is important to note that, due to the occurrence of market inefficiencies, it may be perfectly rational for ecosystem managers to pursue socially inefficient management options. Because this thesis takes a social planner approach, focusing on the perspective of the society as a whole, market inefficiencies are not further examined. However, table 2.3 lists a selection of market inefficiencies that are potentially most relevant for ecosystem management. Clearly, if the efficient and/or sustainable ecosystem management options to be identified with the ecological-economic modelling approach developed in this thesis are going to be transformed into environmental policy, consideration of market inefficiencies is crucial (see also Baumol and Oates, 1988; and Tietenberg, 2000).

Table 2.3. Descriptive listing of market inefficiencies

Market inefficiency	General description (from Mäler, 1985; and Tietenberg, 2000).
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.
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.
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 dynamics of the ecosystem, for instance its responses to pollution, and/or inherent complexity in the system, for instance with relation to climate change processes.

2.5 Discussion and conclusions

This thesis deals with the ecological-economic modelling of complex ecosystems, in order to assess the implications of complex dynamics for pursuing efficient and sustainable ecosystem management. This requires the integration of a number of methodologies from environmental and ecological economics and systems ecology, specifically (i) the modelling of economy-ecosystem interactions; (ii) the modelling of ecosystem dynamics; and (iii) the economic valuation of ecosystem services.

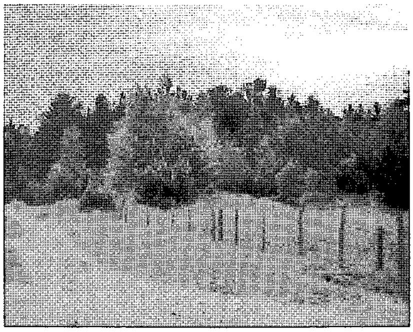
(i) Modelling economy-ecosystem interactions. The interactions between the economic system and ecosystems have been studied extensively in the last century, in particular in relation to the harvesting of natural resources (e.g. Faustmann, 1849; Gordon, 1954; Dillon and Burley, 1961), and economic pressures on ecosystems (e.g. Dasgupta and Heal, 1979; Van Ierland, 1993). Increasingly, there has also been attention for the implications of complex dynamics (Perrings, 1998). For instance, Perrings and Pearce (1994) provide a general model for the efficient management of ecosystems subject to thresholds and irreversible responses. Carpenter et al. (1999) examine the implications of time lags in the responses of a shallow lake ecosystem to eutrophication control measures, and conclude that where such time lags occur, it is under certain conditions efficient to spread pollution control measures in time. Although several general models have been proposed to deal with selected complex ecosystem dynamics, there is a need to further enhance the incorporation of ecosystem complexities in ecological-economic models in order to establish more accurate optimisation models that are better suited to advise policy makers on ecosystem management (Deacon et al., 1998; Hanley, 1999; Turner et al., 2003).

(ii) Modelling ecosystem dynamics. In recent decades, it has become increasingly clear that ecosystems can be subject to a number of non-linear, complex dynamics such as multiple states and thresholds, irreversible responses, and stochasticity and lag effects (Steele and Henderson, 1984; Scheffer et al., 2000; Holling et al., 2002). It is now also recognised that these complex dynamics are common in a wide range of ecosystems including temperate forests, rangelands, shallow and deep lakes, coral reefs, and in relation to oceanic fish stocks (Steele and Henderson, 1984; Scheffer et al., 2001). Hence, there is a need to further examine the implications of these dynamics for the management of ecosystems. This thesis follows a dynamic systems approach to ecological-economic modelling, which offers the best approach to model ecosystem-economy interactions at a disaggregated scale, and examine the implications of ecosystem complexities (Van den Bergh, 1996).

(iii) Valuation of ecosystem services. Economic valuation of ecosystem services is required in order to assess the net costs or benefits of changes in the supply of ecosystem services following the implementation of a management measure. Although substantial progress has been made in the development of valuation methodologies in the past two decades, there are still a number of important methodological uncertainties (Deacon et al., 1998). These relate in particular to the application of the contingent valuation method (Carson, 1998; Deacon et al., 1998), the selection of the discount rate (Howarth and Norgaard, 1993; Hanley, 1999), intertemporal substitution in resource use (Deacon et al., 1998), and the application of valuation methodologies in a developing country context (Dasgupta, 1996; Deacon et al., 1998). Obviously, these issues also constrain the incorporation of value estimates for ecosystem services in ecological-economic models. In this thesis, only relatively simple, well-established valuation methods are used.

On the basis of the literature review, it is anticipated that ecological-economic models constructed according to the proposed framework can be a powerful tool to support environmental decision making. Inclusion of all relevant ecosystem services, and accounting for the complex dynamics of ecosystems provide more realistic indications of the true costs and benefits of ecosystem management options, as opposed to an approach that disregards these services or dynamics (cf. Turner et al., 2003). However, the data requirements of these models are high. Unless suitable ecosystem dynamics model are available from literature, long-term time series on the behaviour of the ecosystem as a function of economic pressures and/or management are required to model the ecosystem. The values of the most relevant ecosystem services need to be known, or the basic data required for their valuation should be collected. A limitation of the framework is that, in its present form, it does not allow for dealing with spatial variability of economy-ecosystem interactions. The framework could, however, be expanded to account for spatial variability by defining the relevant interactions at different spatial scales. In the remainder of the thesis, the framework is applied and tested in order to gain further insights in the nature of ecosystems-economy interactions in general and the implications of complex ecosystem dynamics for the efficient and sustainable management of ecosystems in particular.

3. Efficient and sustainable management of complex forest ecosystems



Adapted from Hein, L.G. and E.C. van Ierland, 2004. Efficient and sustainable management of complex forest ecosystems. Submitted.

3.1 Introduction

The study of the economics and management of renewable resources has a long tradition. Early studies analysed optimal use of forest resources (Faustmann, 1849), while contributions to fisheries management (Gordon, 1954; Schaefer, 1957) are more recent. The standard models assume a logistic growth curve for the resource and consider quality and price changes, cost for inputs and harvesting costs. Faustmann models have been widely used to optimise rotation periods for forest stands (Tahvonen, 1991). 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 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, related to the age of standing stock. Van Kooten et al. (1995) incorporated the benefits of carbon dioxide storage in the model, and Tahvonen and Salo (1999) developed a forest rotation model that includes in situ values and the forest owner's decision making on consumption and savings. Furthermore, Creedy and Wurzbacher (2001) presented a Faustmann model for a forest ecosystem that provides three different ecosystem services (timber, water and carbon sequestration).

Although a large number of modifications of the Faustmann model have been developed, the incorporation of complex ecological feedbacks in these models, has, to date, received little attention (Deacon et al., 1998). Inclusion of ecological feedback effects in the models allows the simulation of complex ecosystem dynamics (Costanza et al., 1993), and it is increasingly recognised that complex dynamics have major implications for ecosystem management (Mäler, 2000; Scheffer et al., 2001). Ecosystem complexities include irreversible responses of the ecosystem, as well as several types of non-linear dynamics, including multiple states, thresholds and hysteresis (Scheffer et al., 2000).

The aim of this paper is to examine the implications of irreversible responses for the efficient and sustainable management of a forest ecosystem. To achieve this, an extension of the existing Faustmann models is developed that captures the potentially irreversible response of a forest ecosystem to degradation. The paper is built around two simple dynamic systems models of a hypothetical forest ecosystem, one representing an ecosystem that responds reversibly to stress, and the other an ecosystem that responds irreversibly to stress. Contrary to the existing Faustmann models, the models contain two components, forest cover and topsoil, and their interactions. 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, that respond in a reversible respectively irreversible manner to overharvesting, are developed in section 3.2. The efficiency and sustainability of different felling rates are compared, for the two models, in section 3.3. A discussion is presented in section 3.4, and the main conclusions are provided in section 3.5.

3.2 Description of the ecosystem models

The modelling framework

The modelling framework for this study is shown in figure 3.1. 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 higher erosion rates. The models represent hypothetical ecosystems, with assumed values based upon literature.

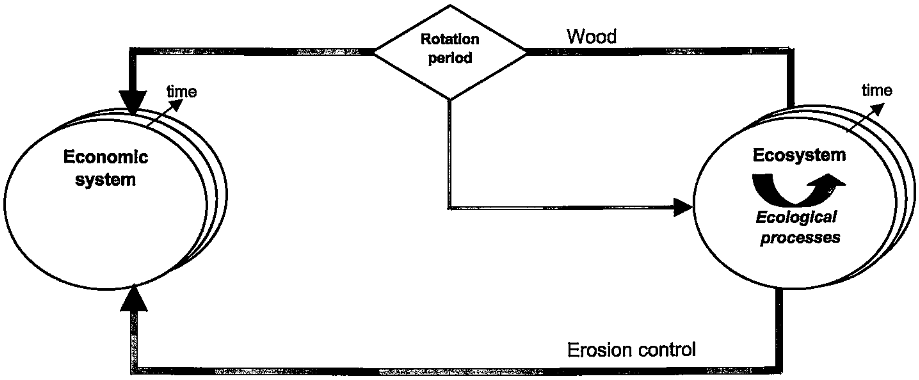


Figure 3.1. Modelling framework for a forest ecosystem supplying wood and erosion control. Both services are first expressed in a physical unit (ton), which is then converted into a monetary unit (US$).

The economic system is considered exogenous. It is assumed that, at 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. The prices are derived from literature (LeDoux and Huyler, 2000; Uri and Lewis, 1998). 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. 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 our models, we assume a constant discount rate and we neglect the capital value of land (but see e.g. Penttinen, 2000 for more information on the implications of variable prices).

Based on the 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 meters). The ecosystem's carrying capacity (maximum forest cover) is assumed to be 200 ton wood per hectare.

With the models developed in this paper, we calculate the efficiency and the sustainability of different ecosystem management options. We define *efficient* ecosystem management as management that maximises a utility function, including the benefits of all services supplied by the ecosystem, and the costs involved in providing or accessing the service (Chiang, 1992). Future benefits and costs are discounted, using a fixed discount rate, in order to compare them with current benefits and costs. The optimal solutions are found through a simulation approach, as it proved highly complex to solve the first order conditions for the two ecosystems in an algebraic manner. Nevertheless, the first order conditions are formulated and included in Annex 1. *Sustainable* ecosystem management is 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 (based upon WCED, 1987; Pearce et al., 1989; and Barbier and Markandya, 1990). Substitutability between production factors and technological progress are not considered in this study, and, hence, sustainability as used in this paper implies non-decreasing natural capital (Carter, 2001).

Ecosystem model #1: reversible response to stress

Model #1 represents a simplified model of a dynamic 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 – for our calculations we assume that the initial topsoil depth measures 3 cm.

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

$TS_{(t)}$ = topsoil depth in meters at time t;
$E_{(t)}$ = erosion in meters at time t; and
$RE_{(t)}$ = recovery of the topsoil from erosion through accumulation of sediment and organic material in year t, expressed in meters.

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 meter plot (which equals 200 ton/ha). Forest cover develops as a function of the harvest of wood, and the natural growth of the vegetation:

$$FC_{(t)} = FC_{(t-1)} - W_{(t)} + G_{(t)}\text{, with} \tag{2}$$

$FC_{(t)}$ = 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:

$$W(t) = 0.6 * FC(t) \;\Big|\; \textit{once every R years}\text{, with} \tag{3}$$

$W(t)$ = 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 meter 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 and Morgan, 1995) :

$$E_{(t)} = \alpha * e^{-2.5 * FC_{(t)}}\text{, with} \tag{4}$$

$FC_{(t)}$ = 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 m/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. As we expect (i) the accumulation to increase with forest cover; and (ii) marginal increases to progressively diminish, the following logarithmic relation is assumed in this study:

$$RE_{(t)} = \gamma * FC_{(t)}{}^{\phi}, \text{ with} \qquad (5)$$

γ and ϕ being constants for a specific ecosystem ($\phi < 1$), depending in particular on soil type 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:

$$G_{(t)} = r_{max} * FC_{(t)} * (1 - FC_{(t)} / K) \qquad (6a)$$

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

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 years period:

$$NPV = \sum_{t=1}^{100} (p_w . W_{(t)} - c_e . E_{(t)})\ \delta^t, \text{ with} \qquad (7)$$

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 (7) are shown in table 3.1. For simplicity, the net price of wood and the costs of erosion are average, constant values derived from literature. The NPV of different rotation periods is calculated for discount rates of 2.5% and 5%.

Table 3.1. 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 North-eastern 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)

Overview of the variables and parameters used in the model. An overview of the *variables* used in the model is presented in table 3.2. The various *parameters* used in the model are shown in table 3.3. The parameters have been selected in such a way that potentially realistic erosion, sedimentation and regrowth rates are obtained (as explained in the table).

Table 3.2. 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 3.3. 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 e.g. 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

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 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 ration of 20. Based upon Smith et al. (2000) and Williamson and Nielsen (2003), we assume a linear relation between topsoil depth and forest regrowth up to a topsoil depth of 30mm. For a topsoil of 30 mm or more, there is no further change in forest growth. We also assume that the absence of topsoil leads to a complete stop of the regrowth of the vegetation (see figure 3.2). This alters equation (6a) to equation (6b), as described below. All other equations remain the same.

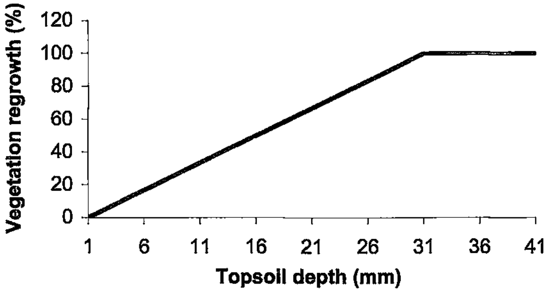


Figure 3.2. Relation between topsoil depth and vegetation regrowth

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 3 cm (figure 3.2):

$$G_{(t)} = r_{max} * FC_{(t)} * (1 - FC_{(t)} / K) * Min(1; 0.0333 * TS_{(t)}) \quad (6b)$$

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

3.3 Efficient and sustainable management practices for the modelled ecosystems

Introduction

The paper follows a simulation modelling approach to the optimisation of ecosystem management. The models are used to calculate the economic efficiency of three types of management for each ecosystem model. First, we calculate the fixed rotation period that provides maximum benefits, i.e. the efficient rotation period. Second, we calculate the economic efficiency of the shortest rotation period that can be qualified as sustainable (this corresponds to the most efficient sustainable rotation period). Third, we calculate, for each model, an alternative management option that provides a compromise between efficient and sustainable forest management.

For ecosystem #1, this compromise 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. We calculate the most efficient rotation scheme, based upon a period of intensive harvesting and a recovery period, that qualifies as sustainable over a hundred years period.

For ecosystem #2, which responds irreversibly to stress, the forest and topsoil stocks irreversibly collapse if certain thresholds are exceeded. These thresholds relate to the minimum sustainable stock, i.e. the minimum amount of topsoil and forest cover that need to be preserved in order to allow recovery of the system. For this ecosystem, we calculate the most efficient rotation scheme subject to maintaining the minimum sustainable stock. This involves a variable rotation period.

In all assessments, 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. The general efficiency conditions of a Faustmann model of an ecosystem that supplies the services wood and erosion control are presented in Annex 3.1.

Ecosystem #1 : assessment of efficient and sustainable rotation periods

The efficient rotation period. The model has been run for a range of rotation periods, and the generated NPV has been 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 years rotation period generates the maximum NPV. This maximum NPV is US$ 845, for the 30 by 30 meter 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 years rotation period is presented in figure 3.3, and the development of the forest cover and the topsoil is shown in figure 3.4. It is assumed that, once the topsoil is depleted, erosion of the subsoil will continue, according to the same equation.

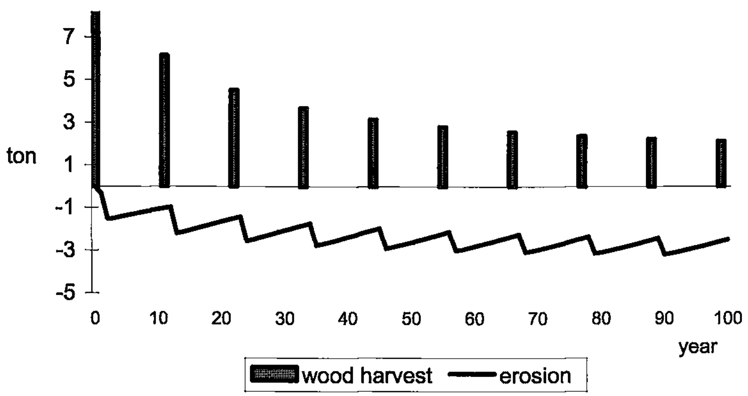


Figure 3.3. Supply of two ecosystem services, wood and erosion control, at a felling rate of 11 years.

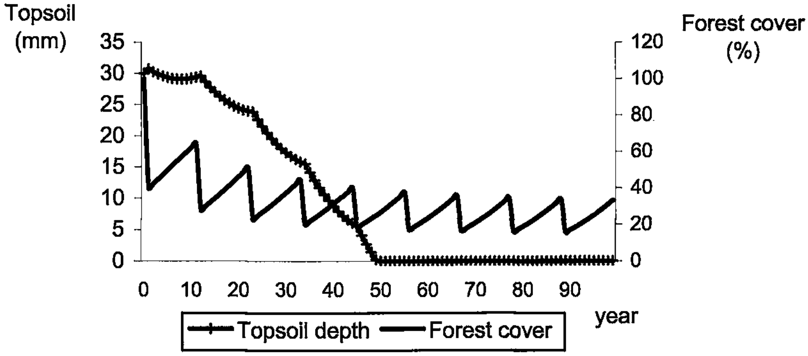


Figure 3.4. Development of the forest cover and topsoil for an 11 years felling cycle.

The sustainable rotation period. Clearly, the 11-years 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 ecosystem #1, regrowth of the forest cover continues even when the topsoil is totally eroded. With the 16 years rotation period, efficient in case 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, 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 3.5 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.

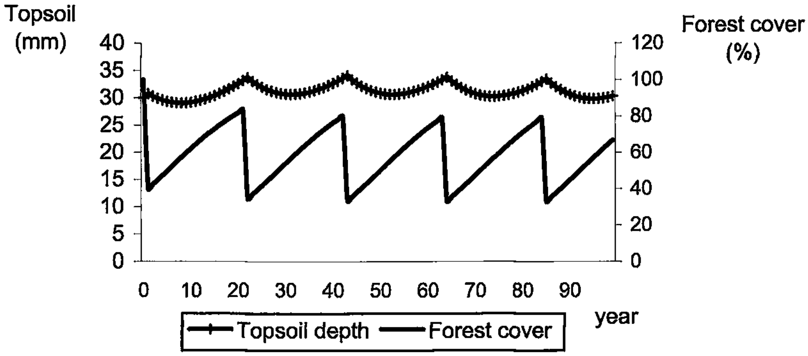


Figure 3.5. Development of the topsoil and forest cover under a sustainable, fixed rotation period of 21 years.

Intensive harvesting followed by a recovery period. For Ecosystem model #1, it is also possible to achieve sustainability, over a 100-years 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 3.6. 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.

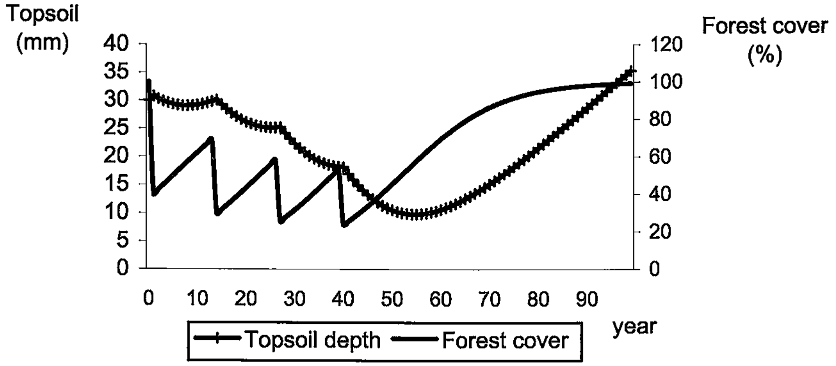


Figure 3.6. Sustainable management with a variable rotation period.

The results for ecosystem #1 are summarized in table 3.4.

Table 3.4. 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 maximization	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 stabilization 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

Assessment of efficient and sustainable felling rates for ecosystem #2

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 3.7. 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.

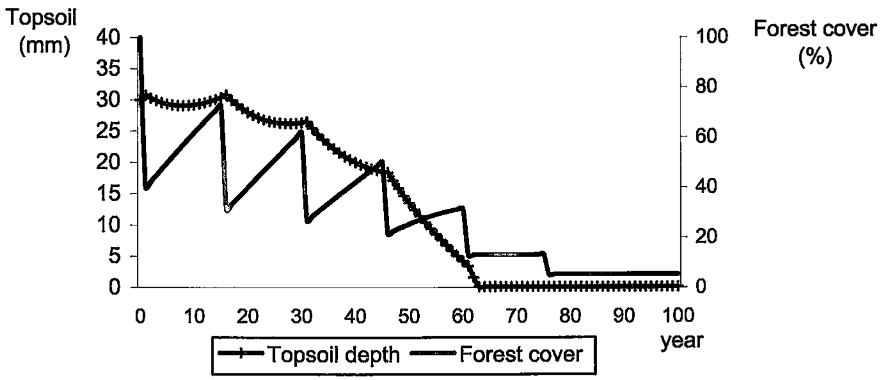


Figure 3.7. Development of the topsoil and vegetation cover of ecosystem #2, for a 15 years felling cycle.

The sustainable rotation period. As with ecosystem #1, the efficient rotation period is not sustainable. At the 15 years rotation period, the topsoil is depleted in year 63, and the forest cover in year 77. The 19 years rotation period, efficient if a discount rate of 2.5% is used, also leads to a, 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.

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 3.8. 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.

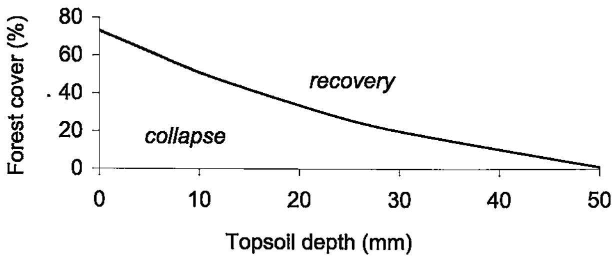


Figure 3.8. Minimum sustainable stock levels for ecosystem #2. At values below the curve, the topsoil and forest cover of the ecosystem will irreversibly collapse to 0.

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 3.9. 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.

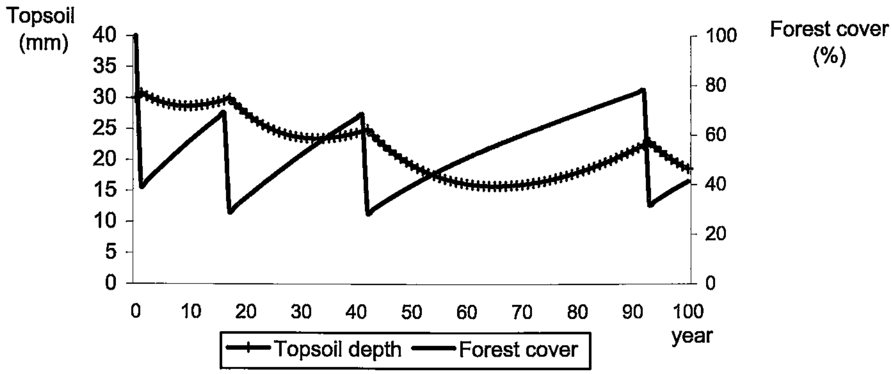


Figure 3.9. Harvesting of forest ecosystem #2, subject to maintaining the minimum sustainable stocks.

The results for ecosystem #2 are summarized in table 3.5.

Table 3.5. 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 maximization	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

3.4 Discussion

Introduction. Using a dynamic systems modelling approach, this paper presents an extension of the existing Faustmann models to calculate optimal rotation periods for a forest stand. In this section, we will discuss the methodological aspects of the followed modelling approach, as well as the potential implications for forest management.

Modelling forest ecosystems. Compared to the Faustmann models (Faustmann, 1849; Brazee, 2001), our models characterise the ecosystem through two instead of one ecosystem component: forest cover and topsoil. This allows the refinement of the response of the ecosystem to harvesting; in particular, we are able to model the development of the topsoil, which, in turn, influences the establishment of forest seedlings and the regrowth of the forest stand (FAO, 1992). Two systems are considered, that respond in a reversible respectively an irreversible manner to intensive harvesting. Although the models provide enhanced ecological accuracy compared to the original Faustmann models, a number of ecological (and economic) 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 are a major cause of erosion because it causes 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 leads to relatively short efficient rotation periods as compared to a situation where loggers also have to consider the fixed costs of harvesting a particular site.
4. The model assumes constant prices for the two ecosystem services involved.

The ecological refinements of the models are most relevant for hillside ecosystems prone to erosion. Adding a topsoil component to the model allows for more accurate modelling of the ecosystem dynamics, as well as for assessing the impact of wood harvesting on erosion. The models show that intensive harvesting increases erosion of the plots. This reduces the growth of the forest stock, and bring 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).

Furthermore, the consideration of two ecosystem components, and their interactions, allows for the inclusion of irreversible ecosystem dynamics in the model. Such dynamics have been found in a range of ecosystems, including forests (Ulrich, 1992; Albers, 1996; Scheffer et al., 2001; Holling and Gunderson, 2002) but they have, to date, to our knowledge not yet been incorporated in the Faustmann models. The occurrence of irreversible responses to management is of particular importance in case 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, see e.g. Levin, 1992 and 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.

Implications for forest management. The assessment 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; Howarth and Norgaard, 1993; Freeman, 1993; and Hanley, 1999). The inclusion of the full set of ecosystem services in the assessment is required in order to obtain a correct picture of the benefits supplied by the ecosystem under various management regimes (Turner et al., 2003). In this paper, the inclusion of the erosion control service of forests leads to the selection of a longer, more sustainable rotation period compared to a situation where 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 (see e.g. Chomitz et al., 1998).

This paper shows how the possibilities to partially reconcile efficiency and sustainability considerations through a rotation scheme based upon variable rotation periods depend upon the specific dynamics of the ecosystem. If an ecosystem responds reversibly to stress, this provides the possibility to exploit the ecosystem intensively during an initial period, and allow it to recover in the subsequent years. When the 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. If high discount rates are applied, this may be a much more profitable option than pursuing sustainability through a constant, low felling rate. 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 case 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, in particular 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).

In the deterministic model developed in this paper, 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), as proposed by Ciricay-Wantrup (1968), modified by Bishop (1978) (see also Randall and Farmer, 1995). The implications of our paper are also relevant to the application of the SMS concept; specifically, our paper shows how the minimum sustainable stock, and hence the SMS, may depend upon two ecosystem state indicators and their interactions.

3.5 Conclusions

This paper examines the implications of irreversible responses for the efficient and sustainable management of a forest ecosystem. This has been examined with an ecological-economic model that includes both a forest cover and a topsoil component, and their interactions. Two different models have been constructed using a dynamic systems modelling approach. They represent ecosystems that respond in a reversible, respectively in an irreversible manner to overharvesting. The models developed in this paper allow for a more ecologically accurate modelling of the dynamics, and the impacts of wood harvesting on a forest ecosystem, compared to the traditional Faustmann approaches to determine optimal rotation periods.

The paper illustrates the potentially large discrepancy between efficient and sustainable management, depending upon the discount rate used. Partial reconciliation of efficient and sustainable management is possible through the use of a rotation scheme based upon variable rotation periods. The specific possibilities for this depend upon the dynamics of the ecosystem. If the ecosystem shows a *reversible* response to stress, overharvesting in one period can be compensated by reduced harvesting in a subsequent period. In this way, long-term sustainability may be achieved, and the corresponding loss in NPV is only small compared to applying a constant, low felling rate.

For the *irreversible* ecosystem model, which may represent a more realistic model of a hill-side forest ecosystem, such an option does not exist. Once the system is depleted below the minimum sustainable stock level, the system will not be able to recover, even if all pressures are eliminated. In this case, a variable rotation period may be used that maintains the minimum sustainable stock. This rotation scheme may yield a significantly higher NPV than a sustainable management scheme based upon a fixed rotation period, in particular at high discount rates, and it allows future generations to recuperate the ecosystem - at the expense of a temporary reduction in harvest rates.

Annex 3.1 Efficiency condition for a forest ecosystem supplying wood and erosion control

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:

$$p_w \,.\, dFC/dR \;+\; c_e \,.\, \Delta E \;=\; i \,.\, p_w \,.\, FC \;+\; i \,.\, \Pi$$

with:

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 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 paper, it is assumed that Π = 0. The efficiency equation becomes:

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

with:
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

4. Spatial scales, stakeholders and the valuation of ecosystem services

Adapted from Hein, L.G., K. van Koppen, R.S. de Groot and E.C. van Ierland, 2003. Spatial scales, stakeholders and the valuation of ecosystem services. Submitted.

4.1 Introduction

Starting in the late 1960s, there has been a growing interest in the analysis and valuation of the multiple benefits provided by ecosystems. This interest was triggered by an increasing awareness that the benefits provided by natural and semi-natural ecosystems were often underestimated in decision-making (Helliwell, 1969, Odum and Odum, 1972, Van der Maarel and Dauvellier, 1978). Since then, economic valuation of ecosystems has received much attention in scientific literature. Methodologies for the valuation of ecosystem services have been developed by, amongst others, Dixon and Hufschmidt (1986), Pearce and Turner (1990), Freeman (1993), and Hanley and Spash (1993), whereas the value of the services of a particular ecosystem has been assessed by, for example, Ruitenbeek (1994), Kramer et al. (1995) and Van Beukering et al. (2003). In addition, several studies have provided frameworks for the valuation of ecosystem services (Turner et al., 2000; De Groot et al., 2002; Millennium Ecosystem Assessment, 2003).

To date, relatively little elaboration of the scales of ecosystem services has taken place. Specifically, there is a need to examine the various scales at which ecosystem services are generated and used, and, subsequently, how the supply of ecosystem services affects the interests of stakeholders at different scales (Tacconi, 2000; Turner et al., 2000; Millennium Ecosystem Assessment, 2003; Turner et al., 2003). An increased understanding of the spatial scales of ecosystem services is especially important for the analysis, and resolution of conflicts in ecosystem management. Analysis of the scales of ecosystem services reveals the interests of different stakeholders, and provides a basis for balancing the various interests in ecosystem management (Tacconi, 2000).

Therefore, in this paper we analyse the *spatial* scales at which ecosystem services are supplied, and the implications of these scales for the values attached to ecosystem services by different stakeholders. For a discussion of the *temporal* scales, the reader is referred to, for example, Howarth and Norgaard (1993) and Hanley (1999). On the basis of existing literature, we first present a consistent framework for the valuation of ecosystem services, specifically considering the issue of double counting of services - one of the remaining issues in ecosystem valuation (De Groot et al., 2002; Millennium Ecosystem Assessment, 2003). Subsequently, we examine the spatial scales at which ecosystem services are supplied - based upon a review of relevant ecological and ecological economics literature. To illustrate the approach and to show the spatial scales of ecosystem services for a concrete ecosystem, a case study is presented that includes a valuation of the ecosystem services supplied by the De Wieden wetland in the Netherlands, and an assessment of the scales at which these services are delivered. The De Wieden case study is based upon fieldwork and interviews with all major stakeholders of the area, conducted in the period January - September 2003.

The chapter is organised as follows. In section 4.2, a framework for the assessment of ecosystem services is established. In section 4.3, the spatial scales of ecosystem services are analysed. In section 4.4, the framework is applied to the De Wieden wetland, and the spatial scales at which selected ecosystem services are supplied are analysed. This is followed by a discussion and a conclusion in sections 4.5 and 4.6, respectively.

4.2 The ecosystem services valuation framework

Introduction. Based upon a literature review, this section establishes a framework for the valuation of ecosystem services. The framework includes three types of services (production, regulation and cultural services) and four types of value (direct and indirect use, option and non-use value). The types of ecosystem services are adapted from De Groot et al. (2002) and Millennium Ecosystem Assessment (2003). The value types included in the framework are based upon Pearce and Turner (1990). The framework is presented in figure 4.1. It is applicable to all ecosystems, but it will in general be more useful to apply it to natural or semi-natural (modified) ecosystems. This because of the specific attention paid to the goods and services provided by the regulation and cultural services, which are often higher in natural and semi-natural systems (De Groot, 1992; Costanza et al., 1997a).

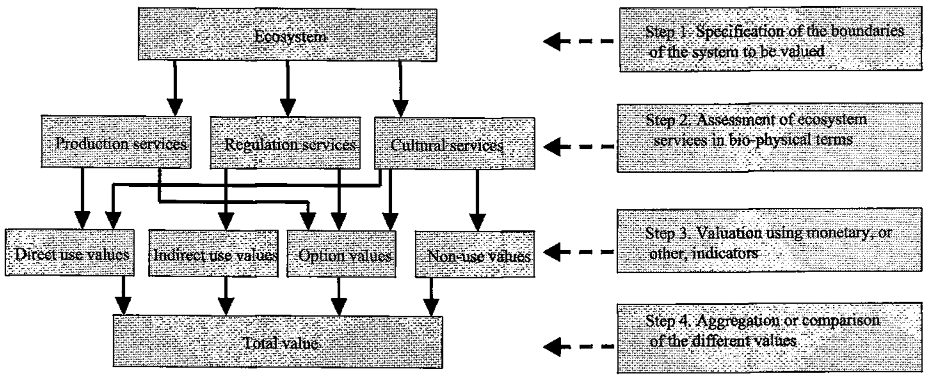


Figure 4.1. The ecosystem valuation framework. The solid arrows represent the most important links between the different elements. The dashed arrows indicate the four principal steps in the valuation of ecosystem services. Based upon Pearce and Turner (1990), De Groot et al. (2002) and Millennium Ecosystem Assessment (2003).

Following this framework, valuation of ecosystem services consists of four steps: (i) specification of the boundaries of the ecosystem to be valued; (ii) assessment of the ecosystem services supplied by the system; (iii) valuation of the ecosystem services; and (iv) aggregation or comparison of the values of the services. These steps are discussed below.

(i) Specification of the boundaries of the ecosystem to be valued. Valuation (as any other analysis) requires that the object of the valuation is clearly defined. The Convention on Biological Diversity provided the following definition of an ecosystem "a dynamic complex of plant, animal and micro-organism communities and their nonliving environment interacting as a functional unit" (United Nations, 1992). However, we argue that a *spatial* definition is required to describe the ecosystem to be valued, and we use the following definition of an ecosystem: '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). The ecosystem to be valued may contain a number of different (sub-) ecosystems.

(ii) Assessment of the services supplied by the ecosystem. Ecosystem services are the goods or services provided by the ecosystem to society, and provide the basis for the valuation of the ecosystem. The supply of ecosystem services will often be variable over time, and, where relevant, both actual and potential future supplies of services have to be included in the valuation (Drepper and Månsson, 1993; Barbier, 2000; Mäler, 2000). We propose to distinguish three different categories of ecosystem services: 'production services', 'regulation services' and 'cultural services', based upon De Groot et al. (2002) and Millennium Ecosystem Assessment (2003). Table 4.1 presents the three categories, as well as an overview of the various ecosystem services in each category.

Contrary to the Millennium Ecosystem Assessment (2003), we do not distinguish the category 'supporting services', which represents the ecological processes that underlie the functioning of the ecosystem. Their inclusion in valuation may lead to double counting – 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 ecosystems, and it is unclear on which basis supporting services should be included in, or excluded from a valuation study.

Before the services can be valued, they have to be assessed in bio-physical terms. For *production* services, this involves the quantification of the flows of goods harvested in the ecosystem, in a physical unit. For most *regulation* services, quantification requires spatially explicit analysis of the bio-physical impact of the service on the environment in or surrounding the ecosystem. For example, valuation of the hydrological service of a forest first requires an assessment of the precise impact of the forest on the water flow downstream, including such aspects as the reduction of peak flows, and the increase in dry season water supply (Bosch and Hewitt, 1982). The reduction of peak flows and flood risks is only relevant in a specific zone around the river bed, which needs to be (spatially) defined before the service can be valued. An example of a regulation service that does usually not require spatially explicit assessment prior to valuation is the carbon sequestration service – the value of the carbon storage does not depend upon where it is sequestered. *Cultural* services depend upon a human interpretation of the ecosystem, or of specific characteristics of the ecosystem. The benefits people obtain from cultural services depend upon experiences during actual visits to the area, indirect experiences derived from an ecosystem (e.g. through nature movies), and more abstract cultural and moral considerations (see e.g. Aldred, 1994). Assessment of cultural services requires assessment of the numbers of people benefiting from the service, and the type of interaction they have with the ecosystem involved.

Table 4.1. List of ecosystem services (based on Van der Maarel and Dauvellier, 1978; Ehrlich and Ehrlich, 1981; Costanza et al., 1997a; De Groot et al., 2002; Millennium Ecosystem Assessment, 2003).

Category	Definition	Examples of goods and services provided
Production services	Production services reflect goods and services *produced* in the ecosystem	- Food - Fodder (including grass from pastures) - Fuel (including wood and dung) - Timber, fibres and other raw materials - Biochemical and medicinal resources - Genetic resources - Ornamentals
Regulation services	Regulation services result from the capacity of ecosystems to regulate climate, hydrological and bio-chemical cycles, earth surface processes, and a variety of biological processes.	- 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	Cultural services relate to the benefits people obtain from ecosystems through recreation, cognitive development, relaxation, and spiritual reflection.	- 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

(iii) Valuation of the ecosystem services. The values that are attributed to ecosystem services depend upon the stakeholders benefiting from these services. The classic definition of a stakeholder is "any group or individual who can affect or is affected by the achievement of the organization's objective" (Freeman, 1984). For ecosystem valuation, we modify this definition into "any group or individual who can affect or is affected by the ecosystem's services". The value of ecosystem services depends upon the views and needs of stakeholders (Vermeulen & Koziell, 2002), and there is a mutual and dynamic relationship between ecosystem services and stakeholders. The services supplied by an ecosystem determine the relevant stakeholders, and the stakeholders determine relevant ecosystem services. The four value types that stakeholders can attribute to ecosystem services are discussed below.

- *Direct use values* arise from human direct utilization of ecosystems (Pearce and Turner, 1990), for example through the sale or consumption of a piece of fruit. All production services, and some cultural services (such as recreation) have direct use value.

- *Indirect use values* stem from the indirect utilization of ecosystems, in particular through the positive externalities that ecosystems provide (Munasinghe and Schwab, 1993). This reflects the type of benefits that regulation services provide to society.

- *Option values.* Because people are unsure about their future demand for a service, they are willing to pay to keep open the option of using a resource in the future – insofar as they are, to some extent, risk averse (Weisbrod, 1964; Pearce and Turner, 1990). 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. Although theoretically correct, the quasi-option value is in practice very difficult to assess (Turner et al., 2000)

- *None-use values* are 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, e.g., related to the notion that animal and plant species may 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 recognize 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.

Applicable valuation methods differ for private and public services. The marginal value of private goods can generally be derived from market prices, whereas marginal values of public goods have to be established using non-market valuation techniques. These include 'stated preference' approaches, such as the Contingent Valuation Method (CVM) and related methods, and 'revealed preference' approaches. Revealed preference techniques use a link with a market good or service to indicate the willingness-to-pay for the service. Valuation of non-use values is particularly cumbersome. Different authors have tried to express them in monetary values (see Nunes and Van den Bergh, 1999) or non-monetary indicators (Wathern et al., 1986; and Margules and Usher, 1991). For details on valuation techniques, see Dixon and Hufschmidt (1986), Pearce and Turner (1990), Hanley and Spash (1993), Pearce and Moran (1994), Willis and Garrod (1995), and Brouwer et al. (1997).

(iv) Aggregation or comparison of the values. In principle, the four value types 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 is the total value of the ecosystem (Pearce and Turner, 1990). If all values have been expressed as a monetary value, and if the values are expressed through comparable indicators (e.g. consumer and/or producer surplus), the values can be summed. If non-monetary indicators are used for the non-use values, the values can be presented side-by-side –leaving it to the reader to compare the two value types (as in Strijker et al., 2000). Alternatively, they can be compared using Multi Criteria Assessment (MCA). With MCA, stakeholders can be asked to assign relative weights to different sets of indicators (non-monetary as well as monetary), enabling comparison of the indicators (Nijkamp and Spronk, 1979; Costanza and Folke, 1997). Different stakeholder groups can be expected to have different perspectives on the importance of the different types of value (Vermeulen en Koziell, 2002). Through group valuation, or the use of deliberative processes, stakeholders can be encouraged to converge to a representative assessment of the values of different ecosystem services (O'Neill, 2001).

Double counting. An important issue in the valuation of ecosystem services is the *double counting of services* (Millennium Ecosystem Assessment, 2003; Turner et al., 2003). The various processes involved in the regulation services are paramount to the functioning of ecosystems, and in that sense underlie many other services. However, including both the regulation and these other services in the assessment of the total value of an ecosystem may lead to double counting. For example, pollination is crucial to sustaining the fruit production of an area. Including both the pollination service and the service 'production of fruit' would lead to double counting – the value of the pollination of fruit trees is already included in the value of the fruits. In this paper, it is proposed to deal with double counting by arguing that regulation services should only be included in the valuation if (i) they have an impact outside the ecosystem to be valued; and/or (ii) if they provide a *direct* benefit to people living in the area (*i.e.* not through sustaining or improving another service). For example, an impact outside an ecosystem that needs to be included in the valuation occurs when an ecosystem is supporting a population of bees that plays an important role in the pollination of crops in adjacent fields. As long as these fields are not part of the ecosystem to be valued, this is an additional service provided by the system which needs to be included in its total value. 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 regulation services is that the ecosystem is defined in terms of it's spatial boundaries – otherwise the external impacts of the regulation services can not be precisely defined.

4.3 Scales of ecosystem services

Introduction. Scales refer to the physical dimension, in space or time, of phenomena or observations (O'Neill and King, 1998). Ecosystem services are supplied to the economic system at a range of spatial and temporal scales, varying from the short-term, site level (e.g. amenity services) to the long-term, global level (e.g. carbon sequestration) (Turner et al., 2000; Limburg et al., 2002;). Scales and stakeholders are often correlated, as the scale at which the ecosystem service is supplied determines which stakeholders may benefit from it (Vermeulen and Koziell, 2002). This section analyses (i) scales of ecosystems; (ii) scales of socio-economic systems; (iii) scales of ecosystem services; and (iv) the relation between scales and stakeholders' interests.

Scales of ecosystems. According to its 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 it is usually required to define the scale of a particular analysis, 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 4.2. In such a classification of ecological scales, it is common to include the ecosystem itself as a particular scale, for example in terms of a 'forest ecosystem'.

Ecological processes, that guide the development of the ecosystem, take place over a large range of spatial and temporal scales. This ranges from competition between individual plants at the plot level, via meso-scale processes such as fire and insect outbreaks, to climatic and geomorphologic processes at the largest spatial and temporal scales (Clark et al., 1979; O'Neill and King, 1998; Holling et al., 2002). In general, large-scale, long-period phenomena set physical constraints on smaller scale, shorter period ones (Limburg et al., 2002). However, large scale processes may be driven by the joint impact of small scale processes (Levin, 1992). For example, microbes operate on the scale of micrometers and minutes, but their cumulative activity determines a larger scale process such as the nutrient cycle, e.g. through demineralisation of organic material and nitrogen fixation.

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

Scales of socio-economic systems. 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 utilization of capital, labour and natural resources are taken (North, 1990). At the lowest institutional level, this includes individuals and households. At higher institutional scales can be distinguished: the communal or municipal, state or provincial, national, and international level (see figure 4.2). 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 (Limburg et al., 2002).

The supply of ecosystem services may affect stakeholders at all institutional levels (Berkes and Folke, 1998; Peterson, 2000). Households, as well as local or internationally operating firms may directly depend upon ecosystem services for their income (e.g. fishermen, ecotourism operators). Government agencies at different levels are involved in managing ecosystems, and in regulating the access to ecosystem services. They may also receive income from specific ecosystem services (park entrance fees, hunting licenses). Ultimately, all individuals depend upon the essential regulation (life-support) services of ecosystems. Ecological and institutional boundaries seldom coincide, and stakeholders in ecosystem services often cut across a range of institutional zones and scales (Cash and Moser, 1998).

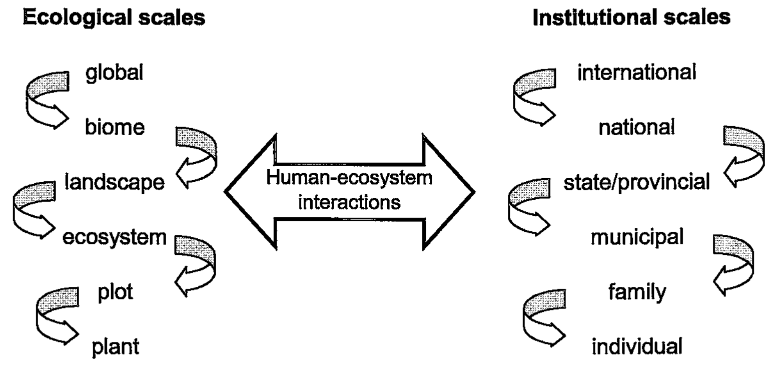


Figure 4.2. Selected ecological and institutional scales (adapted from Leemans, 2000)

Scales of ecosystem services. In the previous paragraphs, we argued that ecosystem services can be generated at a range of ecological scales, and can be supplied to stakeholders at a range of institutional scales (see figure 4.2). We will now briefly discuss this in more detail for the three categories of ecosystem services distinguished in our framework.

Production services. The possibility to harvest products from natural or semi-natural ecosystems depends upon the availability of the resource, or the stock of the product involved. The development of the stock is determined by the development of the ecosystem as a function of ecological processes and human interventions. To analyse the ecological impacts of the resource use, or the harvest levels that can be (sustainably) supported, the appropriate scale of analysis is the level of the ecosystem supplying the service (e.g. the lake, or the Northern Atlantic ocean) (Levin, 1992). The benefits of the resource may accumulate to stakeholders at a range of institutional scales (Turner et al., 2000). Local residents, if present, 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, this is the case if a marine ecosystem is fished by an international fleet, or if a particular genetic material or medicinal plants is processed and/or consumed at a larger institutional scale (see e.g. Blum, 1993).

Regulation services. A regulation service can be interpreted as an ecological process that has (actual or potential) economic value because it has an economic impact outside the studied ecosystem and/or if it provides a *direct* benefit to people living in the area (see the previous section). Because the ecological processes involved take place at certain, ecological scales, it is often possible to define the specific ecological scale at which the regulation service is generated (see table 4.2). For many regulation 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.

Table 4.2. Most relevant ecological scales for the regulation services – note that some services may be relevant at more than one scale. Based upon Hufschmidt et al. (1983), De Groot (1992), Kramer et al. (1995) and Van Beukering et al. (2003).

Ecological scale	Dimensions	Regulation services
Global	> 1,000,000 km^2	Carbon sequestration
		Climate regulation through regulation of albedo, temperature and rainfall patterns
Biome – landscape	10,000-1000,000 km^2	Regulation of the timing and volume of river and ground water flows
		Protection against floods by coastal or riparian ecosystems
		Regulation of erosion and sedimentation
		Regulation of species reproduction (nursery service)
Ecosystem	1-10,000 km^2	Breakdown of excess nutrients and pollution
		Pollination (for most plants)
		Regulation of pests and pathogens
		Protection against storms
Plot – plant	< 1 km^2	Protection against noise and dust
		Control of run-off
		Biological nitrogen fixation (BNF)

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, in particular 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 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).

Scales and stakeholders' interests. 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 will be found depending upon the institutional level at which the analysis is performed. For example, local stakeholders may particularly value a production 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 indicate on whose perspectives the values are based.

4.4 Analysis of the ecosystem services provided by the De Wieden wetland

Introduction

The spatial scales at which ecosystem services are supplied are examined for the De Wieden wetland in the Netherlands. Subsequently, we analyse (i) the values of selected ecosystem services supplied by De Wieden; and (ii) the scales at which these ecosystem services are supplied to stakeholders. As the main aim of this section is to examine the scales of ecosystem services, relatively simple valuation techniques have been used.

Valuation of the ecosystem services supplied by De Wieden

Step 1. Specification of the study area. De Wieden is one of the most extensive lowland peatlands in north-western Europe, and it includes a large range of waterbodies of different sizes (lakes, canals, marshlands), reedlands, extensive agricultural land and forests. For this study, a case study area has been selected that comprises the central part of De Wieden, in total around 5200 ha. It includes the four biggest lakes and the surrounding area (figure 4.3).

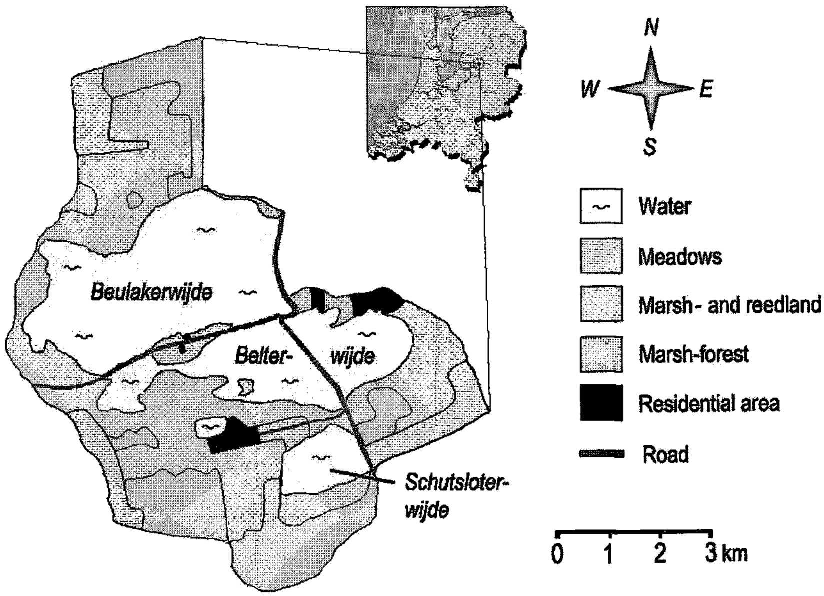


Figure 4.3. Map and location of the study area

Step 2. Assessment of ecosystem services. Four ecosystem services have been selected for this study: (i) reed cutting; (ii) fisheries (both production services); (iii) recreation; and (iv) the habitat service (both cultural services). These services have been selected in order to obtain a mix of services important for stakeholders at different levels, and because of the availability of data for these four services. Two other ecosystem services of the wetland that are not further considered in this study are the amenity service and the water purification service. The *amenity* service reflects that De Wieden enhances the local living conditions by the provision of an attractive environment. The service accrues to local stakeholders. It is excluded because the fisheries and recreation services also provide benefits to local stakeholders, and data is more readily available for these two services. The *water purification* service is based upon the breakdown and absorption of pollutants in the wetland. The water purification service is reflected in enhanced opportunities for recreation and nature conservation, but, to avoid double counting, this should not be included in the valuation. Nevertheless, the water purification service also leads to the inflow of cleaner water in lake Ijsselmeer downstream, compared to a situation without De Wieden. This represents an economic value of the water purification service. However, as only around 0.7% of the water that Lake Ijsselmeer receives flows through De Wieden, the impact on the overall water quality of the lake is probably modest and the service is excluded from further analysis.

(i) Reed cutting. The reed of De Wieden has been cut since several centuries, and is used mainly 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.

(ii) Fisheries. 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, 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).

(iii) Recreation. De Wieden is an important area for recreation, attracting visitors that come for short holidays as well as day-trips. Visitors enjoy a range of activities including boating, sailing, hiking, fishing, canoeing, surfing, swimming and sun-bathing (see table 4.3). The authors of this study have estimated the number of people visiting the beaches for swimming and/or sunbathing, and the number of recreational fishermen. The visitors to the 9 (mostly small) *beaches* in the case study area have been counted during 10 sunny days spread over the spring, summer and autumn of 2003. The average number of visitors for these days (447) has been multiplied with the yearly average number of warm (> 25^{o} C) and sunny days (18), derived from climate statistics (KNMI, 2003). Furthermore, it has been assumed that *recreational fisheries* takes place on days without rain, during the period 1 April – 30 September (interviews with fishermen showed that the large majority didn't fish during the cold winter season, or on rainy days). The number of fishermen, as found in surveys on 20 dry days in the selected period (28) has been multiplied with the average number of days without rain in these 6 months (72) (KNMI, 2003).

Table 4.3. Estimate of the number of people involved in various recreational activities in De Wieden per year

Activity	Number of visitor-days per year	Source
People visiting the walking trails and information centre (in 2002)	61,404	Natuurmonumenten, 2003a
Swimming/sunbathing	8,040	This study
Fishing	2,050	This study
Boating		
- motorboats	82,165	Number of boats: PoO (2003); number of people per boat: Moonen (1992)
- sailing boats > 5 meters	15,123	Number of boats: PoO (2003); number of people per boat: Moonen (1992)
- speedboats, canoes	3,674	Number of boats: PoO (2003); number of people per boat: Moonen (1992)
- total boating	100,962	
Total	172,456	

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.

(iv) Habitat service. De Wieden is highly important for biodiversity conservation. 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 large subspecies of the large copper butterfly (*Lyacena dispar*). The otter, which became extinct in the Netherlands some 12 years ago, was reintroduced to the area in June 2002. The area is protected under national laws, is included in the EU habitat and birds directives, and was recently (November 2002) appointed a Ramsar site.

Step 3. Valuation of the ecosystem services of De Wieden. On the basis of existing data, and limited surveys, the four selected services have been valued in monetary terms, using revealed preference methods. Due to deficiencies in available data, different approaches have been used to assess the value generated by the four services, see table 4.4. For the two production services, and for the benefits of the recreation service accruing to the providers of recreation services (for instance hotels, or boat rental agencies), the net value added generated by the service is used as indicator of its value. To assess the value for visitors to De Wieden, the consumer surplus is used, calculated with the travel cost method. For the habitat service, payments to the NGO protecting and managing the site ("Natuurmonumenten") are used as an indication of the lower value of the willingness-to-pay of the Dutch public for the habitat service of De Wieden. This willingness-to-pay reflects the consumers' surplus of the service (Pearce and Turner, 1990; Carson et al., 1992). It is a lower value because some of the members may be willing to pay a larger sum if required to preserve De Wieden, and because there may be other people also willing to pay for the preservation of De Wieden.

Hence, our calculations are based on a significant simplification of the complex issue of ecosystem services valuation, involving the use of two conceptually different indicators of value: consumers' surplus and value added. The two indicators compare as follows. On the one hand, the surplus gained through the reed production and fisheries service, and the provision of recreational services, may be larger than indicated through the respective values added because not all utility gained by people working in these sectors will be reflected in their income. For example, fishermen may enjoy their profession and gain utility through the fishing activities themselves. On the other hand, the concept of value added does not account for the shadow costs of labour and capital. This aspect, *cp.,* causes the value added to be higher than the consumer surplus. The use of two different indicators restricts the possibilities to add the values of the services.

Table 4.4. The approaches used to assess the surplus generated by each service

Stakeholder	**Calculation method**	**Type of value indicator used**
Reed cutters	Net value added	Income generated
Professional fishermen	Net value added	Income generated
Recreation service - Value for visitors to De Wieden - Value for the providers of recreation services (e.g. hotel owners, boat rental agencies)	 - Travel cost method - Net value added	 - Consumer surplus - Income generated
Nature conservationists	Donations to the NGO protecting the site	The donations are a lower value of the consumers' surplus generated by the service

(i) Reed cutting. The total turn-over 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). It is 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 that the consumer surplus resulting from reed production is zero.

(ii) Fisheries. Total annual turnover of the fishery sector is estimated to be only around 215,000 euro (Klinge, 1999; De Bruijn et al., 2001; Van Dijk, 2003). Investments are small, and the value added is estimated at around 140,000 euro (De Bruijn et al., 2001; Van Dijk, 2003). As with reed cutting, it is assumed that the consumer surplus generated by the fisheries activities in De Wieden is zero.

(iii) Recreation. The value of the recreation service is estimated by summing the utility gained by visitors to the area and the net value added of the local recreation sector – insofar as dependent upon visitors to De Wieden. The utility of tourists visiting the site is assessed with the zonal travel cost method (Hanley and Spash, 1993). The demand function for the site is constructed on the basis of the visit rate per zone and the travel costs from each zone. Six zones have been defined at increasing distances from De Wieden. The visit rates per zone have been estimated on the basis of a survey among visitors to the area by Van Konijnenburg (1996). There are a lot of camp sites and holiday houses in the area and, to avoid a bias in the number of visitors per zone, only the travel costs of people visiting from their permanent residence were included (n=304). For each zone, the relative visit rates and the average travel cost from the middle of the zone were calculated (see table

4.5). The travel costs include the average transportation costs by car (euro 0.28/km), from Rietveld et al. (2000), and the time costs, based upon the average per person hourly wage rate, from CPB (2003).

Table 5. Travellers and travel costs to De Wieden from different zones

Zone	**Total visits/year**	**Zone population**	**Visits/ population**	**Average travel costs**
<5km	8,548	19,010	0.45	1.17
5-10 km	13,380	45,580	0.29	2.58
10-25 km	35,681	345,790	0.10	4.45
25-50 km	27,875	1,024,390	0.03	7.50
50-100 km	53,521	3,714,010	0.01	12.64
100-250 km	33,451	6,285,990	0.01	19.19
Total	172,456			

Following standard procedures in the zonal travel cost method (Hanley and Spash, 1993), the relation between travel costs and visits per capita per zone is analysed with regression analysis. Subsequently, the demand function for visits to the site is constructed, using the results of the regression analysis to estimate the number of visitors at different hypothetical entry fees (assuming that an entrance fee is viewed in the same way as travel costs by the visitors). The first point on the demand curve is the current amount of visitors to the site (at the moment, there is no entry fee). The other points are found by estimating the number of visitors for different hypothetical entrance fees. The results are presented in figure 4.4. The area under the demand curve, equalling the consumers' surplus, is around 880,000 euro (which equals around 5 euro per visit).

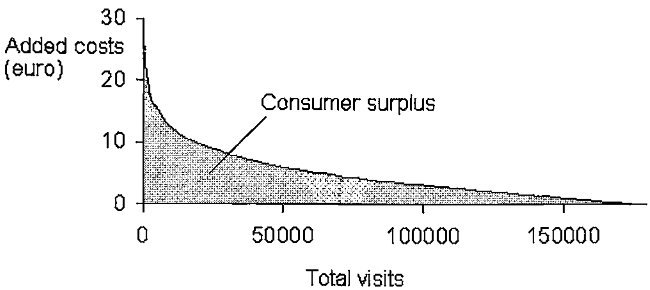


Figure 4.4. Demand curve for visits to De Wieden

The value added generated by the recreation sector is calculated as follows. The total *turn-over* of the recreation sector, as generated by visitors to De Wieden, is calculated by multiplying the number of visitors to De Wieden with their average expenditure. Visitors to the municipality (Steenwijkerland) spend, on average, euro 21.10 per day (TRN, 2002), and there are 172,456 visitor-days per year to De Wieden (this study). Hence, the total turn-over generated by De Wieden is euro 3,638,822 per year. It is assumed that all expenditure of these visitors can be attributed to the De Wieden (whereas in reality some of the visitors may combine a visit to De Wieden with a visit to another attraction in the area). The *net value added* is calculated by multiplying the turn-over with the average value added generated per unit of turn-over. For the recreational companies in De Wieden, the net value added is around 22% of turn-over (De Bruin et al., 2001) - which compares to a national average of 15% for the hotel sector (BHC, 2003). Hence, the value added generated by De Wieden is around euro 800,000 per year.

The total value of the recreational service of De Wieden is found by summing the utility accruing to the visitors, and the net value added of the recreational sector in the immediate surroundings of De Wieden, insofar as based upon the contribution from visitors to De Wieden. Hence, the total value of the service is 880,000 + 800,000 = 1,680,000 euro.

(iv) Habitat service. In order to obtain a crude approximation of the monetary value of the habitat service, it is assumed that the amount of money contributed to the NGO 'Natuurmonumenten' provides a lower value of the willingness-to-pay (WTP) of the Netherlands' public for the conservation of the sites this NGO manages. The actual value will be more because (i) members may be willing to pay more, if this would be required to protect the sites; and (ii) some non-members may also be willing to pay for the conservation of these sites. To estimate the WTP of the members of Natuurmonumenten for *De Wieden,* it is assumed that their WTP for De Wieden is proportional to the aerial surface of De Wieden in comparison to the total area of the sites managed by the NGO.

In the year 2002, the NGO received in total around euro 29 million in donations (Natuurmonumenten, 2003a). The total area of the sites managed by the NGO is 71,200 ha (June 2002), of which 5,400 ha (7.6%) are located in De Wieden (Natuurmonumenten, 2003a). Hence, the minimum value of the habitat service of the De Wieden wetlands can be estimated at around euro 2.2 million per year.

Step 4. Aggregation and comparison of the values. 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, as discussed earlier in the paper. Nevertheless, the values of the four services have been added to provide a crude indication of their total value, as presented in table 4.6. The approximate, combined monetary value of the four selected ecosystem services provided by De Wieden is in the order of euro 4,500,000 per year, or 830 per ha per year.

Table 4.6. 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
Habitat service	2,200,000
Total value of the selected services	4,500,000

Analysis of the scales of the ecosystem services provided by De Wieden

We now turn to the spatial scales at which the four services of the De Wieden ecosystem are supplied to stakeholders. Four institutional scales are distinguished: municipal (<5 kilometre from the Wieden), provincial (between 5 and 25 kilometre from De Wieden), national (the Netherlands excluding the area < 25km from De Wieden), and global (excluding the Netherlands).

Reed cutting. All reed cutters live in the proximity of the area (Van Dijk, 2003), and the benefits of reed cutting accrue to the stakeholders at the municipal scale.

Fisheries. All fishermen live in the proximity of the area (Van Dijk, 2003), and, as with reed cutting, the benefits of fisheries accrue at the municipal scale.

Recreation. Van Konijnenburg (1996) assessed the amount of visitors arriving from different areas to De Wieden, see table 4.7. The numbers do not match with table 4.5 as table 4.7 indicates the numbers of visitors from different institutional zones, instead of different distances. The international scale comprises all visitors from other countries; foreign visitors come in particular from Germany and Belgium. It is assumed that the WTP is the same for all visitors, and that this information can be used to assess the value of the recreation service at different scales. The recreational value for the tourism industry is entirely attributed to the municipal level as all companies are located in a distance of less than 5 kilometres from De Wieden.

Table 4.7. Visitors to De Wieden and consumers' surplus at different scales.

Scale	Share of visitors /1	Number of visitors	Consumers' surplus (euro) (rounded)
Municipal	9%	15,521	80,000
Provincial	25%	43,114	220,000
National	55%	94,851	480,000
International	11%	18,970	100,000
Total	100%	172,456	880,000

/1 source: Van Konijnenburg (1996)

Habitat service. The membership of Natuurmonumenten at different institutional scales provides a proxy for the value of the habitat service at these different levels, see table 4.8. Obviously, this approach to split the monetary value of the habitat service over different scales is very crude, for example because not all the appreciation of local people for the nature close to their home is reflected in a membership of Natuurmonumenten. However, better data is currently not available to analyse the value of this service at different levels within the Netherlands. The De Wieden wetlands are also of international importance, evident from its qualification as a Ramsar site and its inclusion in the EU habitats and bird directives. This international importance is not reflected in any indicator that can be used to establish a monetary indication of this international value, and it is included *p.m.* in the valuation.

Table 4.8. Lower bound value of the habitat service at different scales

Scale	Share of members /1	Number of members	Minimum value (euro) (rounded)
Municipal	0.3 %	2,883	6,600
Provincial	6.6 %	63,426	145,200
National	93 %	894,691	2,048,200
International	-	-	*p.m.*
Total	100%	961,000	2,200,000

/1 source: Natuurmonumenten, 2003b

Figure 4.5 shows how the values of the four services are distributed over the four scales. The production services are attributed to the municipal scale, whereas the habitat and recreation service are spread according to the approach explained above. Obviously, the analysis is very crude and provides only an order of magnitude indication of the values at different scales. Nevertheless, the figure demonstrates how scale determines the value of the services for the stakeholders at the different levels. At the municipal scale, the most important stakeholder interests relate to recreation, reed cutting and fisheries. At the provincial scale, the main stakeholder interests are in recreation, whereas the habitat service is also important. At the national level, the habitat service is by far the most important service. The value of the habitat service at the global scale is not known.

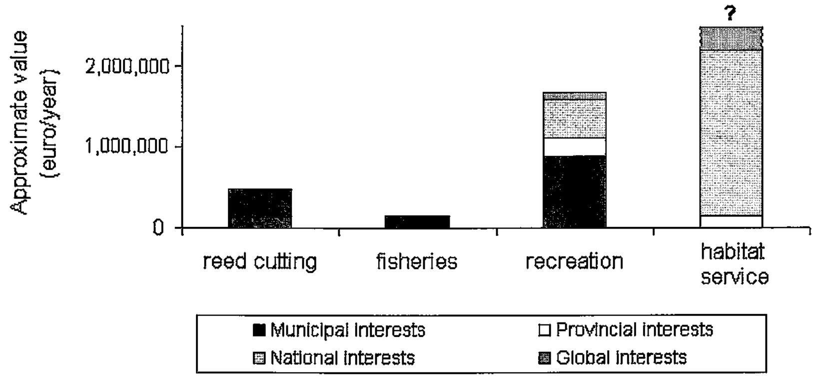


Figure 4.5. The relation between institutional scale and the value of ecosystem services (note that, at the municipal, provincial and national level, the value of the habitat service presents a lower boundary of the actual value; at the global level, the value of this service is not known)

4.5 Implications for ecosystem management

The ecosystem valuation framework

Contrary to Turner et al. (2000), but in line with De Groot et al. (2002) and Millennium Ecosystem Assessment (2003), we classify the different services provided by ecosystems in different categories (production, regulation and cultural services). In order to avoid double counting, we argue that regulation services included in the valuation should be limited to those services that (i) have an impact outside the ecosystem to be valued; and/or (ii) provide a direct benefit to people living in the area (*i.e.* not through sustaining or improving another service).

Scales of ecosystem services

Ecosystem services are generated at different, sometimes overlapping, spatial (and temporal) *ecological* scales. A service may depend upon a range of earth system processes taking place over various spatial scales. For example, carbon sequestration involves a range of processes taking place mostly at the scale of the plot (e.g. plant production) and the ecosystem (e.g. fire). Nevertheless, the service is generated at the global scale – it is the *global* amount of carbon sequestered that is one of the determining factors in the development of the world's climate. Some ecosystem services can also be supplied at different scales, for example the climate regulation service. Stands of forest affect the micro-climate of a particular area (temperature, humidity), whereas forests also influence climatic conditions at the scale of biomes (e.g. rainfall patterns) (Van Wijk et al., 2002; Durieux et al., 2003). Ecosystem services are supplied to the socio-economic system according to a range of *institutional* scales, varying from the individual to the global level (Grimble and Wellard, 1997; Tacconi, 2000). This is illustrated by the De Wieden example. In De Wieden, the stakeholders at the municipal level benefit in particular from the production services, and the local tourism sector. At the provincial level, the recreation services is most important. At the national and global levels, the conservation of nature and biodiversity is most important, whereas the production services are of little significance.

Accounting for scales in ecosystem management

Stakeholders at different scales often attach a different value to ecosystem services, depending upon the impact of the service on their income or living conditions, as well as their cultural background. These different interests often result in different visions on the management of the area (see also Brown, 1996 and Tacconi, 2000). This is confirmed by the stakeholders' preferences in De Wieden. Local stakeholders stress the importance of unrestricted access to the reed and fish resources of the area. Nature conservationists at the national and provincial level stress the biodiversity of De Wieden, and indicate that protection of this biodiversity requires certain restrictions on local activities. For example, reed cutters prefer to cut reed when it is one year old, but breeding birds need two to four year old reed for nesting.

If an optimal management strategy is sought on the basis of the interests of one particular scale alone, this may lead to unacceptable solutions for stakeholders at other scales. For example, a management plan for De Wieden based upon local interests only would probably not do justice to the substantial value of the biodiversity conservation service of De Wieden at the national and international scale. On the other side, a management plan for a natural park based upon national interests would leave little opportunity for local activities - and risk confrontation with local residents. In De Wieden, compromise solutions are found to balance the use of ecosystem services. For example, fishermen cooperate with nature conservationist by installing (subsidised) otter-protection devices on their hoop nets, and the nature conservation NGO managing the area poses relatively few restrictions on reed cutting in most of De Wieden.

In order to make informed choices on ecosystem management, it is necessary to assess the scales at which the relevant services are supplied. Consideration of the scales of ecosystem services enhances the possibility to use ecosystem valuation to support information of stakeholders, stakeholder negotiation, and consensus building during both policy formulation and implementation. Dividing a country in separate areas, and managing each area according to local preferences, will not lead to an optimal management from the national perspective. Accordingly, management of the globe according to national preferences risks to lead to sub-optimal management from the global perspective.

4.6 Conclusions

Ecosystem services are generated at a range of ecological scales, and are supplied to stakeholders at a range of institutional scales. Across the institutional scales, stakeholders can have very different perspectives on the values of ecosystem services, based, among others, on their dependency upon specific services to provide income or sustain their living environment. Therefore, it is crucial to consider the scales of ecosystem services when valuation of services is applied to support the formulation or implementation of ecosystem management plans. Formulation or implementation of management plans on the basis of stakeholders' interest at one institutional scale is bound to lead to sub-optimal ecosystem management from the perspective of stakeholders at other scales.

Analysis of the values of ecosystem services at different scales appears, in principle, feasible for the four services tested in this paper: two production services, recreation and nature conservation. However, the difficulties encountered in other studies in the monetary valuation of the non-use value of the habitat service (Spash and Hanley, 1995; Nunes and Van den Bergh, 2001) were confirmed in our study. Furthermore, it appears that monetary valuation of the habitat service at the *global* scale is particularly difficult as it is difficult to find a benchmark with which the habitat service can be compared. Further study is required in order to allow quantification of the global value of nature and biodiversity, and enhance its consideration in local and national ecosystem management.

5. Cost-efficient eutrophication control in a shallow lake ecosystem subject to two steady states

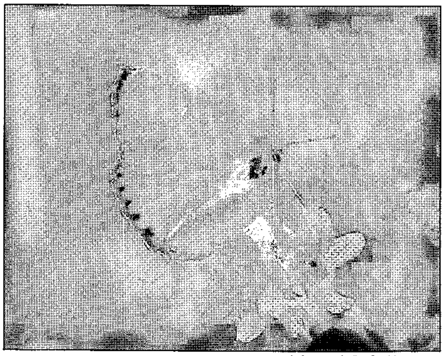


Photo: Mark Grutters

Adapted from: Hein, L.G., 2004. Cost-efficient eutrophication control in a shallow lake ecosystem subject to two steady states. Submitted.

5.1 Introduction

In recent years, the understanding of the complex interactions between ecosystems and the economic system has strongly increased (Gunderson et al., 1995; Perrings et al., 1995; Turner et al., 2003). It has become clear that efficient and sustainable ecosystem management requires consideration of the complex dynamics of ecosystems, including such issues as alternate states, thresholds and irreversible changes in ecosystems (Carpenter et al., 1999; Scheffer et al., 2001; Holling et al., 2002). For several types of ecosystems, general models have been constructed that explain the ecosystem's dynamics and complexities (Scheffer et al., 2001). However, to date, few studies have been conducted that analysed the implications of complex dynamics for the economic efficient management of an ecosystem supplying multiple services (Turner et al., 2003).

The aim of this study is to examine the implications of steady states and thresholds for the formulation of an economically efficient management strategy for a shallow lake ecosystem. The study is based upon an ecological-economic model constructed for the De Wieden wetland, the Netherlands. The wetland contains four interconnected shallow lakes, which are subject to hysteresis. Since the 1960s, the lakes have been subject to heavy nutrient loading, which caused a bifurcation from the original clear water state to the current turbid water state. The study examines the efficiency of various strategies to control eutrophication and rehabilitate the ecosystem to a clear water state.

In particular, the ecological-economic model analyses the ecosystem's response to a reduction in nutrient loading, without and with biomanipulation. It also allows for comparison of the costs and benefits of eutrophication control. The benefits of eutrophication control relate to an increased supply of ecosystem services following a change to clear water in the lakes. These benefits are not calculated, but the model presents the net benefits of eutrophication control for a range of assumed values of the annual benefits of a switch to clear water. Specific attention is paid to uncertainty in the model.

The water quality data used in this paper is extracted from an extensive database of the local waterboard (Waterboard Reest and Wieden, 2003). The De Wieden lakes have been selected because there is relatively good understanding of the dynamics of shallow lake ecosystems (Jeppesen et al., 1990; Scheffer, 1998; Mäler, 2000; Van Nes et al., 2002) and because of the availability of a large amount of water quality data for De Wieden.

The chapter is organized as follows. Section 5.2 describes the theoretical background and the modelling framework. Section 5.3 describes the study area including the ecosystem services it provides. Section 5.4 presents the ecological-economic model. Section 5.5 presents the results, for eutrophication control without and with biomanipulation. Sections 5.6 and 5.7 present a discussion of the implications of the study, and the main conclusions, respectively.

5.2 Theoretical background and modelling framework

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; Scheffer, 1998; Van Nes et al., 2002). The two states represent alternative equilibriums that exist over a certain range of nutrient conditions (Scheffer, 1998). 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 (i) stabilizing lake sediments; (ii) providing a shelter to daphnia (waterflees) that graze upon the phytoplankton; and (iii) 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 (Scheffer, 1998) - see figure 5.1.

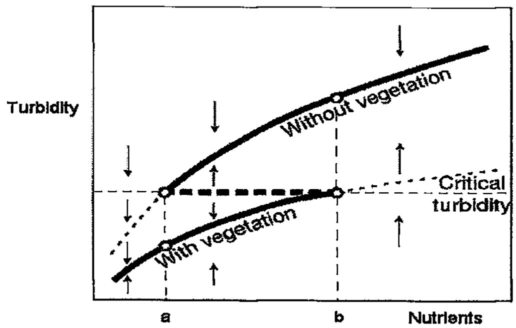


Figure 5.1. Hysteresis in shallow lake ecosystems. Source: Scheffer, 1998

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 (i) reduction of the nutrient loading to point a in figure 5.1; or (ii) partial reduction of nutrient concentrations to below point b in figure 5.1 in combination with biomanipulation (Scheffer, 1998; Meijer, 2000). Biomanipulation involves 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).

5.2.2 The modelling framework

The costs and benefits of achieving a transition to a clear water state are compared according to the framework presented in figure 5.2. The costs relate to the eutrophication control measures, which include measures to control the inflow of nutrients and biomanipulation. The benefits of eutrophication control stem from an increase in the supply of ecosystem services that depend upon water quality. For example, recreation may benefit from enhanced transparency of the water. Application of the framework involves analysis of the costs of eutrophication control measures, the response of the ecosystem to these measures; and the benefits of a switch to clear water.

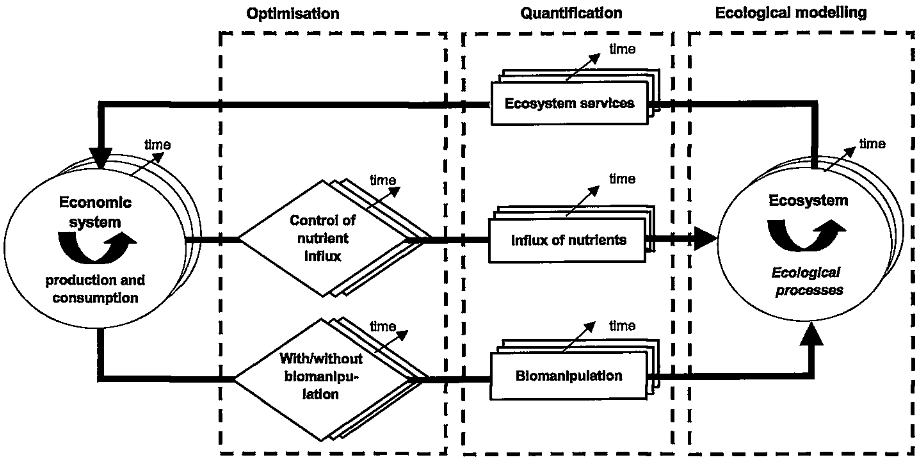


Figure 5.2. Framework for the analysis of the costs and benefits of eutrophication management in De Wieden. The benefits include an increased supply of ecosystem services following eutrophication control, the costs relate to the costs of the eutrophication control measures and/or biomanipulation.

This paper will analyse only the shift from the current, turbid water system to a clear water system. Efficient management in case the initial situation is a clear water system is not studied. The paper also does not analyse the implications of cyanobacteria, such as *Oscillatoria,* that occur under specific conditions in shallow lakes, and that can be a great nuisance to swimmers (Hosper, 1997; Scheffer, 1998). Nevertheless, it is expected that a shift to a clear water system will also reduce the occurrence of these blooms (Reeders et al., 1998; Scheffer, 1998).

5.3 Case study area

The setting

The De Wieden wetland is located in the North-eastern part of the Netherlands (52°42'N; 06°03'E). The area includes numerous lakes and canals as well as marshlands, extensive agricultural land and forests. 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 19th century. For the purpose of this study, the central part of De Wieden has been selected. The study area consists of the four biggest lakes of De Wieden (see table 5.1). The lakes are located in close proximity to each other and there is frequent exchange of water between them.

Table 5.1. General characteristics of the four lakes in the study area.

Lake	Area (ha)	Average depth (meter)
Beulakkerwijde	964	1.83
Belterwijde West	292	1.75
Belterwijde East	243	2.01
Schutsloterwijde	141	1.38
Total	1640	1.82

The four lakes receive water from a number of streams and canals flowing into the area, and through precipitation on the lakes. The most important source of water is the canal 'Steenwijk-Beulakkerwijde' which is fed by two rivers coming from the northeast of the area. This canal also receives (nutrient rich) excess water from a number of surrounding polders and the effluent of a sewage treatment plant. During the winter months, and most of the summer, the inflow of water in the lakes is much larger than the loss of water through evaporation and seepage to the groundwater. Under these circumstances, water is discharged to lake 'Zwartewater', the main outlet of the four lakes. However, in dry summers, water is occasionally let in from lake 'Zwartewater' in order to maintain the water levels.

Water quality

Up to the 1960's, the lakes were oligotrophic and the transparency was over 2 meters, 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 the input of nutrients in the area, which caused major ecological changes in the lakes. The original burbot (*Lota lota*) – 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 decreased and the water quality gradually improved.

Currently, the summer averages for total N and total P are around 2 mg/l and 0.1 mg/l respectively, and average transparency has increased to some 40 cm. Nevertheless, the water is still eutrophic, turbid, and the fish community remains dominated by bream. The regional waterboard is considering measures to further improve the water quality in order to reach the provincial water quality targets. For De Wieden, these targets are 1 mg total N/l and 0.05 mg total P/l (summer values) (PoO, 2001).

In the four lakes, phosphorus is the main limiting nutrient (Van Berkum, 2000; Waterschap Groot Salland, 2000). A phosphorus budget for the four lakes is shown in table 5.2. The net absorption of phosphorus in the lake bottom is estimated to be some 3.6 ton per year (Van Berkum, 2000). The other terms of the budget are calculated by multiplying water flows with the total phosphorus (total-P) concentrations in these flows. The balance is calibrated through the factor 'inflow of

P through streams and canals', for which no precise data on flows and total P concentrations were available.

Table 5.2. P-balance (ton P), average for the period 1998-2002

Flows of total-P (ton)	summer	winter	year
IN			
Inflow canals and streams	2.7	11.3	14.0
Recreation	0.5	0.1	0.6
Inflow from lake 'Zwartewater'	0.2	0.0	0.2
TOTAL IN	3.4	11.4	14.8
OUT			
Outflow to lake 'Zwartewater'	1.8	9.3	11.1
Net sedimentation	1.6	2.0	3.6
Outflow to polders	0.1	0	0.1
TOTAL OUT	3.5	11.3	14.8

Source: this study.

Ecosystem services supplied by the four lakes

The two most important ecosystem services provided by the lakes are nature conservation and recreation. The ecosystem also provides opportunities for reed cutting and fisheries, two services that are of local importance. De Wieden is one of the largest remaining peat wetland of western Europe and harbours a rich biodiversity (Natuurmonumenten, 2000). It provides a habitat to a wide range of water- and meadow-birds, fish and dragonflies, and contains, together with the adjacent wetland 'De Weerribben', the world's only population of a large subspecies of the Large copper butterfly (*Lyacena dispar*). De Wieden is also an important area for recreation, with the total number of visitors to the area estimated at around 170,000 per year. People visit De Wieden for boating, sailing, hiking, cycling, fishing, canoeing, surfing, swimming and/or sun-bathing (Van Konijnenburg, 1996). Reed cutting (for thatched roofs) and fisheries provide a source of income for in total some 220 people (Klinge, 1999; Natuurmonumenten, 2000).

5.4 The ecological-economic model

Introduction

The ecological-economic model developed in this paper describes the response of the ecosystem to eutrophication control measures. Total-P concentrations are used as the control variable of the models because P is the main limiting nutrient in the lakes (Van Berkum, 2000; Waterschap Groot Salland, 2000), en because reduction of nitrogen only may enhance blooms of certain *Oscillatoria* species that are able to fix atmospheric nitrogen (Hosper, 1997; Van der Molen et al., 1998; Scheffer, 1998).

The model includes the four lakes of the central part of De Wieden. It is assumed that the exchange of water between the lakes is sufficient for the inflow of phosphorus to be spread uniformly among the lakes. However, the study assesses the costs and benefits of achieving clear water *in three of the lakes* only. The Beulakkerwijde is by far the biggest lake (see table 5.1), and, through it's south-west orientation in line with the dominant wind direction in the Netherlands, it is particularly subject to resuspension of sediments through wind action. In addition, a relatively large proportion of the lake is deeper than 1.50 meter. Therefore, achieving clear water in De Beulakkerwijde would require a substantially higher reduction of total-P concentrations as compared to the three other lakes. Hence, it is chosen to analyse the costs and benefits of the potentially more feasible option of achieving clear water in the other three lakes.

The various steps included in the model are presented in figure 5.3. The model contains two approaches to eutrophication control, respectively without and with biomanipulation. This is expressed through a modified relation between P-loading and algae growth that reflects the different ecological processes that occur after biomanipulation (such as the enhanced grazing of algae by waterflees). Application of biomanipulation is also reflected in the costs of the approaches. The model follows a *steady state* approach up to the part 'macrophyte growth'. If macrophyte growth passes a threshold, it is assumed that the whole lake is converted to a clear water state in a period of 5 years – conform current experiences with large Dutch lakes (Meijer, 2000). The benefits of this transition are expressed as net present value (NPV) in order to compare them with the costs of eutrophication control measures.

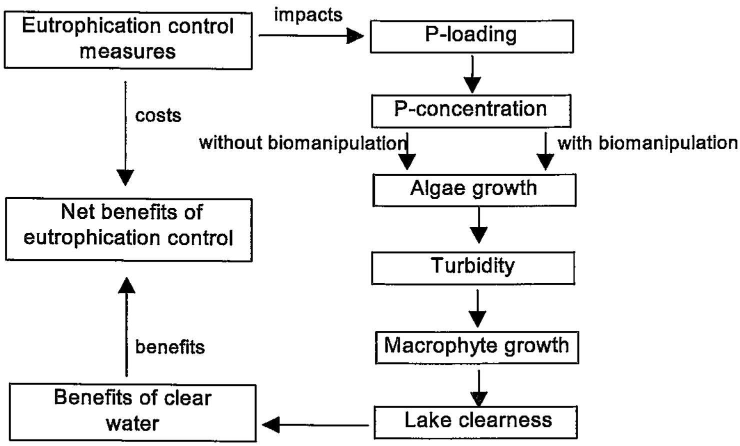


Figure 5.3. Model lay out (see text for explanation).

The three main steps of the model deal with: (i) the costs and impacts of mitigation measures; (ii) the modelling of the response of the ecosystem to eutrophication control measures; and (iii) analysis of the benefits of clear water. These three steps are described below.

Costs and impacts of mitigation measures

Potential measures available to reduce the inflow of phosphorus in the De Wieden wetland have been examined by the local waterboard (Van Berkum, 2000). In collaboration with the main stakeholders in the area (nature conservationists, farmers, 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 5.3. The measures have been ranked according to their cost-effectiveness.

Impacts of the measures are conform Van Berkum (2000). They represent the impact of the measures on the inflow of total phosphorus (total-P) into the four lakes. The costs have been recalculated on the basis of Van Berkum (2000) and 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. All measures have an impact upon the inflow of phosphorus through canals and streams, except measure 5 that addresses the supply of phosphorus through recreation. The measures are independent of each other, and their impacts are additive (Van Berkum, 2000).

Table 3. List of measures available to control the inflow of phosphorus.

Measure	**P-reduction (ton/year)**	**Costs (NPV) (mln euro)**	**Cost-effectiveness (mln euro/ton P)**
1. Diverting eutrophic polder water	4.2	8	1.90
2. Enhancing sewage treatment plant Steenwijk	0.8	2	2.50
3. Phosphorus reduction inflowing surface water	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. Reduction sewage spill-over	0.1	1.5	15.00
Total	10.7		

Source: Adapted from Van Berkum, 2000.

The measures in table 5.3 have been plotted in order to obtain an approximate cost curve (figure 5.4). 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 measure, diversion of eutrophic polder water, can be implemented for one up to in total 4 surrounding polders.

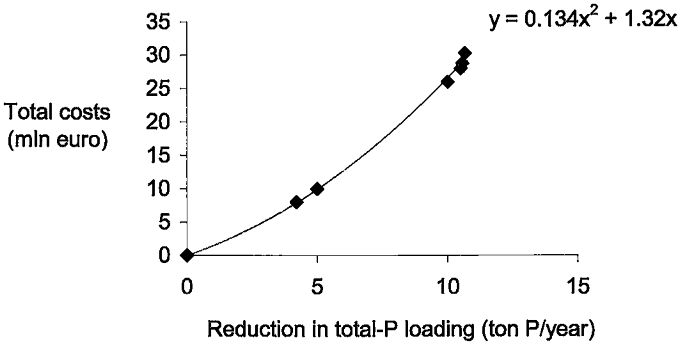


Figure 5.4. Total discounted cost curve for P control measures

Without biomanipulation. All costs in this approach reflect the costs of measures aimed at reducing the inflow of total-P, as plotted in figure 5.4. The corresponding cost function is:

$$TC = 0.134\ \Delta L^2 + 1.32\ \Delta L \quad (n{=}6;\ r^2{=}0.997;\ t(1^{st}\ \text{coeff.}){=}\ 7.3;\ t(2^{nd}\ \text{coeff.}) = 7.1) \qquad (1)$$

with:
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.

With biomanipulation. In case biomanipulation is applied, the costs of biomanipulation have to be accounted for. In general, these costs depend upon the area of the lakes to be treated. The three lakes in which biomanipulation would be applied have a total area of 676 ha. The corresponding per hectare costs of biomanipulation were, in 1992, estimated at 450 euro/ha (Hosper et al., 1992). Corrected for a 2% price increase per year, the total 2003 costs of biomanipulation are estimated at 380,000 euro, occurring in year 1. The cost function becomes:

$$TC = 0.134\ \Delta L^2 + 1.32\ \Delta L + CB$$

with:
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).

Modelling the response of the ecosystem to eutrophication control measures

From P loading to P concentration. The relationship between P loading and P concentration has been modelled, using a steady-state approach, since the late 1960s (Vollenweider, 1968). 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 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 (Hosper, 1997, Scheffer, 1998). Using data for 14 Dutch shallow lakes, Hosper (1997) updated the original Vollenweider equations and provided the following equation :

$$P = L\ .\ (0.201 \log (z/\tau) + 0.322)\ /\ (z/\tau) \qquad (n{=}63;\ r^2{=}0.72,\ t{=}1.7) \qquad (2)$$

with:
P = Phosphorus concentration (mg/l)
L = Loading (g P /m2 / 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* is 1.82 meter, and *τ* is 0.43 (Van Berkum, 2000). Including these numbers in equation (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 (2) also provides a valid equation to model the relation between P loading and P concentration in De Wieden.

Furthermore, 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: (i) 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 (ii) the water column of De Wieden is very rich in iron (concentrations range from 0.4 to 2 mg/l), which enhances the immobilization 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 the indicator for algae biomass. A positive correlation can be expected between the total concentration of the most limiting nutrient, and the chlorophyll a concentrations (Vollenweider, 1968; Dillon and Rigler, 1974). When P loading is substantially reduced, it can be assumed that P becomes the main limiting nutrient. 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 (Scheffer, 1998). 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 is partly driven by the release of phosphorus from the sediment as a consequence of algae growth, and 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.

(i) 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 5.5. The unexplained variation is caused by variations in turbulence, light regime and grazing (Scheffer, 1998).

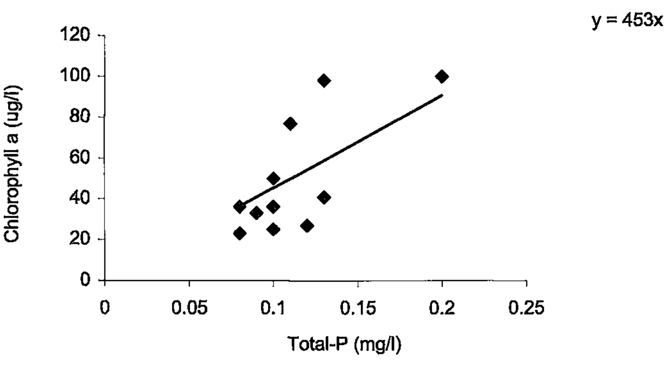


Figure 5.5. Relation between total-P and chlorophyll a

A linear relation between algae growth and total-P concentrations can be expected (Hosper, 1997). The trend line inserted in figure 5.5 is used as the function describing the development of chlorophyll a as a function of P concentrations. The equation used in the model is:

$$Ch = 453 \, . \, P \qquad (n=12;\ r^2 = 0.88;\ t= 8.6) \qquad (3a)$$

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

(ii) 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, the Duinigermeer lake is used to establish a new relation between these two factors. Lake Duinigermeer is also located in De Wieden. 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 5.6 shows the chlorophyll a concentrations of the lake Duinigermeer as a function of total-P concentrations, and a trend line. Because of the more limited availability of data, we use all available samples instead of yearly averages. However, all samples that are potentially nitrogen limited (with a N:P ratio below 20) have been excluded from the analysis.

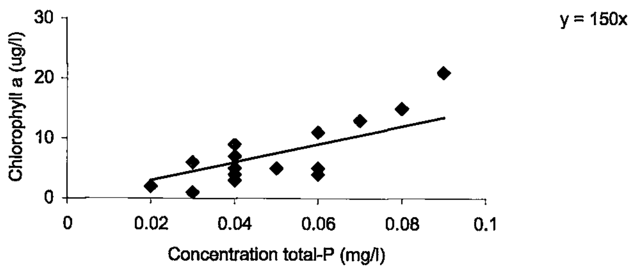


Figure 5.6. The relation between total-P and chlorophyll a for the Duinigermeer.

For the approach with biomanipulation, equation 3, which describes the relation between total-P and chlorophyll a, becomes:

$$Ch = 150 \, . \, P \qquad (n= 15;\ r^2 = 0.92,\ t=10.2) \qquad (3b)$$

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

From algae concentration to transparency. Transparency is generally measured in terms of Secchi depth (Scheffer, 1998). Secchi depth samples are available for the period 1991 to 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, upon 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 variations. Therefore, the equation is based upon the yearly averages of algae concentration and inverse Secchi depth (figure 5.7). A linear relation between chlorophyll a concentrations and the inverse of the Secchi depth is assumed (cf. Hosper, 1997).

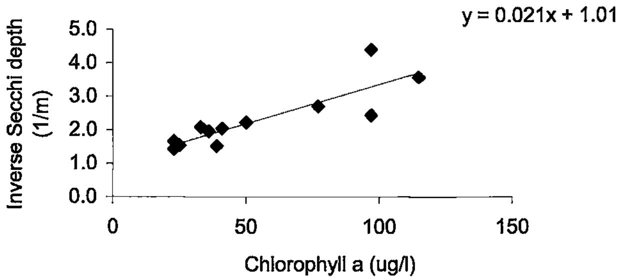


Figure 5.7. Relation between Chlorophyll a and the inverse Secchi depth.

The *yearly average* inverse Secchi depth can be reasonably well explained through the algae concentrations:

$$SD = 1 / (0.021 \,.\, Ch + 1.01) \qquad (n=12;\ r^2 = 0.74;\ t(\text{coeff})=5.2\ ;\ t(\text{constant})= 3.6) \qquad (4)$$

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

From transparency to the depth at which plants can grow. The Secchi depth is converted to the depth at which sufficient light penetrates to allow growth of charophyte waterplants. 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):

$$E(z) = E(0) \,.\, e^{-K \,.\, z} \quad \text{or: } \ln E(z) / E(0) = - K \,.\, z$$

with:
E(z), E(0) = the values of downward irradiance 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):

$$K = PA / SD$$

with:
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). Hence, at depth z_{ch}, at which 10% of the surface light remains, ln {E(z)/E(0)} = -2.3, and:

$$Z(_{ch}) = 1.53 \,.\, SD \tag{5}$$

with:
$Z(_{ch})$ = the maximum depth at which charophytes can develop (m)
SD = Secchi depth (m).

Estimation of the percentage of the lake with clear water. The part of the lake that can be covered with plants at a given light penetration depends upon the depth profile of the lakes. The depth profile of the lakes of De Wieden has been studied by Van Berkum (2000). As mentioned earlier, only three lakes of De Wieden are considered in this step. The depth profile of the three lakes has been plotted in a scatter diagram (figure 5.8), and an S-curve has been fitted to the profile.

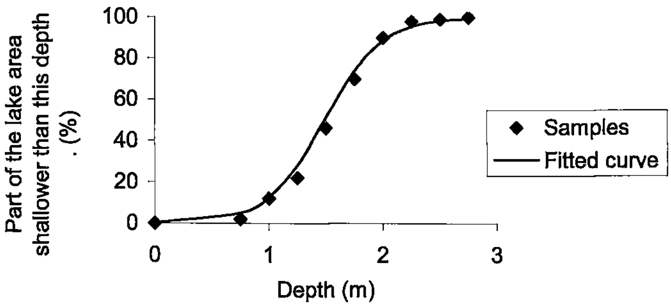


Figure 5.8. Relation between depth and part of the lake (three lakes).

The fitted curve indicates how much of the lake bottom will be covered by waterplants at a certain light penetration. The curve has the following equation:

$$PL = 100 \,/\, (1 + 375.e^{-4 \cdot z_{ch}}) \qquad \text{(n=10; F=21; } r^2\text{=0.98)} \tag{6}$$

with:
PL = part of the lake covered in year 1; and
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 threshold (corresponding to the critical turbidity in figure 5.1), it can be expected that 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 upon 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 I will use the 25% threshold, providing extra analysis for the implications of using a threshold of 20% and 30%. In the model, it is assumed that the establishment of charophyte waterplants in the whole lake would take five years, and would proceed in line with the development of vegetation in lake Wolderwijd. If the threshold is not passed, charophyte cover is assumed to remain constant. A logistic 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:

$$PL(t) = 81 / (1+70 * 2.71^{-1.5*t}) + 20.1 \qquad (7)$$

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

Analysing the benefits of a switch to clear water

It is expected that only the two services nature conservation and recreation will benefit from a switch to clear water in De Wieden. Regarding nature conservation, a range of threatened species is expected to benefit from a switch to clear water, and there would be no rare or threatened species that can be expected to decline from such a shift (see Annex 5.1). As for the recreation service, especially swimmers 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). The establishment of water plants may, however, reduce the possibilities for boats to explore the shallowest parts of the lakes, unless the water plants are mown. As the lakes contain substantial areas with deep water where water plants will not hamper boats, this effect is not further taken into account in the study. 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 (i) the level of eutrophication control and the type of measures implemented (without or with biomanipulation); and (ii) the assumed value of the marginal 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*) of the eutrophication control measures are expressed as NPV (discounted over 25 years, using a 5% discount rate). The annual benefits are discounted with a discount factor. It is assumed that the benefits increase proportionally with the percentage of the lake that has clear water (which may vary over time).

$$U = \sum_{t=0}^{25} 1/(1+r)^t \,.\, (TB * PL_{(t)}) - TC \tag{8}$$

with:

$U =$ Net benefits (utility) of the eutrophication control measures (euro)
$r =$ Discount rate at time t (5%)
$TB =$ Annual marginal 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) – this includes the costs of the measures that control the influx of total-P respectively without and with the costs of biomanipulation.

5.5 Comparison of the costs and benefits of eutrophication control measures

Introduction

The ecological-economic model described in section 4 calculates the net benefits of achieving clear water in three of the four big lakes of De Wieden. The costs of eutrophication control relate to the costs of the P control measures and, in the second approach, 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 these benefits are difficult to value in monetary terms, the net benefits of eutrophication control are calculated for a *range of assumed values* of the annual benefits of a switch to clear water. In the model, these annual benefits are converted to a NPV in order to compare them with the costs of the measures.

Efficiency of a reduction in P-loading – without biomanipulation

Figure 5.9 shows the economic efficiency of reducing the inflow of P in De Wieden without biomanipulation. Figure 5.9 has three axes, representing (i) the pursued reduction in P inflow; (ii) the annual value attributed to a switch to clear water; and (iii) the net benefits (expressed as NPV) as a function of the two previous variables. For instance, the line in the lower left part of figure 5.9 represents the NPV of different reductions in P loading when the annual benefits of a switch to clear water are assumed to be worth 2.5 million euro per year. In this case, there is a bimodal distribution; there is a local maximum efficiency at zero reduction in P-loading, and a – higher – maximum for a reduction in P-loading of 10 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. This P concentration is 0.03 mg/l. If the annual marginal benefits provided by clear water (through enhanced biodiversity protection and better opportunities for recreation) are valued at at least 2 million euro, it is economically efficient to reduce the inflow of total-P with 10 ton/year in order to obtain clear water.

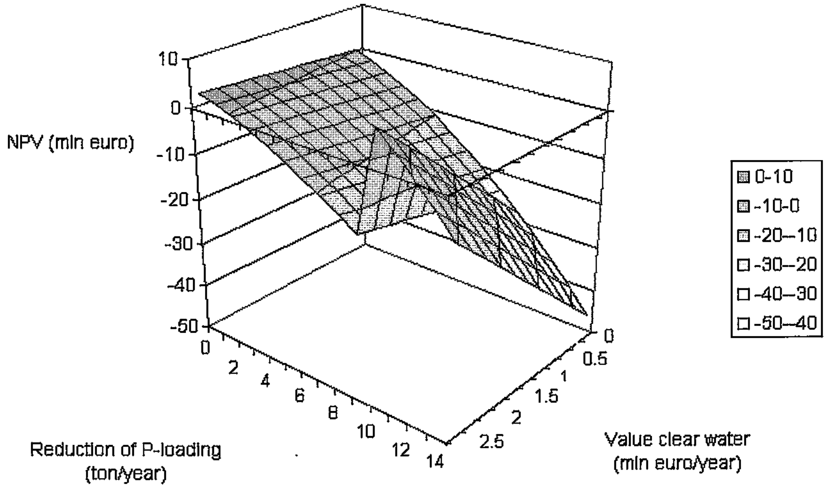


Figure 5.9. Efficiency of reducing P loading at different values of a switch to clear water – without biomanipulation.

Efficiency of 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). The model calculations show that, for the De Wieden lakes, a reduction in P-loading is required to reach the assumed threshold of 25% water plant cover following biomanipulation. The model indicates that, prior to the biomanipulation, P-loading should be reduced with 2 ton/year, bringing the P concentrations in De Wieden to 0.09 mg/l. Further reduction of total-P concentrations is not very useful, as the costs are high, and the clear water state can be achieved at this level through relatively cheap biomanipulation (figure 5.10). It becomes economically efficient to recuperate the clear water state of the three lakes if the marginal benefits from clear water are valued at least 0.21 million euro per year.

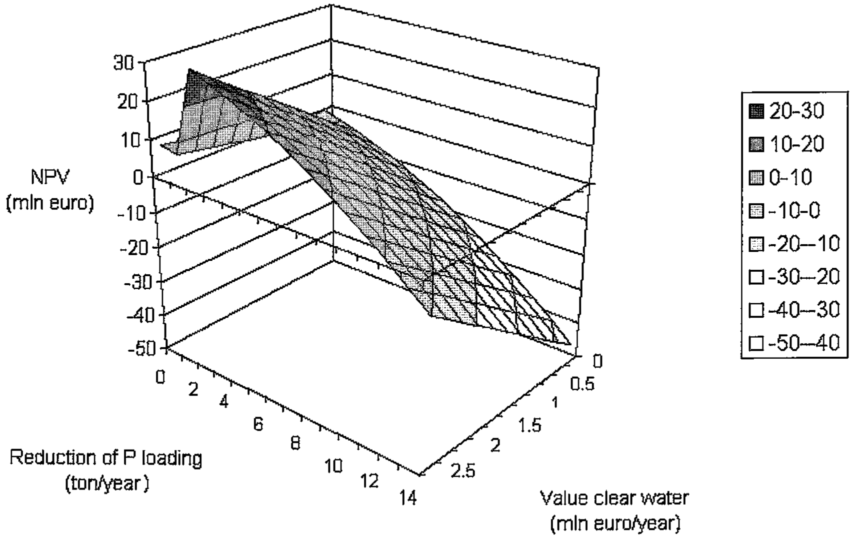


Figure 5.10. Efficiency of P loading reduction at different values of a switch to clear water – with biomanipulation.

5.6 Discussion

Uncertainties in the model

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

(i) 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.

(ii) 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 (PA = 1.5). The PA of shallow lakes in the Netherlands varies from around 1.0 to 2.1 (Hosper, 1997), and it is unsure how well the average PA of four nearby lakes of comparable size represents the PA of the De Wieden lakes.

(iii) 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 7). 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 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.

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	0.2	0.8

Clearly, the uncertainties of the analysis are substantial, with a main factor being the threshold level, and the model can not 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.

Implications for ecosystem management

Management of ecosystems subject to multiple steady states. The presence of alternative steady states has been recognized 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; and 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 is conform the current experiences with 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 e.g. 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 0.2 million euro per year to economically justify implementation of the eutrophication control measures. In comparison, 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). 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). In case of success, biomanipulation could be extended to the other lakes.

5.7 Conclusions

This paper shows how the response of a shallow lake ecosystem to reduced nutrient loading can be modelled in order to reveal the economic efficiency of different management strategies. 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 studied shallow lake ecosystem, 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 with 2 ton/year, in combination with 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 marginal benefits of an increased supply of ecosystem services following rehabilitation. The implementation eutrophication control measures that do not lead to a rehabilitation of the clear water state is not cost-effective.

The thesis also indicates that the current Dutch policies, as expressed in the 'Fourth National Policy Document on Water Management' (VW, 1998), are not 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/litre. 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, in case 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 cheaper 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, then through setting a standard for total-P concentrations only. In new water quality policies, the Dutch government could consider to increase the allowable total phosphorus concentration in waters with as 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 meter would allow the establishment of water plants up to a depth of around 1.5 meter, depending upon the lake characteristics (such as substrate and wind exposure). Considering that the large majority of Dutch lakes is very shallow, with average depths of 1.5 to 2 meter (Hosper, 1997), this norm would bring clear water in a substantial part of the Netherlands' lakes. This would have significant benefits for nature conservation and provide an additional benefit for swimmers, at lower costs then reaching the current 0.05 mg total-P/litre norm.

Annex 5.1 Biodiversity gains resulting from a shift to clear water in De Wieden

This annex lists the 'target species' occurring in the study area that would benefit from a transition to clear water (table A5.1). 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 in the Netherlands, including mammals, birds, reptiles, fish, insects, bivalvae, snails, worms, plants and mosses - of which 108 occur in De Wieden.

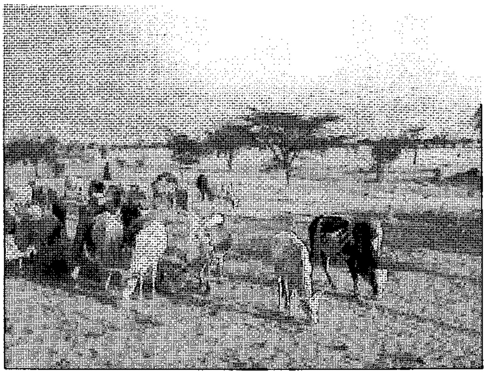
The species would benefit either directly from a shift to clear water, because their possibilities to forage or reproduce would increase, or indirectly, because they depend upon species that would benefit from a transition to clear water. There are no target species that are likely to decline following a switch to clear water in De Wieden. The assessment has been conducted for the four lakes, and their immediate surroundings, in total 5400 ha. It is based upon Natuurmonumenten (2000), the Netherlands Association for Dragonfly Studies (2002) and Noordhuis et al. (2002).

Table A5.1. 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
Burdot	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
-	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

6. The impact of grazing and rainfall variability on the productivity of a Sahelian rangeland



Adapted from Hein, L.G., 2004. The impact of grazing and rainfall variability on the productivity of a Sahelian rangeland. Submitted.

6.1 Introduction

Rangelands are the large areas 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, 1993; Walker and Abel, 2002). The understanding of rangeland dynamics has greatly increased in recent years (Westoby et al., 1989; Walker, 1993; Mortimore, 1998). Nevertheless, the impact of high grazing pressures on rangelands is still strongly debated (Briske et al., 2003). Some 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 ecosystem. In their 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, in particular 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). It has also been suggested that the impacts of a high grazing pressure may 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, 1989; Sullivan and Rohde, 2002; Briske et al., 2003). Clearly, the two approaches have very different implications for the optimal management of rangeland ecosystems.

The goal of this paper is to analyse the combined impacts of grazing and rainfall variability on the Ferlo rangeland ecosystem in Northern Senegal. In particular, I examine the impact of these two factors on the species composition of the herbaceous layer, the yearly biomass production, and on the rain-use efficiency of the ecosystem. The rain-use efficiency expresses the net primary production per mm of rainfall, and is an important indicator for the functioning of rangeland ecosystems (Snyman, 1998; Diouf and Lambin, 2001). Furthermore, the implications of the findings for efficient management of rangeland systems are discussed. The Ferlo has been selected as case study area because it provides a representative case of the larger Sahelian zone, and because there were sufficient data available to analyse the impact of different management strategies on the dynamics and productivity of the system.

The study is based upon an analysis of primary data of 10 years of grazing experiments conducted in the Ferlo, as reported in Klug (1982), Miehe (1992), Miehe (1997) and Andre (1998). The analysis re-examines the data reported in these studies with the perspective of the emerging thinking on rangeland dynamics (Walker, 1993; Ludwig et al., 2001; Walker and Abel, 2002). The dynamics of the shrub- and tree cover are not examined. It is anticipated that the effect of trees and shrubs on the grazing experiment is small, because the local tree and shrub cover is sparse (< 20 individuals per ha) and because it is dominated by small (up to 10 meter high), deep-rooting trees, in particular *Balanites aegyptiaca* (cf. Nizinski et al., 1994).

The chapter is organized as follows. In section 6.2, a brief review of the relevant theoretical background is provided. Subsequently, in section 6.3, the case study area and the methodology are described. This is followed by presentation of the results of the paper in section 6.4. In section 6.5, the implications for rangeland management are discussed. Finally, the main conclusion are presented in a section 6.6.

6.2 Theoretical background

General characteristics and processes of rangeland ecosystems.

Rangelands consist of a herbaceous layer, and an open bush and tree cover. Vegetation growth is constrained by water availability and, in some systems, nutrient availability (Breman and De Ridder, 1991). In the Sahel, water is the key limiting factor (Walter, 1971, Walker, 1981). Rangeland dynamics are influenced by three main processes (i) rainfall; (ii) grazing; and (iii) fire. These three processes are briefly described below; however, the implications of fire are not further analysed.

(i) Rainfall. Rainfall and soil texture determine the amount of water available to plants (Walker, 1981). A significant part of the rainfall may be lost through evaporation or run-off (Snyman, 1998). Characteristic of rangelands is the high annual variation in total rainfall and distribution patterns (Walker, 1981). This evokes an inter-annual variation in species composition and productivity of the rangeland (Le Houérou, 1984; Illius and O'Connor, 1999).

(ii) Grazing. Herbivore grazing influences the vegetation layer in several ways. A first impact is the enhanced recycling of nutrients as animals convert standing dead material to animal biomass and manure (Powell et al., 1986). Secondly, animals influence the soil structure. Whereas high stocking rates may lead to compaction, low stocking rates may be beneficial to soil structure by breaking crusts formed on the topsoil (Walker, 1981). Thirdly, herbivores control plant diversity through mechanisms that influence local plant colonisation and extinction dynamics (Olff and Ritchie, 1998). Although many species benefit from light grazing, favoured species may be overgrazed and disappear from the plant community at high stocking densities (Walker, 1981).

(iii) Fire. Fires are naturally occurring phenomena in rangelands and most plants are adapted to it (West, 1971). Fires are used by pastoralists to induce the regrowth of fresh leaves that provide valuable nourishment for livestock in the dry season. Although rangeland plant communities may be able to cope well with incidental fires, an excessive use of fire by pastoralists increases the pressure on woody species, which are relatively vulnerable to fire (Snyman, 2003), and may causes a loss of soil organic matter and a reduced fertility status of the soils (Walker, 1981).

The response of the ecosystem to these drivers depends upon its resilience and persistence for the drivers involved (Holling, 1973; Walker and Noy-Meir, 1982). Resilience is defined as the capacity of a system to withstand disturbances without changing into a different system (Holling, 1973). Resilience can take two forms. First, a system may resist change, for instance if cattle grazing has little impact on a rangeland. Second, a system may recover rapidly following a period of stress. For example, cattle may graze off the tops of tussock grasses, but if their crowns remain healthy, they will rapidly recover (Ludwig et al., 2001). The persistence of the ecosystem measures how much change a system can undergo without disturbing its structure and pattern of behaviour (Holling, 1992). The persistence of a rangeland may relate, for instance, to the amount of herbaceous biomass present in the system (Ludwig et al., 2001).

Developments in ecological models of rangeland succession

Early models of rangeland dynamics were based upon the Clementsian theory of ecological succession (Clements, 1916; Weaver and Clements, 1938; Tobey, 1981). These models assumed that any particular rangeland site 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 grazing, drought, fire or other disturbances. A given stocking rate will result in an equilibrium state of the vegetation (Walker, 1993).

In recent years, new insights in rangeland dynamics have emerged (Westoby et al., 1989; Laycock, 1991; Walker and Abel, 2002). This was initiated by the increased recognition that the early models were not capable of adequately predicting rangeland development following changes in management, and facilitated by the overall development of new concepts in ecology (Walker, 1993). It is now broadly accepted that various mechanisms influence rangeland dynamics, such as (i) multiple states and thresholds; (ii) irreversibility; and (iii) spatial variation at different scales. These three issues are analysed below.

(i) Multiple states and thresholds. It is increasingly recognized that rangelands can be subject to multiple steady states (Walker, 1993; Scheffer et al., 2001). Positive feedback mechanisms reinforce the system to be in a particular state. For instance, the existence of two steady states has been demonstrated in relation to a cover dominated by woody plants versus a cover dominated by grasses. A thick grass cover may restrict the establishment of tree seedlings and maintains domination of the plant cover by grasses, whereas in a cover dominated by trees, trees and tree seedlings can out-compete grasses in accessing sub-soil water and prevent the establishment of new grasses (Friedel, 1991). To shift from one state to the next, the system needs to pass a certain threshold in terms of tree cover (Friedel, 1991).

(ii) Irreversibility. The occurrence of irreversibility has been demonstrated for several rangelands. For example, large numbers of domestic livestock were introduced to North-American rangelands in the late 19th century. Initially, these rangelands consisted of a fairly open stand of unpalatable sagebrush (especially big sagebrush, *Artemisia tridentata*) with a productive herb layer consisting of grasses and forbs. Overgrazing led to degradation of the herb layer, and increased density of sagebrush. A subsequent reduction of grazing pressure does usually not lead to a reduction in sagebrush dominance, or recovery of the herb layer (Laycock, 1991). The possible irreversibility of species disappearance in the Sahel is still topic of discussion (Breman and De Ridder, 1991).

(iii) Spatial variation. Spatial variation occurs at different scales, and influences the dynamics of the rangeland (Walker, 1993). At the micro-scale, relief, soils and other geomorphologic characteristics influence the distribution of water through run-off and run-on, influencing the resilience of the system to climate variability and grazing pressure (Tongway and Ludwig, 1989; Van de Koppel et al., 2002). At higher scales, there exist patterns of more and less heavily grazed fields, for example at different distances from watering points (Walker, 1981). At the highest scales, rangelands will be subject to gradients in rainfall patterns (Walker, 1993).

These new insights have led to the development of a non-equilibrium approach for the analysis of rangeland dynamics (Briske et al., 2003). The non-equilibrium approach is able to account for state-and-transition dynamics as well irreversible responses (Briske et al., 2003). However, a main outstanding issue in the analysis of rangeland dynamics is the impact of grazing versus rainfall variability on the rangeland (Le Houérou, 1984; Ellis and Swift, 1988; Scoones, 1994; Fernandez-Gimenez et al., 1999; Illius and O'Connor, 1999; Sullivan and Rohde, 2002; Briske et al., 2003).

6.3 Area description and methodology

Area description

The study site is the Widou-Thiengoly catchment, located in the Western part of the Ferlo Region, Northern Senegal (15°20' N, 16°21' W, see figure 6.1). The site is part of the Sahel biogeographic zone, which extends from Senegal in the west to Somalia in the east. The Widou-Thiengoly area is generally flat, with a very modest, undulating, relief. Annual rainfall varies between around 120 and 450 mm, with an average of 290 mm (Andre, 1998). Rain falls predominantly during the short rainy season from July to September. The natural vegetation consists of dry grassland with scattered trees and bushes. The herbaceous layer comprises a mix of annual and perennial grasses, leguminous species and other plants. Most annual species have a relatively short life cycle, conducted for the largest part during the rainy season.

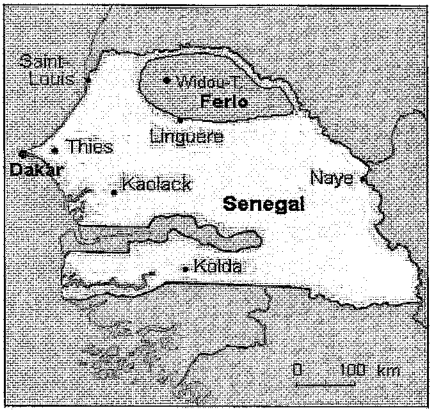


Figure 6.1. Location of the Widou-Thiengoly study site, Senegal.

Livestock keeping is the main economic activity in the Ferlo and is essential for local food security. The principal animals kept are cattle (Zebu), sheep and goats. Traditionally, the herds resided in the Ferlo during the wet season in order to take advantage of the feed resources. Various small, temporal ponds provided drinking water for the animals. In the dry season, these ponds disappeared and the feed resources declined. Consequently, the pastoralists migrated southwards to the more humid Sudan zone, where fallow lands and crop residues provided food for the animals, and where more perennial water resources are available (Sutter, 1987; Breman and De Ridder, 1991; Guerin et al., 1993).

Since the 1950s, population and livestock densities strongly increased due to improved medical and veterinary aid. At the same time, there has been a progressive sedentarisation of the pastoralists, caused by three main factors. First, a substantial number of wells were constructed in the Ferlo, drawing water from deep aquifers (Ministère de l'Hydraulique, 1987). Second, there was an expansion of agricultural activities in the Sudan zone, which limited the possibility for pastoralists to migrate south in the dry season (Sinclair and Fryxell, 1985; Guerin et al., 1993). Third, from the 1960s onward, the newly established, independent government stimulated people to settle in order to live within the boundaries of the country, and to facilitate their incorporation in the administrative system (Sinclair and Fryxell, 1985). The process of sedentarisation caused a concentration of grazing pressures, and led to the creation of denuded zones around the wells. Currently, average livestock densities in the Ferlo are in the order of 0.15-0.20 TLU/ha (De Leeuw and Tothill, 1990; Miehe, 1997). A TLU is a Tropical Livestock Unit, which 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). A lack of feed resources for the animals still evokes a southward migration in the dry season, albeit at a smaller scale than previous to the development of the wells (Guerin et al., 1993).

Methodology

This studies compares the impacts of two different grazing regimes in the study area on its species composition, biomass production and rain-use efficiency. The two grazing regimes are a medium, fixed maximum stocking of 0.10 TLU/ha, and variable, uncontrolled grazing, corresponding to an average stocking density of around 0.15-0.20 TLU/ha. The study is based upon data collected in a 10 years grazing experiment conducted in the Widou-Thiengoly site in the Ferlo, Senegal, as reported in Klug (1982), Miehe (1992), Miehe (1997) and Andre (1998). For this study, data reported in these four reports have been combined and re-analysed on the basis of new insights in rangeland dynamics. The methodology is further explained below for the three aspects analysed in this study: (i) species composition; (ii) biomass production; and (iii) rain-use efficiency.

(i) Species composition. Vegetation data (species composition, biomass) were recorded once a year in 10 1-ha sample plots. Five plots were located in intensively grazed plots, and five in plots with medium grazing. Species composition of the herbaceous layer in the plots was recorded according to Braun-Blanquet (Miehe, 1997). Both the herbaceous biodiversity per plot, and the biodiversity in the 5 plots together has been recorded. Subsequently, the frequency of occurrence of each species has been examined, using four classes: (i) one individual specimen; (ii) several specimen with less than 1% cover; (iii) 1 to 5 % cover; and (iv) 5-25% cover (Miehe, 1992; Miehe, 1997). This paper first analyses the impacts of different grazing regimes on herbaceous biodiversity. The impact on biodiversity is analysed by comparing the total number of herbaceous species occurring in the sample plots in the last year of the experiment, 1990, with a Student's t-test (following Blalock, 1987). Second, the impacts of grazing on the feed resources provided by the rangeland is examined. Specifically, this paper analyses the impact of grazing on the composition of the herbaceous species mix, including all species that occur in a cover of at least

1%, in at least one of the plots. Dry years, and years with normal or above average rainfall are analysed separately. Based upon long-term rainfall data, dry years are defined as having less than 150 mm effective rainfall per year, whereas years with normal or above average rainfall have at least 150 mm/year (following Andre, 1998). The impacts of the different grazing regimes on feed resources have been analysed, in a qualitative manner, through comparison of the species composition in dry and wet years, for medium and high grazing pressures. Furthermore, the selected species are classified in terms of family and forage quality. The forage quality of the different herbaceous species has been classified in five different categories expressing the relative palatability and the nutrient contents of the species. Classifications were conducted according to Boudet (1975), FAO (1990) and Breman and De Ridder (1991).

(ii) Biomass production. Biomass production is expressed as kg DW biomass ha^{-1} $year^{-1}$. The primary biomass production was measured as follows. In each 1-ha sample plot, the above ground biomass (peak standing crop) of all herbaceous species was collected in 25 separate one square meter plots. This was oven-dried and weighted (Miehe, 1992; Miehe, 1997; and Andre, 1998). Biomass was summed over the different plots and multiplied by 400 in order to obtain the per hectare biomass production. The one square meter plots were selected to be at least 15 meters away from a tree in order to avoid a bias in the result due to interference with the tree cover. The significance of the differences in biomass production under the two grazing regimes is tested with a matched-pairs t-test (Blalock, 1987).

(iii) Rain-use efficiency. Rain-use efficiency (kg dry weight biomass ha^{-1} mm^{-1}) is calculated by dividing primary biomass production with the annual effective rainfall. The effective rainfall is the amount of rain that is not lost through evaporation or run-off (Haan et al., 1994). Runoff depends upon the site-specific interception, infiltration and surface storage rates (Soil Conservation Service, 1972), and the effective rainfall needs to be calculated on the basis of local potential evaporation and runoff rates (Dunin, 1969; Mishra et al., 1999). Based upon measurements of run-off and infiltration rates, the effective rainfall in the Widou-Thiengoly study site has been calculated as follows (Haan et al., 1994). First, half of the potential daily evapotranspiration (derived from climate statistics) is deducted from the daily rainfall. Second, all rainfall above 12 mm/day is assumed to be lost through run-off and excluded (Miehe, 1992). The relation between rainfall, grazing and rain-use efficiency is analysed in two ways. First, a regression analysis is conducted in order to reveal the relation between rainfall and water-use efficiency for the two grazing regimes. Second, the statistical significance of differences in the rain-use efficiency between areas with high and medium grazing pressure is analysed for the whole period (1981-1990), as well as for dry and wet years separately, with a matched pairs test (Blalock, 1987).

6.4 Results

Impact on species composition

The *total* number of herbaceous species found in the sampling plots in 1990 is 47 in the intensively grazed plot, and 60 in the plots with medium grazing pressure.

Statistical analysis of the differences in biodiversity in the sample plots showed that these differences in biodiversity were not significant at the p=0.05 level (Annex 6.1). The impacts of grazing on the composition of species occurring in the Widou-Thiengoly study site with a cover of at least 1% is shown in table 6.1. Dry years (<150 mm effective rain) and years with normal or above average rainfall (at least 150 mm effective rain) are analysed separately. The table distinguishes annual grasses (A), perennial grasses (P), leguminous plants (L) and other plants (O). Leguminous and other plants may be either annual or perennial species. Species have been ranked in order of occurrence, with the most frequently occurring species listed highest in the lists. The list also indicates the approximate feed quality of the plants, ranked from 1 (highest) to 5 (lowest) – based upon Boudet (1975), FAO (1990) and Breman and De Ridder (1991).

Table 6.1. Species composition under different grazing regimes, for dry and wet years (see text for explanation). Sources: Boudet (1975), FAO (1990), Miehe (1992), Miehe (1997) and Andre (1998).

	High, variable grazing pressure (0.15-0.20 TLU/ha)			**Medium, fixed grazing pressure (0.10 TLU/ha)**		
	Grass species	Type	Quality	Grass species	Type	Quality
Normal/ wet year	*Aristida mutabilis*	A	3	*Schoenefeldia gracilis*	A	2
	Eragrostis cilianensis	A	5	*Zornia glochidiata*	L	1
	Zornia glochidiata	L	1	*Gisekia pharnaceoides*	O	2
	Chloris prieurii	A	1	*Indigofera aspera*	L	3
	Mollugo nudicaulus	O	3	*Monsonia senegalensis*	O	1
	Indigofera aspera	L	3	*Ceratotheca sesamoides*	O	1
	Eragrostis tremula	A	4	*Mollugo nudicaulus*	O	3
	Schoenefeldia gracilis	A	2	*Trianthema portulacastrum*	O	2
	Frimbistylis hispidula	O	4	*Alysicarpus ovalifolius*	L	1
	Cassia obtusifolia	L	5	*Aristida longiflora*	P	4
				Commelina forskalaei	O	2
				Echinochloa colonna	A	2
				Indigofera diphylla	L	2
				Latipes senegalensis	P	1
				Tephrosia purpurea	L	2
Dry year	*Tragus berteronianus*	A	5	*Schoenefeldia gracilis*	A	2
	Aristida mutabilis	A	3	*Aristida longiflora*	P	4
	Chloris prieurii	A	1	*Cenchrus biflorus*	A	3
	Tribulus terrestris	O	5	*Brachiaria xantholeuca*	A	3
	Dactyloctenium aegyptium	A	2	*Dactyloctenium aegyptiacum*	A	2
	Heliotropium bacciferum	O	5	*Trianthema portulacastrum*	O	2
				Crotolaria podocarpa	O	2
				Latipes senegalensis	P	2
				Digitaria horizontalis	A	3
				Tetrapogon cenchriformis	A	?
				Aristida adscensionis	A	5

Key: A: annual grasses; P: perennial grasses; L: leguminous plants; and O: other plants.

Table 6.1 shows that, in both wet and dry years, the species composition of plots with a high and medium grazing pressure is markedly different. The occurrence of most palatable species has decreased in the plots with high grazing pressures, with the notable exception of *Chloris prieurii*, a highly palatable, fast growing annual grass that benefits from reduced competition and locally higher availability of manure in situations with high grazing pressure. This species occurs in both wet and dry years, but it is reasonably drought resistance and it is more abundant in dry years. Furthermore, the quadrant representing drought and high grazing pressure has only 6 species occurring with a cover exceeding 1% in at least one plot in the herb layer, compared to 10 and 11 for the quadrant with high grazing and low rainfall, respectively medium grazing and drought. This indicates that the diversity of frequently occurring species decreases most in a situation with intensive grazing combined with drought. The amount of perennial grasses is reduced in the intensively grazed pastures – in both dry and wet years. Furthermore, leguminous species occur more frequently in years with normal or high rainfall – both in plots with a high and a medium grazing pressure.

Impact on biomass production

The production of herbaceous biomass in the plots with medium and high grazing pressure during the period 1981-1990 is shown in figure 6.2. The relative differences in biomass production are most pronounced in the driest years, 1983 and 1984. In these years, the plots with medium grazing pressure were still able to supply some biomass, 230 and 100 kg/ha, respectively. However, the heavily grazed plots supplied only around 80 kg/ha in 1983 and 0 kg/ha in 1984. It has been tested with a matched pairs test if the differences in biomass production between the plots with medium and high grazing pressure are statistically significant (Annex 6.1). The analysis shows that for the period 1981-1990, the difference in biomass production between the plots is significant at the p=0.05 level. It is also tested (Annex 6.1) if the difference is significant if only years with dry (1981-1984), respectively normal and above average rainfall (1985-1990) are considered. The tests show that these difference are significant at the p=0.05 level for the dry years, but not for the wet years. Hence, the experiment shows that a high grazing pressure has a significant impact on herbaceous biomass production in the Ferlo, and that this impact is concentrated in dry years.

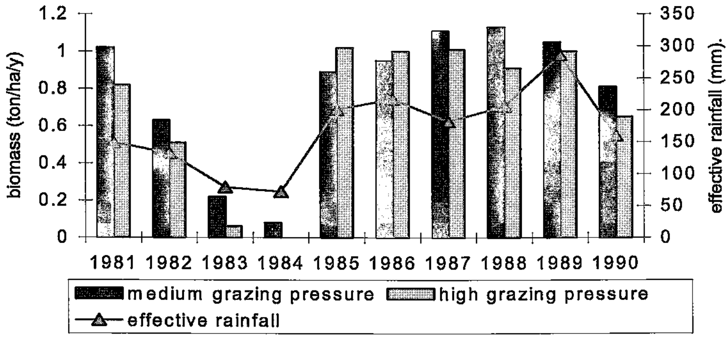


Figure 6.2. Biomass production in fields with medium (0.10 TLU/ha) and high (0.15-0.20 TLU/ha) grazing pressure. Data from Miehe (1992, 1997) and Andre (1998).

Impact on the rain-use efficiency

For the period 1981-1990, the effective rainfall appears to be, on average, around 65% of the total rain – which indicates that an average of 35% of the rain is lost through evaporation or run-off. 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 rain-use efficiency of the herb layer under both grazing regimes, plotted as a function of effective rainfall, is presented in figure 6.3. A second order polynomial has been fit through the observed values (cf. Le Houérou, 1984). Neither linear nor S-curve relations could be fitted with the measurements in a statistically significant way (they led to an R^2 of at most 0.24). The curve demonstrates that rain-use efficiency 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 * 290 = 189 mm per year.

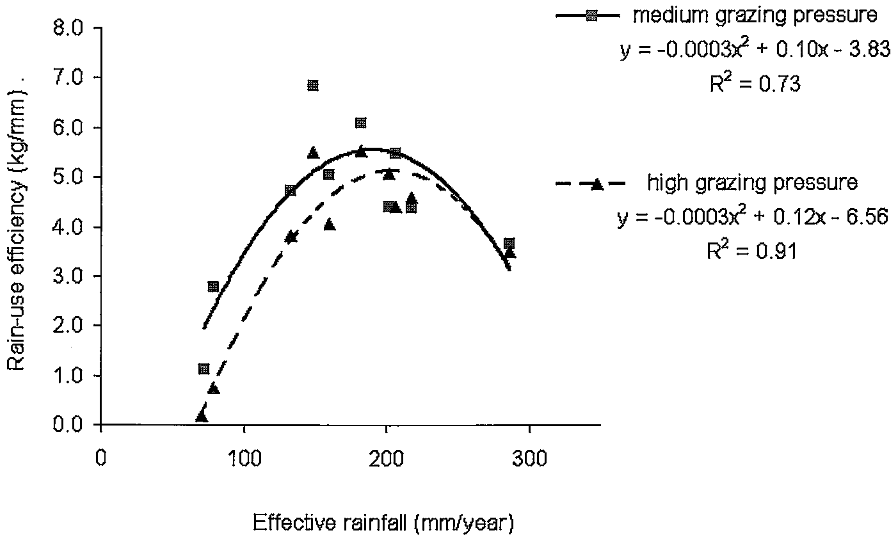


Figure 6.3. Rain-use efficiency as a function of effective rainfall, for sites with medium (0.10 TLU/ha) and high (0.15-0.20 TLU/ha) grazing pressure. Data from Miehe (1992, 1997) and Andre (1998).

Figure 6.3 shows that the rain-use efficiency of plots with medium and high grazing pressure does not markedly differ for *normal* and *wet* years – defined as having over 150 mm of effective rain (Andre, 1998). However, for the dry years, there is a substantial difference between the plots. These differences have also been tested with a matched-pairs test (Annex 6.1). This test shows that for the period 1981-1990, the difference in rain-use efficiency between the plots is significant at the p=0.05 level. This difference has also been examined for the dry and wet years separately (Annex 6.1). At the p=0.05 level, the difference in rain-use efficiency is significant in dry years (<150 mm), but not in normal and wet years ($\geq$ 150 mm).

6.5 Discussion

This paper examines how grazing and rainfall variability influence the productivity of the Ferlo rangeland in Northern Senegal, based upon data reported in Klug (1982), Miehe (1992, 1997) and Andre (1998). The results of the study are discussed below, with respect to (i) species composition; (ii) biomass production; and (iii) rain-use efficiency. Subsequently, the potential implications for remote sensing studies that analyse degradation processes in rangelands are examined.

(i) Species composition. The grazing experiment showed no significant difference in biodiversity between plots with medium (0.10 TLU/ha) and high (0.15-0.20 TLU/ha) grazed pressure. However, there were a number of marked differences in the occurrence of species with a cover of at least 1% in dry or wet years. The main differences in species composition concerned the occurrence of palatable versus unpalatable species and perennial versus annual grasses. Rangeland ecological theory predicts that the most palatable, and the least resistant species disappear due to preferential grazing in areas with high pressure (Walker, 1981). This is illustrated in the Widou-Thiengoly grazing experiment. Whereas some of the most palatable species decrease in abundance in the highly grazed pastures (e.g. *Monsonia senegalensis* and *Latipes senegalensis*), the highly palatable species *Chloris prieurii* is particularly abundant in the intensively grazed plots. Characteristic for this species is a very short lifecycle that enables it to persist under the current grazing pressures in the Ferlo (Miehe, 1997). The grazing experiment confirmed that perennial grasses (*Aristida longiflora, Latipes senegalensis*) have difficulties persisting in areas with high grazing pressure. This is related to the response of perennials to grazing, which triggers the plant to develop new leaves using underground biomass. Intensive grazing causes progressive resorbtion of the root biomass, which reduces the capacity of the roots to access soil moisture (Sinclair and Fryxell, 1985). The experiment also showed that leguminous species appear in smaller numbers in dry years. A possible explanation is that the biological nitrogen fixation process, which gives leguminous species a competitive advantage over other plants, is hampered by water stress (Chapman and Muchow, 1985; Ledgard and Steele, 1992).

(ii) Biomass production. The experiments demonstrate how biomass production varies with effective rainfall under different grazing pressures. High grazing pressures have a significant impact on herbaceous biomass production in the Ferlo. This confirms the statements of Le Houérou (1984), Sinclair and Fryxell (1985) and Illius and O'Connor (1999) who state that high grazing pressures affect the functioning and productivity of rangelands, in particular in the medium and long term. The impact of grazing is concentrated in dry years; in wet years or years with normal rainfall, there is no statistically significant reduction in biomass production as a function of effective rainfall. Note that if the relation between rainfall, grazing pressures and rangeland productivity in the Ferlo had been analysed for a period containing (almost) only years with average or above average rainfall, the data would show that there is no relation between grazing pressure and biomass production.

(iii) Rain-use efficiency. The analyses show that rain-use efficiency of vegetation varies considerably with the effective rainfall (cf. Le Houérou et al., 1988). For the grazing experiment conducted in the Ferlo (Miehe 1992, 1997), a parabola, second order equation best describes the relation between the two variables. The maximum of the curve corresponds very well to the average effective rainfall in the catchment over the last 50 years (189 mm/year), which indicates a high degree of adaptation of the vegetation to the local rainfall conditions. Note that correlation of the *total* annual rainfall and the rain-use efficiency did not reveal any statistically significant relation. This implies that, due to large interannual variations in the ratio 'effective annual rainfall' – 'total annual rainfall', it is necessary to use the *effective* rather than the total annual rainfall when analysing the impacts of climate variability on rangeland productivity (cf. Pilgrim and Cordery, 1992; Haan et al., 1994; Lewis et al., 2000). The paper also confirms that rain-use efficiency is an important indicator for the functioning of the ecosystem (cf. Snyman, 1998), as it determines the system's productivity, and because it is sensitive to both effective rainfall and grazing pressure.

The analysis of the species composition, biomass production, and rain-use efficiency provides some general insights in the impacts of a high grazing pressure on the productivity of the Ferlo. The Widou-Thiengoly grazing experiment shows that the composition of the dominant species mix in the Ferlo rangeland is determined by both the grazing pressure, and the effective annual rainfall. Although there are considerable differences in the herbaceous cover between plots with medium and high grazing pressure, the forage quality of the herb layer was not very different, due to the frequent occurrence of the highly palatable species *Chloris prieurii* in the intensively grazed plots. However, the current, high grazing pressure of 0.15-0.20 TLU/ha reduces the productivity of the Ferlo rangeland compared to a grazing regime based upon a medium stocking density of 0.10 TLU/ha. A high grazing pressure reduces the rain-use efficiency of the rangeland, in particular during droughts. In years with below average rainfall, biomass production is reduced both by the low availability of water to plants, and by a reduced efficiency of herbaceous plants to use the available water. Consequently, the impacts of high grazing pressures on the productivity of the Ferlo are hardly noticed during years with normal or above normal rainfall, but they are particularly strong during a drought. This mechanism may partly explain the devastating impact of droughts on the production of herbaceous biomass, and hence on livestock survival, as experienced in e.g. 1973 and 1984 (Sinclair and Fryxell, 1985). Drought is a normal characteristic of the Sahelian climate – for instance, years with below 150 mm of total rain have occurred in the Ferlo almost every decade since 1940 (Andre, 1998). Therefore, the impacts of high stocking densities on rangeland productivity during drought need to be considered in the formulation of management strategies for the Ferlo as well as other Sahelian rangelands that respond in a similar manner to high grazing pressures. Whereas high stocking densities may be beneficial in years with above average rainfall (through a high output from livestock raising), this has to be balanced with the negative consequences of high stocking rates during droughts (Bartels et al., 1990; De Leeuw and Tothill, 1990; Dodd, 1994; Bruce and Mearns, 2002).

The relation between rain-use efficiency and rainfall revealed through the Ferlo grazing experiments also has important implications for the analysis of degradation in the Sahel with satellite images. In these assessments, rain-use efficiency is often used as an indicator of degradation in the Sahel (Kerr, 1998). The rain-use efficiency is calculated for a range of years on the basis of biomass production, derived from a vegetation index calculated with satellite images, and rainfall data from a number of weather stations. A decline of the rain-use efficiency over the years would indicate degradation of the vegetation in the Sahel. Crucially, in several of these studies, a linear relation between rainfall and biomass production has been assumed (e.g. in Nicholson et al., 1998; Kerr, 1998; and Prince et al., 1998). This implies that rain-use efficiency is independent of rainfall. However, the Widou-Thiengoly grazing experiment demonstrates that rain-use efficiency is a quadratic function of the rainfall – and that rain-use efficiency is significantly lower in years with low rainfall. This has a considerably impact upon the validity of the mentioned studies. Nicholson et al. (1998) assess rain-use efficiency in the western Sahel in the period 1982 to 1993, and Prince et al. (1998) analyse rain-use efficiency in the Sahel at large for the period 1982-1990. Kerr's article is largely based upon the two other studies (Kerr, 1998). Both studies do not find a downward trend in the rain-use efficiency in the analysed periods and conclude that satellite images do not demonstrate significant degradation in the Sahel. However, the rainfall data that they provide themselves shows that the early years of their analyses were substantially drier than average (each year in the period 1982 to 1987 had below average rainfall), whereas the last years of their analysis (1989-1993) had average rainfall (Nicholson et al., 1998; Prince et al., 1998). The Ferlo data show that years with below average rainfall tend to have a lower rain-use efficiency than years with an average rainfall. Therefore, in the absence of degradation, the more recent years with average rainfall would have had a *higher* rain-use efficiency than the earlier, drier years. To the contrary, the studies find that rain-use efficiency does not significantly decrease or increase in the examined periods. Therefore, it is likely that the conclusion of these studies that satellite data do not show an increasing degradation in the Sahel (Nicholson et al., 1998; Kerr, 1998; and Prince et al., 1998) is wrong. Taking the rainfall characteristics of the examined years into account, the relatively constant rain-use efficiency found in these studies does indicate an ongoing process of degradation in the Sahel.

6.6 Conclusions

This paper compares the impacts of two different grazing regimes on species composition, herb biomass production and rain-use efficiency in the Ferlo rangeland in the Northern part of Senegal. The analysis is based upon 10 years of data for two grazing regimes: a high, variable stocking (on average 0.15-0.20 TLU/ha), and medium, fixed stocking (0.10 TLU/ha). The high stocking rate corresponds to the current grazing practices in the area. The experiments show that the two grazing pressures lead to a markedly different herbaceous species cover, in terms of the occurrence of dominant species. This does not evoke substantial differences in the overall feed quality of the herb layer. However, the production of herbaceous biomass and the rain-use efficiency are significantly different for the two grazing regimes. A high grazing pressure strongly reduces the rain-use efficiency of the herbaceous layer in dry years. Hence, during a drought, biomass production in the Ferlo is affected both by the low availability of water to plants, and by a reduced efficiency of herbaceous plants to use the available water. Consequently, the impacts of high grazing pressures on the productivity of the Ferlo are hardly noticed during years with normal or above normal rainfall, but they are particularly strong during a drought. In view of the frequent occurrence of droughts in the Ferlo (Andre, 1998), and the large impacts of drought on livestock mortality and food security for local pastoralists (Sinclair and Fryxell, 1985; Guerin et al., 1993), this needs to be considered in the formulation of management strategies for the Ferlo as well as other rangelands that show similar responses to high grazing pressures.

Annex 6.1 Statistical analyses

In this annex, the statistical significance of the differences between plots with medium and high grazing pressure are tested for, respectively: (i) biodiversity; (ii) herbaceous biomass production; and (iii) rain-use efficiency.

(i) Biodiversity. The aim of this test is to analyse if the differences in biodiversity between plots with medium and high grazing pressure are statistically significant, at the $p=0.05$ level. A standard difference-of-means test is used (Blalock, 1987). Data are available for the year 1990, for 5 plots with a medium grazing pressure, and 5 plots with a high grazing pressure, see table A6.1 below.

Table A6.1. Average numbers of species per plot under two grazing regimes

	Number of samples	Average	Standard deviation
Medium grazing pressure	5	49.3	7.42
High grazing pressure	5	42.8	3.49

1. Assumptions
Random sampling; population differences distributed normally.
0-hypothesis: no differences in biodiversity between plots with medium and high grazed pressure: $\mu d = 0$

2. Sampling distribution
Population standard deviation is not known, hence it is required to use the t distribution with $n1 + n2 - 2 = 8$ degrees of freedom

3. Significance level and critical region
At the 0.05 level, for a two-tailed test, for 8 degrees of freedom: if $t \geq 2.306$, the 0-hypothesis is rejected

4. Computing the test statistic
$\sigma(_{x1-x2}) = 3.76$; $t = 1.73$

5. Decision
With $t < 2.306$, the 0-hypothesis can not be rejected and it can be assumed that there is no difference in biodiversity between plots with medium and high grazing pressure, at a significance level of 95%.

(ii) Biomass production. The statistical test used is a matched pairs test, following Blalock (1987). The test is conducted for three cases: (i) all years; (ii) wet years; and (iii) dry years. The test data are shown in the table A6.2. below.

Table A6.2. Biomass production (ton/ha)

	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990
Effective rainfall (mm)	148	133	79	72	201	217	182	206	286	160
Rainfall conditions	Dry	dry	dry	dry	wet	wet	wet	wet	wet	wet
Biomass production: Medium grazing pressure	1.02	0.63	0.22	0.08	0.89	0.95	1.11	1.13	1.05	0.81
Biomass production: High grazing pressure	0.82	0.51	0.06	0	1.02	1	1.01	0.91	1	0.65

The results of the matched pairs test are shown in table A6.3. In all tests, random sampling and normally distributed population differences are assumed. A two-tailed test is conducted, for a confidence level of 95%. The 0-hypothesis is that there is no difference in biomass production between the plots with a high grazing pressure and the plots with a medium grazing pressure.

Table A6.3. Statistical test results

Test	N	Degrees of freedom	Critical t- value	Calculated t-value	Decision
All years	10	9	2.262	2.609	Rejected
Wet years (>150 mm)	6	5	2.571	1.093	Accepted
Dry years (<150 mm)	4	3	3.182	5.422	Rejected

Hence, for the overall period, as well as for the dry years, the impact of grazing on biomass production is significant at the 0.95 confidence interval. For the years with a rainfall above 150 mm, the impact is not significant at this confidence interval.

(iii) Rain-use efficiency. The statistical test used is a matched pairs test, following Blalock (1987). The test is conducted for three cases: (i) all years; (ii) wet years; and (iii) dry years. The test data are shown in table A6.4 below.

Table A6.4. Rain-use efficiency (kg/ha/mm)

	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990
Effective rainfall (mm)	148	133	79	72	201	217	182	206	286	160
Rainfall conditions	dry	dry	dry	dry	wet	wet	wet	wet	wet	wet
Rain-use efficiency: Medium grazing pressure	6.8	4.7	2.8	1.1	4.4	4.4	6.1	5.5	3.7	5.1
Rain-use efficiency: High grazing pressure	5.5	3.8	0.8	0.0	5.1	4.6	5.5	4.4	3.5	4.1

The results of the matched pairs test are shown in table A6.5. In all tests, random sampling and normally distributed population differences are assumed. A two-tailed test is conducted, for a confidence level of 95%. The 0-hypothesis is that there is no difference in rain-use efficiency between the plots with a high grazing pressure, and the plots with a medium grazing pressure.

Table A6.5. Statistical test results

Test	N	Degrees of freedom	Critical t-value	Calculated t-value	Decision
All years	10	9	2.262	2.974	Rejected
Wet years (>150 mm)	6	5	2.571	1.161	Accepted
Dry years (<150 mm)	4	3	3.182	5.536	Rejected

Hence, for the overall period, as well as for the dry years, the impact of grazing on rain-use efficiency is significant at the 0.95 confidence interval. For the years with a rainfall above 150 mm, the impact is not significant at this confidence interval.

7. Managing stochastic dynamic ecosystems: livestock grazing in a Sahelian rangeland

Adapted from Hein, L.G. and H.P. Weikard, 2004. Managing stochastic dynamic ecosystems: livestock grazing in a Sahelian rangeland. Submitted.

7.1 Introduction

Rangelands are the vast tracks 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 many rangelands, the erratic nature of the rainfall causes livestock keeping to be an insecure and marginal source of income. Therefore, efficient and sustainable management of rangelands is of utmost importance for the local populations. One of the most important aspects of rangeland management is the *stocking rate* of livestock. Assessment of the optimal stocking rate requires accounting for stochasticity in the ecosystem (e.g. related to rainfall) and for the dynamic feedback mechanism in the ecosystem (e.g. through overgrazing) (Karp and Pope, 1984; Westoby et al., 1989; Torell et al., 1991; Sullivan and Rohde, 2002).

In this paper, we study optimal long-term livestock stocking rates for semi-arid rangelands. We propose a general model, and apply the model to the Ferlo pastures in northern Senegal. In contrast to earlier ecological-economic work on rangeland management, we pay special attention to ecosystem dynamics as well as stochasticity, while applying model parameters measured in ecological studies. Specifically, we account for the fact that the livestock grazing regime will influence the long-term capacity of the land to produce animal feed.

Our ecological-economic model and the case study are based upon recent insights in rangeland dynamics (Walker, 1993; Ludwig et al., 2001; Walker and Abel, 2002), as well as an analysis of primary data of 10 years of grazing experiments conducted in the Ferlo, as reported in Klug (1982), Miehe (1992, 1997) and Andre (1998). These data have been re-examined in order to reveal the joint impact of rainfall variability and stocking rate on rangeland productivity. The outcomes are used to enhance the existing standard model for rangeland management (Hildreth and Riewe, 1963).

The chapter is organised as follows. Section 7.2 introduces the theoretical background, and presents as a benchmark a simple stochastic single period grazing model. This single period model does not account for the impact of high stocking rates on the long-term productivity of the ecosystem. In section 7.3, we make a novel contribution to the literature by modifying the benchmark model to account for impacts of grazing on forage production under variable rainfall conditions. This has, to our knowledge, not been studied before in the literature. In section 7.4, we apply our model in order to calculate the optimal long-term stocking rate for the Ferlo, northern Senegal. Section 7.5 presents the discussion, followed by the main conclusions in section 7.6.

7.2 Theoretical background

The study of economics and management of renewable resources has a long tradition. Early contributions have been devoted to forestry (Faustmann, 1849; Von Thünen, 1863) while contributions to 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 for the resource and consider quality and price changes, cost for inputs and harvesting costs. Forest management has, in particular, focused on the choice of optimal harvest time. In fisheries and grazing systems the key decision variable is the size of the stock. Although fisheries and grazing systems are similar in many respects, an important difference is that costs are assumed to be decreasing in stock in fisheries models (as it is easier to harvest from a large stock), but increasing in stock in grazing models (due to labour costs, veterinary costs and interest).

In this study we restrict attention to the case of grazing and rangeland management. The literature comprises simple single-period models that implicitly assume exogenous forage production of the rangeland (e.g. Dillon and Burley, 1961; Hildreth and Riewe, 1963; Walters 1968; Riewe, 1981); stochastic single period models with a risk averse manager (e.g. McArthur and Dillon, 1971); and fully dynamic models. These latter models consider effects of the stocking rate on forage production, see e.g. Karp and Pope (1984), Rodriguez and Taylor (1988) and Torell et al. (1991).

To date, however, there are few ecological-economic rangeland models that encompass both dynamics and stochasticity (Janssen et al., 2004). For example, Perrings and Walker (1997) have developed a model for fire driven rangelands that accounts for stochasticity in the occurrence of fires and that allows for multiple states in terms of grass and woody plants cover. Cooper and Huffaker (1997) have studied optimal grazing in a rangeland driven by both grazing and competition between perennial grasses and weeds. Finally, Perrings (1997) presents a model that includes stochasticity and the impact of grazing based upon a Clementsian approach to rangeland productivity, where the impacts of high grazing pressure on the vegetation can be reversed in the short term by reducing the stocking rate.

Our model goes beyond the existing analysis. We enhance the ecological realism of existing modelling approaches by explicitly modelling the interconnected impacts of grazing and rainfall variability on rangeland productivity. The importance of our approach lies in modelling the feedback effect of grazing on rangeland productivity, and including in the model that the impact of this feedback effect depends upon stochastic weather conditions. In line with recent ecological insights, we base our model on the assumption that it is the long-term grazing pressure, rather than the annual grazing pressure, that drives the development of the vegetation cover of the rangeland (Le Houérou, 1989; Illius and O'Connor, 1999).

Rangeland dynamics and productivity

Rangelands consist of a herbaceous layer, and an open bush and tree cover. The herbaceous layer normally comprises a mix of annual and perennial grasses, leguminous species and other plants (Walter, 1971; Walker, 1981). Rangeland dynamics are driven by a range of processes including rainfall, grazing, fire and, in some rangelands, frost. The precise, cumulative impact of these different processes is still subject to discussion (Westoby et al., 1989; Walker, 1993; Illius and O'Connor, 1999; Sullivan and Rohde, 2002; Briske et al., 2003). In line with the focus of this paper, we will summarise recent insights in the effects of rainfall variability and grazing, two important driving factors for rangeland dynamics.

The close relation between annual rainfall and rangeland productivity is widely acknowledged. As water is the main limiting factor in most rangelands, high rainfall will result in increased productivity. Commonly, rangelands experience high temporal and spatial variability in rainfall, which causes high seasonal, interannual, and spatial variations in animal feed availability. Herbivore grazing influences the vegetation layer in a more complex way. A first impact is the enhanced recycling of nutrients as animals convert standing dead material to manure and animal biomass (Powell et al., 1996). Secondly, animals influence the soil texture. Whereas high stocking rates may lead to compaction, low stocking rates may be beneficial to soil texture by breaking crusts formed on the topsoil (Walker, 1981). Thirdly, herbivory influences the species composition of the rangeland. Overgrazing may cause the disappearance of certain species from the plant community (Walker, 1981).

The combined impact of grazing pressure and rainfall variability has been examined in, among others, Sinclair and Fryxell (1985), Le Houérou (1989), Illius and O'Connor (1999), and Sullivan and Rohde (2002). From these studies, we derive that the year-to-year variation in rainfall is the most important factor that determines the annual productivity of the rangelands. Grazing pressure influences the composition, functioning and productivity of the ecosystem in particular in the medium and long term. Continuously high, unsustainable grazing pressures cause a decline in the rain-use efficiency of the vegetation. Rain-use efficiency is an important indicator for the functioning of ecosystems, and expresses the amount of biomass produced per unit of rainfall (Le Houérou, 1984; Sinclair and Fryxell, 1985; Snyman, 1998; Illius and O'Connor, 1999).

The benchmark – deterministic and stochastic one period models

In this section, we describe a standard, one period rangeland model, as it is commonly applied in literature to calculate the optimal stocking density for a rangeland. In the next section (section 7.3), this model is enhanced through a more realistic modelling of the impact of grazing and rainfall variability. We first describe a deterministic, one period model, followed by a stochastic, one period model.

(i) A deterministic, one period model. In its simplest form, a standard model for rangeland management takes the annual forage production of a pasture as fixed (Hildreth and Riewe, 1963). This implies a given annual rainfall. The standard model assumes no ecological impact of the stocking rate s, and given prices. A pasture's forage production F can be translated into its grazing capacity s_{max}. Let φ be the amount of plant biomass required to allow the subsistence of a livestock unit. Then

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

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

$$\Delta s = \beta(1 - \frac{s}{s_{max}}) \cdot s \tag{2}$$

where s is livestock, Δs is the gain in livestock, $s_{\max}$ is the grazing capacity of the rangeland and $\beta > 0$ is a scaling parameter capturing the potential natural growth in livestock. We assume that the objective of the rangeland manager is to maximise profits, and that livestock can be sold at price p per livestock unit. Furthermore there is a variable cost c per livestock unit (e.g. interest and veterinary services) and a fixed cost c_0 (e.g. land tax and fencing). The profit function can be written as

$$\pi(s) = p\Delta s - cs - c_0 . \tag{3}$$

In order to optimise rangeland management, the rangeland manager chooses the stocking rate s to maximise profits π:

$$\max_s \pi(s) . \tag{4}$$

From the first order condition of the maximisation problem (4) we derive that

$$c = p\beta - \frac{2p\beta}{s_{\max}} \cdot s^* , \tag{5}$$

where s^* is the optimal stocking rate. Equation (5) is a standard efficiency condition stating that marginal costs equal marginal revenues (see figure 7.1). Rearranging (5), the optimal stocking rate is written as

$$s^* = \frac{p\beta - c}{2p\beta} \cdot s_{\max} . \tag{6}$$

(ii) A stochastic, one period model. In order to examine the impact of variable, stochastic rainfall on the optimal stocking rate we modify the assumption that forage production is given and fixed. Instead we assume that forage production is a function of rainfall r ($r \in [r_{\min}, r_{\max}]$; $0 < r_{\min} < r_{\max}$):

$$F = F(r) . \tag{7}$$

We assume here that rainfall is productive, $F(r) > 0$, but its marginal productivity is decreasing, $d^2F / dr^2 \le 0$ for the relevant range of rainfall.[2] Hence, applying (1), the grazing capacity is (in the relevant range) a concave function of rainfall. Consider a case where the stocking rate must be chosen at the beginning of the season before rainfall and grazing capacity are known. Let $g(r)$ be the probability density function of rainfall. The rangeland manager maximises *expected* profits, π^e:

$$\pi^e = \int_{r_{\min}}^{r_{\max}} g(r) \cdot \pi(s_{\max}(r))\, dr . \tag{8}$$

[2] Note that we do not require $dF / dr > 0$ and allow the possibility of a reduction in rain-use efficiency at rainfall levels that are substantially above the average annual rainfall.

The optimal stocking rate in a stochastic setting will be lower than the stocking rate for a given average rainfall (cf. equation (6)); see Annex 7.1 for further analysis of the first order conditions of the rangelands manager's maximisation problem.

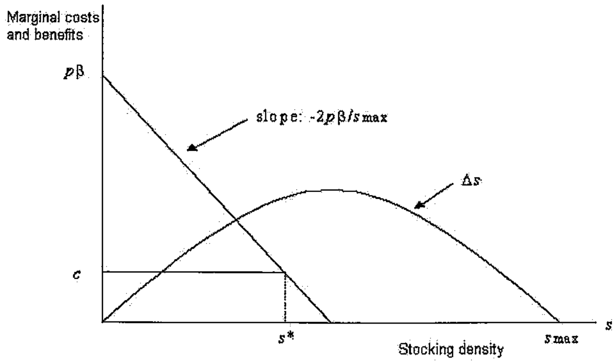


Figure 7.1: Optimal stocking rate

7.3 An enhanced ecological-economic model for rangeland management

In this section we develop a conceptual model that constitutes a refinement of the benchmark model presented above. Specifically, the model captures the ecological economic link between grazing and rangeland productivity. It provides a tool for rangeland management that can be used to calculate the expected income derived from a pasture under different stocking rates and variable, stochastic rainfall conditions. The model indicates how these factors influence grass production, animal production and pastoralists' income. For simplicity reasons, our model does not account for the spatial variability of rangelands, but it would also be applicable in a spatially explicit context (see e.g. Turner, 1999, for more information on this topic). The applicability of the model is demonstrated in the next section. The ecological feedback mechanism included in the model is inspired by data from grazing experiments in the Ferlo region, northern Senegal. The structure of the model is, however, more general and applies to a wider class of rangelands.

Description of the conceptual model

The set-up of the model is presented in figure 7.2. The model comprises the assessment of (1) grass production, (2) livestock production and (3) income. These three steps are described below. In the development of the conceptual model, we focus on the ecology-economy link (steps 1-2) as this is the main contribution of our paper. Step 3 specifies the problem of choice of an optimal long-term stocking rate under constant prices. The issue of price movements is only dealt with in the case study of the Ferlo in section 4. We include this issue in the case study only because the primary focus of this paper is on the impacts of the ecological feedback effect, and because the impact of price changes depends upon the management strategy pastoralists chose to cope with drought, which needs to be examined and modelled specifically for each area.

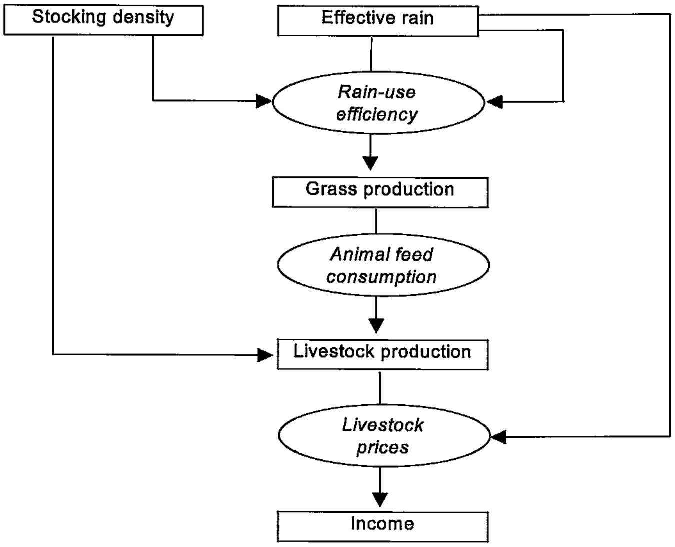


Figure 7.2: Outline of the model to assess the efficiency of livestock stocking in a dynamic rangeland. In squares the main steps of the model, in circles the main variables or parameters that are required to calculate the next step in the model. The arrows indicate the main relations in the model.

Step 1. Assessment of the grass production. The grass production depends upon the effective rainfall and the rain-use efficiency. The *effective rainfall* is the amount of rain left after loss through evaporation or run-off (Haan et al., 1994). It represents the rainfall available to plants. The model requires the input of both the average effective rainfall per year, and the annual variation in rainfall, in the form of a rainfall distribution function. The *rain-use efficiency* expresses the net primary production per unit rainfall. Analysis of the grazing data of the Ferlo shows that the rain-use efficiency depends upon (i) the effective rainfall and (ii) the long-term stocking rate. Our data indicates that the rain-use efficiency is an inverted U-shaped function of the rainfall (see figure 6.3). The rain-use efficiency is highest at the average rainfall in the Ferlo, which reflects adaptation of the plant community to the most common rainfall conditions. At a high stocking rate, the rain-use efficiency decreases, but the inverted U-shape is maintained.

We propose the following quadratic equations to capture these effects. First consider a situation without grazing. Let ρ denote rain-use efficiency and r effective rainfall. We assume that rain-use efficiency is positive in the relevant range of rainfall $(r_{\min}, r_{\max})$.[3] Formally, for the benchmark case without an ecological feedback of grazing on the ecosystem or in case of no grazing, a simple relationship capturing the main features is:

[3] In empirical applications, $r_{\min}$ and $r_{\max}$ are parameters to be estimated.

$$\rho_0 = \alpha(r - r_{\min})(1 - \frac{r}{r_{\max}}), \tag{9}$$

where ρ_0 refers to the rain use efficiency in the no grazing case and $\alpha > 0$ is a scaling parameter. On the basis of an analysis of long-term ecological data of the Ferlo rangeland, it is assumed that there are three main general characteristics of the relation between stocking rate and rain-use efficiency: (i) grazing reduces water-use efficiency: $\partial\rho / \partial s < 0$; (ii) the marginal reduction is increasing in the stocking rate: $\partial \frac{|\partial\rho/\partial s|}{\partial s} > 0$; and (iii) the reduction of the rain-use efficiency due to grazing is the lowest at the level of rainfall that generates maximum forage production. Given a benchmark situation *without* grazing, the rain-use efficiency *with* grazing can be written as follows:

$$\rho(r,s) = \rho_0 - d(r,s), \tag{10}$$

where $d(r, s)$ captures the reduction of rain use efficiency due to grazing. The reduction function d depends upon the specific ecosystem. We suggest a simple format for d that maintains the quadratic form of $\rho(r)$ and captures the main features (i)-(iii).

$$d(r,s) = s^{\theta}(\mu r^2 - 2\mu \bar{r}\, r + \nu), \tag{11}$$

where $\mu > 0$, $\nu > 0$ and $\theta > 1$ are parameters and $\bar{r}$ is the level of rainfall generating maximum forage. Observe that if $s = 0$, then $d = 0$. Also, the reduction variable d has a minimum at $\bar{r}$ as required by condition (iii). It is easy to check that conditions (i) and (ii) are also satisfied.

Grass production is determined by the amount of effective rainfall and by the rain-use efficiency of the pasture. It 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; and Palmer, 2000). Hence, (7) can be specified as

$$F = \rho r. \tag{12}$$

Step 2. Assessment of the livestock production. This step proceeds analogous to the benchmark model. The grazing capacity $s_{\max}$ depends upon the plant biomass production and the minimum amount of plant biomass required to maintain one livestock unit as stated in (1). The growth of the livestock herd is assumed to follow a logistic growth process described in equation (2). Hence, substituting (1) and (12) into (2), we obtain

$$\Delta s = \beta(1 - \frac{s\varphi}{\rho r}) \cdot s. \tag{13}$$

Step 3. Assessment of the pastoralist's income. In order to assess the pastoralist's income we adopt the following assumptions. Income is derived from the products supplied on a continuous basis by the animals (milk, wool), and through sale or slaughter of animals. In many cases, pastoralists have relatively few other sources of income (FAO, 2001a). For the next step of the analysis we assume given prices. The production parameter Δs represents the increase in the herd size through reproduction; p represents the corresponding price per livestock unit. The assumption of fixed prices is relaxed in the case study in section 4

In order to determine the optimal stocking rate we use the standard profit function (3). Substituting (2) into (3) we obtain

$$\pi(r,s) = p\beta(1 - \frac{s}{s_{max}(r,s)})s - c \cdot s - c_0 . \tag{14}$$

The ecology-economy interaction enters the rangeland manager's decision problem because forage production F determines the grazing capacity s_{max} and is determined by the rain-use efficiency which is, in turn, determined by the long term stocking rate s. Thus we have

$$s_{max} = s_{max}(F(\rho(r,s),r)) . \tag{15}$$

Given the ecological feedback of grazing on rangeland productivity the rangeland manager's maximisation problem is

$$\pi^e = \int_{r_{min}}^{r_{max}} g(r) \cdot \pi(s_{max}(r,s))\, dr , \tag{16}$$

where (16) differs from (8) because s_{max} is now also a function of the stocking rate. The first order condition is as follows:

$$\frac{\partial \pi^e}{\partial s} = 0 = \int_{r_{min}}^{r_{max}} g(r) \cdot \frac{\partial \pi(r,s)}{\partial s} dr . \tag{17}$$

We provide further analysis and a comparison of cases (deterministic and stochastic; with and without ecological feedback) in Annex 7.1.

So far we have assumed constant prices. However, during a drought many pastoralists will foresee feed shortages and will try to sell animals while the capacity of local markets to absorb the supply is limited. Relatively few local people will buy and prices tend to decrease, in particular in the absence of export markets which could mitigate regional volatility of supply and demand (Turner and Williams 2002). We return to this case toward the end of the next section.

7.4 Case study for the Ferlo, northern Senegal

The case study area

The Ferlo is located in the northern part of Senegal. It is part of the Sahel biogeographic zone, which extends from Senegal in the west to Somalia in the east.

The area has a very modest, undulating, relief. Annual rainfall varies between around 120 and 450 mm, with an average of 290 mm (Andre, 1998). Rain falls predominantly during the short rainy season from July to September. The natural vegetation consists of dry grassland with scattered trees and bushes. The grazing experiment that provided the data that inspired the modelling of the ecological feedback mechanism was conducted in the Widou-Thiengoly study site in Northern Senegal (Klug, 1982; Miehe, 1992, 1992; Andre, 1998). A further description and a map of the Widou-Thiengoly site are provided in section 6.3.

Livestock keeping is the main economic activity in the Ferlo. The principal animals kept are cattle (zebu), sheep and goats. Traditionally, the pastoral population, mostly Peuhls, migrated on an annual basis between the Ferlo and the more humid Sudan zone to the south. Since the 1950s, livestock densities strongly increased related to strong increases in the local population, as well as the enhanced availability of veterinary aid. At the same time, expanding agricultural activities in the Sudan zone limited the migration possibilities of the pastoralists. In combination with the development of numerous boreholes providing perennial drinking water, this has led to increased sedentarisation in the Ferlo (Sinclair and Fryxell, 1985; Guerin et al., 1993). Currently, the average stocking rate in the Ferlo is in the order of 0.15 - 0.20 tropical livestock units per hectare (De Leeuw and Tothill, 1990; Miehe, 1997). A tropical livestock unit (TLU) corresponds to 250 kg of animal weight (Boudet, 1975; Whiteman, 1980).

Modelling and data

To illustrate our ecological-economic model for rangelands, we calculate the optimal long-term grazing pressure for the Ferlo Sahelian rangeland in northern Senegal using three different models. We will subsequently use: a) the benchmark model – assuming no impact of stocking rate on the ecosystem and constant prices; b) the enhanced ecological-economic model which includes the long-term impact of stocking rate on the forage production; and c) a further enhanced model that also accounts for the fact that prices are contingent on the occurrence of a drought. The efficient long-term stocking rates are identified by maximising the profit function for each model. The control variable of the three models is the long-term stocking rate.

In the models, we assume that pastoralists seek to optimise rangeland management by selection of the efficient long-term stocking rate. It is assumed that all animals that would lead to stocking above this long-term rate are sold. However, we allow for lower stocking rates during dry years that have insufficient animal feed production to maintain the stocking rate. We assume that, during drought, pastoralists try to keep as many livestock as possible on the available feed resources. In other words, in a year of drought, the actual stocking rate s equals the grazing capacity s_{max}. This type of stocking strategy has been suggested for rangelands where pastoralists do not have the possibility to supply supplementary feed during drought and where they face difficulties to restock after a drought (FAO, 1988; De Leeuw and Tothill, 1990). This strategy has the advantage that, following a drought, there is relatively much livestock available to restock on the basis of natural reproduction.

The rainfall probability density function for the Ferlo is constructed on the basis of long term rainfall data reported in Andre (1998). This data series covers a 50 years period (1947-1997). Miehe (1992) reports the relation between rainfall and effective rainfall. Ecological data are derived from a grazing experiment conducted in the Widou-Thiengoly catchment, northern Senegal (figure 7.3), as reported in Klug (1982), Miehe (1992, 1997) and Andre (1998). The experiment included the monitoring of the plant biomass production under three different grazing regimes, for a period of 10 years. The data have been re-analysed for this paper in order to reveal the joint impacts of rainfall variability and stocking rate on biomass productivity. The average prices per animal and the herd composition in the Ferlo are from Thébaud et al. (1995). In order to evaluate the impact of drought on livestock prices, long term livestock price data for Western Niger are analysed, as reported in Turner and Williams (2002). In the absence of such data for the Ferlo, it is assumed that the relative price fluctuations in the Ferlo correspond to the price changes in western Niger, but we are aware that in reality a discrepancy may exist.

a) The benchmark model

We first apply the benchmark model to the case of the Ferlo. The model assumes no ecological feedback and fixed prices for livestock. Analogous to figure 7.2, there are three basic steps: (i) assessment of the grass production; (ii) assessment of the livestock production; and (iii) calculation of the income derived from livestock keeping.

Step 1. Assessment of the grass production. The grass production is calculated as a function of effective rain and the rain-use efficiency. Analysis of the rainfall and the effective rainfall in the Ferlo during the period 1981-1990 has shown that, on average, the effective rain is 65% of the total annual rainfall (Miehe, 1992). The estimate obtained is

$$r = 0.65\, r_{\text{tot}}; \quad (\text{n} = 10, \text{R}^2 = 0.98, \text{F} = 448),$$

where, as before, r is the effective rainfall (mm) and r_{tot} is the total annual rainfall (mm). The rainfall probability density function g(r), required to calculate the expected profits as a function of the long-term stocking density, is based upon a 50 years rainfall data set for the Ferlo (Andre, 1998). This data set shows that the average effective rainfall is 189 mm/year, and the standard deviation of the rainfall is 55 mm/year. Based upon these two parameters, a probability density function has been constructed (Blalock, 1987). The production of forage is calculated by multiplying the effective rain with the rain-use efficiency. In the benchmark model, the rain-use efficiency is a function of the effective rainfall only. Based on regression analysis, the following relation between rain-use efficiency ρ_0 (kg/mm) and effective rainfall is derived:

$$\rho_0 = -\,0.00021\, r^2 + 0.0756\, r - 1.254. \qquad (9^*)$$
$$(\text{n} = 10, \text{R}^2 = 0.62, \text{F} = 5.7)$$

The forage production F (kg) is straightforwardly calculated from $F = \rho_0 \cdot r$ (see equation 12).

Step 2. Assessment of the livestock production. The annual grazing capacity s_{max} is calculated by dividing the annual forage production F by the amount of plant biomass production required to maintain one TLU, φ. In order to calculate this amount, a number of factors have to be taken into account. First, the minimum amount of feed that the animals need to maintain themselves, which has been estimated at 4.3 kg/TLU/day – based upon the local livestock mix (Thébaud et al., 1995) and the energy requirements per animal (Bayer and Waters-Bayer, 1998). Second, not all herb biomass is available for grazing, due to decomposition, fire, or the unpalatability of certain plants. Estimates of the rates of plant material actually available to livestock range from 35% (Penning de Vries and Djitèye, 1982) to 50% (Breman and De Ridder, 1991). Because these rates tend to be higher for drier areas (Bayer and Waters-Bayer, 1998), we assume in our models that 50% of plant biomass is available for grazing. Third, it needs to be taken into account that animals supplement their diet with woody plants, particularly in the dry season. According to Breman and De Ridder (1991), the dietary contribution of woody plants can be estimated at 20% in the Ferlo. On the basis of the above, we estimate that φ equals $4.3 \cdot 365 \cdot 2 \cdot 0.8 =$ 2511 kg herb biomass / TLU / year.

Following the conceptual model, the livestock population grows according to a logistic growth curve as a function of the livestock population and the grazing capacity. 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 $\beta = 0.6$. Using this estimate, the growth of livestock Δs as given in (13) is fully specified.

Step 3. Management of the herd and income. We assume that the aim of the pastoralist is to maintain the most efficient long-term stocking rate, except during drought, when the long-term stocking rate cannot be maintained and the stocking rate will be reduced to the level of the grazing capacity s_{max}. Under such a stocking regime, sales are Δs in a normal year, and $s - s_{max}$ during a drought. After a drought, $s - s_{max}$ animals are bought from outside the region to restock as the growth of the herd is insufficient to stock up to the long-term stocking rate. Under the assumption of constant prices, destocking and restocking affects profits only indirectly as a reduced herd size has a lower opportunity cost of capital. Hence, profits during drought are $-cs_{max}$.

For Senegalese pastoralists, the main source of income is the selling of animals for meat; milk and wool are mainly used for subsistence and are less important in terms of income (Guerin et al., 1993). Therefore, in the model, it is assumed that income is only derived from the sale of animals. Based upon the average price per animal and the local livestock mix (Thébaud et al., 1995), the overall, average livestock price is 24750 CFA/TLU.

Regarding the costs of livestock herding in northern Senegal, we assume that all costs are variable costs, related to capital and labour inputs required to maintain the herd. It is assumed that 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 some 2% (IMF 1999). The average *labour* costs in rural Senegal can be 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 costs. With an average herd size of 44 TLU per family (Thébaud et al., 1995), the annual labour costs amount to 100,000 /44 /2 = 1140 CFA/TLU. Therefore, the profit function is specified as follows:

$$\pi = 24750 \cdot \Delta s - 0.16 p \cdot s - 1140 \cdot s \qquad (3^*)$$

Results. Based on the simple grazing model that assumes no ecological feedback and constant prices, the average annual income (in CFA) that can be obtained from one hectare of rangeland has been calculated for a range of fixed maximum stocking densities. The optimal long-term stocking rate is 0.11 TLU/ha. The corresponding annual income is 525 CFA/ha.

b) The ecological economic model

Next, we include the ecological feedback of grazing pressure on long-term forage production in our model. Analysis of the grass production data of the Ferlo under different rainfall conditions and grazing regimes shows that a high grazing pressure causes a reduction of the rain-use efficiency, and hence productivity, of the grassland. This effect is the strongest in dry years. It reflects a loss of resilience for drought as a consequence of a high grazing pressure. Using regression analysis (ordinary least squares) we estimate from the available data that the rain-use efficiency is

$$\rho = -0.00021 r^2 + 0.0756\, r - 1.254 - s^2 (0.00504 r^2 - 2.016 r + 210.8) \qquad (10^*)$$
$$(n = 30, R^2 = 0.76, F = 28).$$

Results. We find that the optimal long-term stocking rate is 0.10 TLU/ha, corresponding to an annual net income of 492 CFA/ha. The optimal stocking rate is about 10% lower than in the benchmark case that does not account for the ecological feedback effect. The maximum long-term expected annual income is also lower compared to the maximum income in the benchmark model because the inclusion of the feedback mechanism in the model accounts for a reduction of the average annual feed production in the rangeland.

c) Rangeland management with ecological feedback and variable prices

The third model variant accounts for both an ecological feedback and variable prices. In drylands, livestock prices tend to 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 are 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. Based upon these data, it is assumed in the model that prices drop to 43% during years with a drought and that they increase to 146% subsequent to a drought. Hence, when the constant-prices assumption does not apply, a drought causes additional losses as a part of the stock, $s - s_{max}$, has to be sold at low prices, while high prices prevail during restocking. This is reflected in the profit function adjusted for variable prices used for the Ferlo case study (formula 14*). In years with normal or high rainfall, the profit function equals the profit function of the previous model (equation 14). However, during a drought, when s_{max} is lower than the long-term stocking rate s, the profit function changes. The parameter p_{before} reflects the low price at which farmers have to sell during a drought, whereas p_{after} reflects the prices immediately after a drought when farmers need to restock.

The profit function has also been adjusted for the impact of sequential dry years; a second year of drought delays restocking and reduces the price effect. This adjustment is based upon an analysis of dry years in the period 1947-1997 (as reported in Andre, 1998). These data show that, on average, around 20% of the drought years is consecutive to another drought year. As, in our model, the farmer does not have to sell or restock in between two dry years, the price effect does not occur in 20% of the drought years. This is included in equation 14* by means of the adjustment factor η, which equals 0.8.

$$\pi(r,s) = \begin{cases} -c \cdot s_{max} - c_0 + \eta \cdot [p_{before}(s - s_{max}) - p_{after}(s - s_{max})];\ if(s_{max} < s) \\ p\,\beta(1 - \dfrac{s}{s_{max}(r,s)})s - c \cdot s - c_0;\ if(s_{max} \geq s) \end{cases} \qquad (14^*)$$

Results. Including variable prices in the grazing model further reduces the optimal long-run stocking rate. As a drought causes additional losses the vulnerability to drought can be decreased by further decreasing the long-term stocking rate. With an adverse price effect the optimal long-term stocking rate is 0.09 TLU/ha, with a net annual income of 456 CFA/ha.

Comparison of results

The outcomes of the three models are summarised in Table 7.1. Inclusion of the ecological feedback effect reduces the optimal (economic efficient) stocking rate from 0.11 to 0.10 TLU/ha. If the model is run with variable prices, the optimal stocking rate is further reduced to 0.09 TLU/ha. For comparison, we also calculated the approximate annual income derived from the Ferlo at the *current* stocking rate. Model c) predicts that, for a stocking rate of 0.15, the average annual income is 237 CFA/ha/year – significantly below the income that could be obtained at a lower stocking rate.

Table 7.1. Optimal long-term stocking rates for the Ferlo

Model	Optimal long-term stocking rate (TLU/ha)	Annual income (CFA/ha/year)
a) The benchmark stochastic, one period model	0.11	525
b) The ecological-economic model (stochastic + ecological feedback)	0.10	492
c) The ecological-economic model including variable prices	0.09	456

7.5 Discussion

In this section, we first discuss the reliability, and the limitations of the selected modelling approach. Subsequently, we discuss the potential implications of the study for the management of dynamic, semi-arid rangelands in general, and the Ferlo rangeland in particular.

On the reliability and limitations of the modelling approach. Our model presents an extension of existing models in that it is dynamic and stochastic, and that it incorporates a realistic modelling of the ecological aspects of rangeland dynamics. Specifically, our model accounts for the linkage of the impacts of high grazing pressures and stochastic rainfall. The key relations underlying our model are that (i) rain-use efficiency varies with rainfall according to a quadratic function; and (ii) grazing affects the rain-use efficiency of the vegetation, in particular in the long-term. 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.

For our case study, the relations indicating the development of the rain-use efficiency as a function of rainfall and stocking rate have been statistically tested and appeared highly significant. The two equations that establish respectively the relation between rainfall and rain-use efficiency in the absence of grazing (equation 9*) and with grazing (equation 10*) were both significant at the p=0.05 level. Considering that our approach presents an enhanced way of dealing with the ecological feedback of grazing in the rangeland modelling, and that the main equations underlying this part of the model were highly significant, it is anticipated that the approach may be relevant to other modelling exercises of semi-arid rangelands as well.

Nevertheless, the model presented in this study is subject to two main constraints. First, we do not consider the occurrence of multiple steady states. In rangelands, the existence of multiple steady states has been demonstrated in relation to the ratio grass cover – bare soil (Van de Koppel et al., 1997; Scheffer et al., 2001) and in relation to the ratio grass – woody plants cover (Friedel, 1991; Laycock, 1991). Our models do not include the mechanisms that occur during or subsequent to such transitions and, hence, they are applicable only to systems that are dominated by herbaceous plants. This has no significant impact on the outcomes of the models as the found optimal stocking rates represent an ecosystem state dominated by herbaceous species. Second, the paper does not explicitly address 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, we ignore in our model that fire may have a strong impact on the availability of woody plant biomass for browsing. However, we do account for the impact of fire on the availability of herbaceous biomass for grazing (through its impact on the parameter φ that indicates how much of the produced plant biomass is available for grazing). As in the Ferlo only around 20% of animal feed is obtained through browsing (Breman and De Ridder, 1991), we expect that the unaccounted impact of fire has only a moderate impact on the results of our case study.

On the management of dynamic rangelands. There exists a variety of different rangeland management systems, including nomadic, transhumant, agropastoral and ranching systems (FAO, 2001b). Nomadic systems are the most opportunistic, with annually variable migration routes, whereas ranching systems involve fenced pastures with controlled stocking rates. The first three systems often comprise communal grazing lands, where the total stocking densities equal the sum of the stocking rates applied by the individual pastoralists. Only in the case of ranching systems, the stocking rate can be controlled by one owner. For all types of rangeland management, optimal stocking rates have been extensively discussed, see e.g. McCarthy et al. (1998) and Snyman (1998). In the case of the systems with free access to the animal feed resources, government agencies and development institutes require information on current stocking versus most efficient stocking in order to judge if they should promote increases or reductions in stocking rates (Bruce and Mearns, 2002). In the case of the ranching system, the manager of the ranch will be interested in adopting the optimal stocking rate.

This paper shows that accounting for the impact of grazing on rangeland vegetation leads to a lower efficient stocking rate compared to the most efficient stocking rate calculated without this ecological feedback. In other words, overstocking reduces the efficiency of rangeland grazing. This confirms the arguments by Le Houérou (1984) and Illius and O'Connor (1999) – and contradicts the statements of Ellis and Swift (1988) and Scoones (1994) who state that high stocking rates do not affect rangeland productivity. Accounting for variable prices, which tend to decrease during droughts and increase immediately after droughts (cf. Campbell et al., 2000; Turner and Williams, 2002), further reduces the efficient stocking rate. With variable prices, it is worthwhile to better preserve rangeland conditions so that less animals have to be sold during a drought when prices are low.

In the Ferlo, as in the Sahel at large, transhumance is the dominant system. Pastoralist migrate according to specific, seasonal patterns, complemented with more permanent settlements for the least mobile part of the population (e.g. elderly people). The current average stocking rates 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). Our paper shows that, assuming fixed prices, the most efficient stocking rate is 0.10 TLU/ha. With variable prices, this further reduces to about 0.09 TLU/ha. Therefore, the optimal long-term stocking rate is substantially lower than the current stocking rate.

We have not addressed 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). Hence, accounting for risk-aversion of pastoralists may lead to a further reduction of the optimal stocking rate. Government and development projects should therefore not be aimed at increasing stocking rates in the area. Instead, they may be directed at improving the functioning of the livestock markets, which may enhance the capacity of pastoralists to adjust stocking rates to the annual grazing capacities by increasing the opportunities to buy or sell livestock (see e.g. Holtzman and Kulibaba, 1994).

7.6 Conclusions

This paper offers a conceptual approach to include stochasticity related to rainfall in a multi-period grazing model where grazing pressure affects the long-term productivity of the rangeland. The model allows for an assessment of the optimal long-term stocking rate. A key variable in the model is the rain-use efficiency that expresses plant production per mm of rainfall, and which is affected by both rainfall and grazing pressure (cf. Le Houérou et al., 1988). The first order conditions of the model are provided (see Table A1, Annex 7.1) and the model is applied to the Ferlo rangeland in northern Senegal.

With respect to the ecological feedback mechanism, our analysis shows that the impact of high grazing pressures on rangeland productivity is most pronounced in years with low rainfall. Accounting for the impact of grazing reduces the optimal long-term stocking rate in the Ferlo from 0.11 to 0.10 TLU/ha. Hence, in the context of the ongoing discussions on the implications of high stocking rates (Illius and O'Connor, 1999; Sullivan and Rohde, 2002), it can be concluded that high stocking rates can have a significant impact on rangeland productivity, and that this needs to be accounted for in the ecological-economic modelling of semi-arid rangelands.

With regards to income derived from the Ferlo rangeland, we examine a frequently observed phenomenon. During a drought livestock prices fall, whereas after a drought they rise above average levels. This adverse price pattern puts additional pressure on incomes and leads to a further reduction of the optimal stocking rate, to 0.09 TLU/ha. Hence, the analyses indicate that the current stocking density in the Ferlo (around 0.15-0.20 TLU/ha) is considerably above the economic optimal stocking rate. Government policies should be directed towards reducing herd sizes rather than supporting further increases in the stocking rate.

Appendix 7.1 First order conditions

According to (8) the rangeland manager's *expected* profits are:

$$\pi^e = \int_{r_{\min}}^{r_{\max}} g(r) \cdot \pi(s_{\max}(r))\, dr \,.$$

From integration by parts we obtain

$$\pi^e = \pi(s_{\max}(r_{\max})) - \int_{r_{\min}}^{r_{\max}} \frac{\partial \pi(s_{\max}(r))}{\partial r} G(r)\, dr \,, \tag{A1}$$

where $G(r) \equiv \int_{r_{\min}}^{r} g(r)\, dr$. Maximisation of expected profits yields the first order condition:

$$s^* = \frac{p\beta - c}{2p\beta} \cdot s_{\max}(r_{\max}) \underbrace{\frac{1}{1 + s_{\max}(r_{\max}) \int_{r_{\min}}^{r_{\max}} \frac{\partial s_{\max} / \partial r}{s_{\max}^2} G(r) dr}}_{\Omega} \,. \tag{A2}$$

When comparing (6) and (A2) an additional term, denoted Ω, shows up in (A2). In general the denominator of Ω is the larger the larger the impact of rainfall on the grazing capacity and the larger the variance of the distribution $g(r)$. Hence, the optimal stocking rate decreases with increasing impact of rainfall and rainfall variability. Note that (A2) collapses to (6) if there is no impact of rainfall on the grazing capacity ($\partial s_{\max} / \partial r = 0$) or if $g(r)$ is a degenerated distribution such that $r_{\min} = r_{\max}$ (i.e. in the case of certainty).

We further analyse the first order condition (17) for the case of stochastic rainfall and an ecological feedback of grazing on rain-use efficiency.

Integration by parts gives

$$\begin{aligned} \frac{\partial \pi^e}{\partial s} 0 &= \left| {}_{r_{\min}}^{r_{\max}} G(r) \frac{\partial \pi}{\partial s} - \int_{r_{\min}}^{r_{\max}} G(r) \cdot \frac{\partial^2 \pi}{\partial r \partial s} dr \right. \\ &= \frac{\partial \pi}{\partial s} - \int_{r_{\min}}^{r_{\max}} G(r) \cdot \frac{\partial^2 \pi}{\partial r \partial s} dr \,. \end{aligned} \tag{A3}$$

Differentiating (14) we obtain

$$\frac{\partial \pi}{\partial s} = p\beta - c - \frac{2p\beta s}{s_{\max}} + p\beta s^2 \frac{\partial s_{\max} / \partial s}{s_{\max}^2} \,. \tag{A4}$$

Differentiating (A4) we obtain

$$\frac{\partial^2 \pi}{\partial r \partial s} = 2p\beta s \frac{\partial s_{\max} / \partial r}{s_{\max}^2} + p\beta s^2 \left(\frac{(\partial^2 s_{\max} / \partial r \partial s)}{s_{\max}^2} - \frac{2(\partial^2 s_{\max} / \partial r \partial s)}{s_{\max}^3} \right). \tag{A5}$$

Combining (A3), (A4) and (A5) gives the following first order condition:

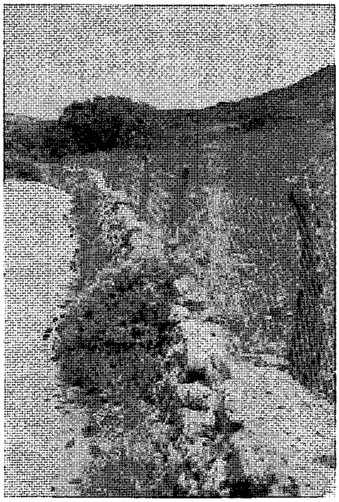
$$\begin{aligned} \frac{\partial \pi^e}{\partial s} = 0 = p\beta - c - \frac{2p\beta s}{s_{\max}(r_{\max})} + \underbrace{\frac{p\beta s^2}{s_{\max}^2} \frac{\partial s_{\max}}{\partial s}}_{\text{ecological feedback}} \underbrace{- 2p\beta s \int_{r_{\min}}^{r_{\max}} G(r) \frac{\partial s_{\max} / \partial r}{s_{\max}^2} dr}_{\text{stochastic effect}} \\ \underbrace{- p\beta s^2 \int_{r_{\min}}^{r_{\max}} G(r) \left(\frac{\partial^2 s_{\max} / \partial r \partial s}{s_{\max}^2} - \frac{2\partial^2 s_{\max} / \partial r \partial s}{s_{\max}^3} \right) dr}_{\text{stochastic ecological feedback effect}} \end{aligned} . \tag{A6}$$

Table A1 below shows a comparison of results.

Table A1: Comparison of first order conditions

	Deterministic rainfall $r_{\min} = r_{\max}$
Single period $\frac{\partial s_{\max}}{\partial s} = 0$	$\frac{\partial \pi}{\partial s} = 0 = p\beta - c - \frac{2p\beta s}{s_{\max}}$
Long-term $\frac{\partial s_{\max}}{\partial s} < 0$	$\frac{\partial \pi}{\partial s} = 0 = p\beta - c - \frac{2p\beta s}{s_{\max}} + \underbrace{\frac{p\beta s^2}{s_{\max}^2} \frac{\partial s_{\max}}{\partial s}}_{\text{ecological feedback}}$
	Stochastic rainfall $r_{\min} < r_{\max}$
Single period $\frac{\partial s_{\max}}{\partial s} = 0$	$\frac{\partial \pi^e}{\partial s} = 0 = p\beta - c - \frac{2p\beta s}{s_{\max}(r_{\max})} \underbrace{- 2p\beta s \int_{r_{\min}}^{r_{\max}} G(r) \frac{\partial s_{\max} / \partial r}{s_{\max}^2} dr}_{\text{stochastic effect}}$
Long-term $\frac{\partial s_{\max}}{\partial s} < 0$	$\frac{\partial \pi^e}{\partial s} = 0 = p\beta - c - \frac{2p\beta s}{s_{\max}(r_{\max})} + \underbrace{\frac{p\beta s^2}{s_{\max}^2} \frac{\partial s_{\max}}{\partial s}}_{\text{ecological feedback}} \underbrace{- 2p\beta s \int_{r_{\min}}^{r_{\max}} G(r) \frac{\partial s_{\max} / \partial r}{s_{\max}^2} dr}_{\text{stochastic effect}}$ $\underbrace{- p\beta s^2 \int_{r_{\min}}^{r_{\max}} G(r) \left(\frac{\partial^2 s_{\max} / \partial r \partial s}{s_{\max}^2} - \frac{2\partial^2 s_{\max} / \partial r \partial s}{s_{\max}^3} \right) dr}_{\text{stochastic ecological feedback effect}}$

8. Discussion and conclusions



8.1 Introduction

This thesis examined how selected complex dynamics can be included in an ecological-economic model, and how they influence the possibilities to manage the ecosystem in an efficient and/or sustainable manner. Complex dynamics are increasingly recognised to be of major importance for ecosystem management (e.g. Holling, 1986; Costanza et al., 1993; Perrings and Pearce, 1994; Carpenter et al., 1999; Mäler, 2000; Scheffer et al., 2001). To structure the research, four research questions have been formulated (chapter 1). These are:

1. How can the efficiency and sustainability of ecosystem management options be analysed ?
2. How do complex dynamics influence the response of the ecosystem to management measures ?
3. How can ecosystem services valuation be applied to analyse ecosystem management options ?
4. How can the management of complex ecosystems be optimised, from an efficiency and sustainability perspective ?

In sections 8.2 to 8.5, the main conclusions of the thesis with respect to these four questions are presented. Each conclusion is followed by a brief discussion of its potential scientific implications and/or its relevance for ecosystem management. In section 8.6, I provide a number of general recommendations for ecosystem management, based upon the results of this thesis. Subsequently, in section 8.7, I present the main recommendations for further research.

8.2 Analysing the efficiency and sustainability of ecosystem management options

Economic and ecological literature comprises a multitude of studies examining the efficiency and sustainability aspects of ecosystem management (Pezzey and Toman, 2002; McMichael et al., 2003). Based upon a literature review, a framework for the assessment of the efficiency and sustainability of management options for dynamic ecosystems has been developed (chapter 2). The framework contains three categories of management options: controlling the harvest/use levels of ecosystem services; limiting pollution loads; and direct interventions in the ecosystem (e.g. through specific mitigation or rehabilitation measures). Both environmental management and ecological processes are included as drivers for ecosystem change. In this section, I evaluate (i) the general applicability of the ecological-economic framework; (ii) how it can be used to assess the efficiency of ecosystem management; and (iii) how it can be used to analyse the sustainability of ecosystem management.

The general applicability of the framework

The case studies conducted in this thesis show that the ecological-economic framework provides a consistent and logical structure to model economy-ecosystem interactions, and to analyse the efficiency and sustainability of management options. Compared to existing frameworks (e.g. Keyzer, 2000; Turner et al., 2000; De Groot et al., 2002; Millennium Ecosystem Assessment, 2003; Turner et al., 2004), an advantage of the framework is that it is capable of accounting for complex ecosystem dynamics. Complex dynamics can be included through a dynamic systems modelling approach. This requires the modelling of the relevant ecosystem components and their interactions. Each component needs to be represented through a normal or differential equation that describes the development of the component as a function of human management and ecological processes. Complex dynamics can be included through the modelling of non-linear and stochastic processes, and feedback mechanisms between the components. For instance, the irreversible response of a forest ecosystem to overharvesting of wood (chapter 3) has been modelled through the inclusion of two feedback mechanisms in the ecological-economic model: a low forest cover increases erosion, and a degraded topsoil reduces the growth of the forest cover. Once certain thresholds in forest cover and topsoil depth have been passed, recovery of the ecosystem is no longer possible. General models describing ecosystem dynamics have been developed for a range of ecosystems (Holling, 1986; Smith, 1990; Scheffer, 1998; Holling et al., 2002). However, the dynamics of individual ecosystems vary as a function of their specific biotic and abiotic structure. Therefore, it is necessary to adapt the appropriate general model to the particular dynamics of the considered system. This requires medium to long term data series (5-20 years) of the dynamics of the ecosystem, depending upon the complexity of the dynamics involved (cf. Grasso, 1998; Settle et al., 2002).

Analysing the efficiency of ecosystem management

This thesis studied the economic efficiency of ecosystem management. This comprised both (i) the analysis of the net present value generated by different ecosystem management options, and (ii) the identification of the economic efficient ecosystem management option. These two aspects are discussed below.

(i) Analysis of the net present value of ecosystem management options. The net present value (NPV) of ecosystem management options depends upon their costs, the benefits they generate and the discount rate used. 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). Benefits depend upon changes in the flow of the complete set of ecosystem services (Turner et al., 2003). Besides ecosystem services valuation (discussed in section 8.4), an important element in the calculation of the NPV of ecosystem management options is the quantification of the flows of ecosystem services.

The thesis shows that the link between the ecosystem state and the supply of ecosystem services needs to be assessed for each ecosystem service separately. For production services, this requires the linking of flows (e.g. wood harvest) to the specific stock involved (e.g. trees). Incorporation of ecological realism in the models requires consideration of potential feedbacks, in particular where the harvest of one ecosystem service (e.g. wood) reduces the supply of another ecosystem service (e.g. non-timber forest products). Most regulation and information services depend upon a combination of components, or upon the ecosystem as a whole. For example, the hydrological service of a forest depends upon a range of biotic (e.g. tree cover, herbaceous layer) and abiotic factors (soil depth, soil porosity, etc.) (Bosch and Hewlett, 1982). This means that the supply of the service needs to be connected to the development of a range of components, or the ecosystem as a whole. The specific methodology applied to link the development of one or more ecosystem components to the supply of an ecosystem service needs to be defined per ecosystem and per ecosystem service. Some services, in particular several of the regulation services (e.g. the hydrological service, protection against floods), have an important spatial aspect. For instance, the value of the flood control service differs with the flood risk, which depends upon a number of factors such as the distance to the water body and the topography. Although not tested in this thesis, spatially explicit modelling will often be required in order to quantify the relation between management and the supply of these ecosystem services (see e.g. Voinov et al., 1999).

(ii) Identification of the efficient ecosystem management option. The efficient option is the option that provides maximum utility, given a certain utility function, and including the benefits of all services supplied by the ecosystem and the costs involved in providing or accessing these services (based upon Chiang, 1992). For a given discount rate, the efficient management option is the option that provides the maximum net present value based upon the current and discounted future flows of net benefits provided by the ecosystem. Two approaches to assessing the efficient option have been applied: a simulation or programming approach, and 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 special algorithms applied in computer software such as GAMS and Mathematica.

The case studies provide some general insights in the potential applicability of both optimisation approaches in the context of this thesis' modelling approach. An advantage of the *algebraic optimisation* approach is that it is more suitable to deal with stochastic ecosystem behaviour. This is illustrated in chapter 7, where the expected values for the income gained from livestock keeping in the Ferlo rangeland in Northern Senegal are calculated. They are calculated with an algebraic optimisation approach, based upon the rainfall probability density function. This results in an estimate of the efficient stocking rate considering the expected rainfall patterns in the coming years. If a simulation approach was used, these two factors could only have been calculated for an assumed rainfall pattern (e.g. by using actual rainfall data for a certain period in the past), which leads to a bias as the outcomes depend upon the selected time series. However, the *simulation approach* deals more easily with non-linearities in ecosystem behaviour and ecological feedbacks, as illustrated in chapters 3 and 5. Solving the first order conditions in the algebraic optimisation approach rapidly becomes highly complex, in particular if the costs or benefit functions are non-linear and/or discontinuous, or if the differentiated cost or benefit function is 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. For the Ferlo, the equations used in the algebraic optimisation approach needed to be specifically adjusted in order to account for this effect (chapter 7). Hence, in the application of algebraic or numerical optimisation procedures, it can be highly complex to solve the first order conditions and determine optimal solutions.

Analysing the sustainability of ecosystem management

In this thesis, 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; Pearce et al., 1989; Barbier and Markandya, 1990). This definition implies 'strong sustainability' (Carter, 2001; Pezzey and Toman, 2002). This section discusses the findings of this thesis with respect to sustainable ecosystem management.

This thesis shows that it is often difficult to define a benchmark to evaluate the sustainability of ecosystem management options, in particular for real-world ecosystems. For the hypothetical forest ecosystem (chapter 3), analysis of the sustainability of ecosystem management options was straightforward. It was assumed that sustainability involves the long-term maintenance of the initial forest and soil cover, which, for this ecosystem, is a sufficient condition to guarantee the sustained supply of the two considered ecosystem services. For the two case studies that involve actual ecosystems, it is much harder to identify a benchmark for sustainability. In the De Wieden wetland, the current state of the lakes is substantially degraded compared to their natural state. Although the surrounding marshlands are rich in biodiversity, the water of the main lakes is turbid, and has a poor water plant and fish community. This is the consequence of the strong increases in nutrient loading into the lakes since the early 1960s (Van Berkum, 1996). Although nutrient loading has decreased again since the mid 1970s, the ecosystem has still not recovered and has not returned to a clear water state (chapter 5). According to the proposed definition, leaving the water quality as it now is would be sustainable. However, if 1960 would have been taken as the baseline condition, rehabilitation would be required to reach sustainability. This would require a reduction in the inflow of nutrients in the system in combination with biomanipulation in order to obtain a clear water ecosystem. 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 stochastic events, such as rainfall and fire (Walker, 1993). Hence, the selection of the baseline year is of major importance in the formulation of sustainable ecosystem management strategies.

Furthermore, the thesis confirms that maintenance of the resilience of ecosystems is an important element in ensuring the sustainability of ecosystem management (cf. Common and Perrings, 1992; Levin et al., 1998). For instance, in the De Wieden wetland (chapter 5), 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 (cf. Carpenter et al., 2001; Brock et al., 2002).

Maintenance of the resilience of this system is a precondition for maintaining the quality of the natural resource base and the supply of services. This also holds for the Ferlo where maintenance of the ecosystem's resilience is required to ensure the productivity of the herbaceous layer, in particular during droughts (chapter 6). The question arises if maintenance of the resilience is a sufficient condition for sustainable development. As already shown by Common and Perrings (1992) in a formal manner, this is not the case. For instance, recent hypotheses state that a loss of biodiversity does not necessarily lead to a loss of resilience of an ecosystem, provided that there are other species in the same functional guilds that can take over the role of the lost species (Walker, 1995; Mageau et al., 1998). However, a loss of species would affect the service 'maintenance of nature and biodiversity' of ecosystems (chapter 2). Therefore, maintenance of the resilience does not necessarily imply sustainability.

Summarising, the efficiency and sustainability of ecosystem management options can be analysed by modelling their impact on the state of the ecosystem and the subsequent implications for the supply of ecosystem services. The efficiency of the options can be revealed by comparison of their costs with the benefits resulting from changes in the supply of ecosystem services (requiring the monetary valuation of ecosystem services). The sustainability of the options can be analysed by examining their long-term impact on the state of the ecosystem inclusive of its resilience for external disturbances. The framework provided in chapter 2 provides an outline of an ecological-economic model capable of such analyses; complex dynamics can be included through the consideration of these dynamics in the modelling of the ecosystem's response to management.

8.3 Impacts of complex dynamics on ecosystem responses to management

In the last decades, there has been ample attention for the potential implications of complex ecosystem dynamics for management (e.g. Holling, 1973; Costanza et al., 1993; Perrings and Walker, 1997; Scheffer et al., 2001; Holling et al., 2002). This thesis considers three types of complex dynamics: (i) irreversible responses; (ii) multiple states and thresholds; and (iii) stochasticity and lag effects. An overview of the implications of these three types of complexities for ecosystem management is provided below. The overview is limited in scope because only one case study per type of complexity has been conducted in this thesis.

Irreversible responses

Irreversible ecosystem dynamics involves changes in ecosystems that can not, or only to a very limited extent, be undone through natural processes (see section 2.2.2). For instance, the extinction of a particular species, or the loss of an ecosystem can be irreversible (Barbault and Sastrapradja, 1995). Irreversibility may also refer to changes in the state of an ecosystem, for example in case of the transition of a rangeland dominated by palatable grasses to one dominated by unpalatable shrubs (Laycock, 1991).

The implications of irreversible ecosystem changes for the optimisation of ecosystem management are demonstrated in chapter 3. It is shown that partial reconciliation of efficiency and sustainability considerations in the management of a *reversible* forest ecosystem is possible through the application of a strategy comprising clear-cut of the forest in the initial years followed by recovery of the ecosystem in subsequent years. In particular if a high discount rate is used, this strategy allows reaching sustainable management at a limited reduction in net present value compared to the efficient strategy, which involves continuous, high harvest rates. However, for an *irreversible* ecosystem, such a strategy is not possible; depletion of the forest ecosystem to below the minimum stock level leads to an irreversible collapse of the system. The minimum sustainable stock level expresses the critical ecosystem condition at which point further degradation leads to an irreversible collapse of the system. For an irreversible ecosystem, a compromise between efficient and sustainable involves maximum harvesting subject to the condition that the minimum sustainable stock levels are maintained. This 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).

Chapter 3 of the thesis also shows that, in the case of strong interdependencies between the different components of an ecosystem, establishment of the minimum stock level needs to be based upon analysis of multiple ecosystem components. This is illustrated by the ecosystem model developed in chapter 3. The model contains two components, forest cover and topsoil, and their interactions. For this system, it is incorrect to assess the minimum sustainable stock level based upon only one variable, such as the forest cover. Specifically, in case of a healthy topsoil, the forest is able to recover from a significantly higher amount of logging than in case of a degraded topsoil. This demonstrates that assessments of the state of an ecosystem using only one indicator can be misleading; its response to management will often depend upon a combination of state indicators. Note that, in the deterministic model developed in this paper, the minimum sustainable stock levels are known with certainty. In reality, of course, this will not always be the case (e.g. Cole, 1954; Smith, 1990). In case of uncertainty, the corresponding concept, indicating the minimum ecosystem stock that should be preserved in order to maintain the functioning of the ecosystem, is the 'safe minimum standard' (SMS), as proposed by Ciricay-Wantrup (1968), modified by Bishop (1978). Conform to 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.

Multiple states and thresholds

Multiple steady states are relatively stable configurations of an ecosystem, characterised by a certain abiotic and biotic configuration (based upon 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 (Scheffer et al., 2002). In chapter 5, multiple states and thresholds have been examined for the De Wieden wetland, the Netherlands. The examined 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 poor fish community dominated by bream (*Abramis brama*) (the actual state). Nutrient loading is the main driver for ecosystem change examined in this case study. In line with the expectations (Hosper, 1997; Meijer, 2000), the model shows that a strong reduction of the current nutrient concentrations would be required to shift the system to a clear water state (from the current 0.1 to some 0.03 mg total-P/litre). A short cut to a clear water state can be made through a modest reduction in nutrient concentration (to 0.09 mg total-P/l) in combination with biomanipulation. Biomanipulation involves catching a substantial part (>70%) of the benthivorous fish in order to force a bifurcation in ecosystem state from turbid to clear water (Meijer, 2000).

The case study demonstrates that the presence of multiple states and thresholds has important consequences for the management of the ecosystem (chapter 5). The system is relatively inert, and the supply of services will not strongly be modified, as long as the threshold is not passed. Hence, in De Wieden, the benefits of a limited reduction in nutrient inflow that does not lead to a switch to clear water, are small. However, at the threshold level, the benefits of eutrophication control rapidly increase. This is illustrated in figure 8.1. The occurrence of the threshold causes the damage cost curve for pollution to be non-convex (cf. Mäler et al., 2003). This results in the presence of a point of minimum total cost (S1) and maximum total costs (S2). Calculation of these points of minimum and maximum total costs is possible with the ecological-economic modelling approach pursued in this thesis. The point of minimum total costs represents the efficient management option.

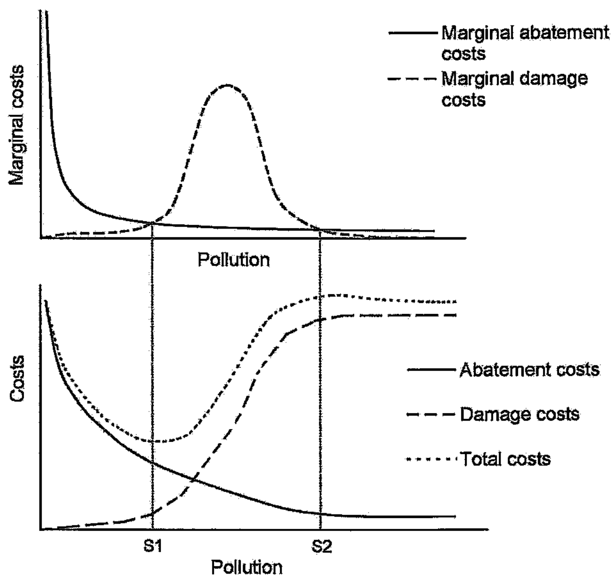


Figure 8.1. Pollution damage and abatement costs: marginal and total costs. The point of minimum total costs (S1) has been calculated in chapter 4 with reference to eutrophication control in the De Wieden wetland.

The thesis also indicates that, due to the threshold effect occurring in the response of shallow lakes to reduced eutrophication loading, the current Dutch policies, as expressed in the 'Fourth National Policy Document on Water Management' (VW, 1998), are not 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/litre. 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 (chapter 5). Furthermore, in case 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 cheaper 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, then through setting a standard for total-P concentrations only. Currently, the Dutch government is revising its national water management policies (Government of the Netherlands, 2004). In view of the above, the Dutch government could consider to increase the allowable total phosphorus concentration in waters with as 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 1 meter (see chapter 5).

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 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 occurrence of an event that is big enough to modify the ecosystem. Stochasticity has been examined for the Ferlo rangeland (chapters 6 and 7). In this ecosystem, stochasticity is caused by the substantial variations in annual rainfall. The impact of rainfall variation depends upon the livestock stocking density. A lag effect was modelled for the De Wieden wetland; it was assumed that the establishment of water plants in De Wieden, characteristic for the clear water phase of the lake, takes several years (cf. Meijer, 2000). In addition, in the Ferlo, there is a lag effect regarding the impact of high grazing pressures on the productivity of the rangeland. Chapter 6 shows that the impacts of long-term, high grazing pressures on the productivity of the Ferlo are only significant during a drought that occurs, on average, around once in 5 to 10 years.

The two case studies conducted in the thesis confirm that stochasticity and lag effects have significant implications for ecosystem management (cf. 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 measures leading 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 case 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 impacts of high grazing pressures on rangeland productivity are most pronounced during drought, and, hence, the rainfall conditions determine when these impacts become apparent. If the timing of a change in ecosystem conditions can not 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.

The case studies also illustrate a second impact of stochasticity and lag effects; 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. The analysis conducted in chapter 6 of the thesis shows that in the Ferlo, grazing significantly affects the functioning and productivity of the rangeland, but that this impact is only significant during dry years. This confirms the hypothesis that high grazing pressures can affect the functioning and productivity of semi-arid rangelands (Le Houérou, 1984; Sinclair and Fryxell, 1985; Illius and O'Connor, 1999), which is a widely debated topic in rangeland literature (Sullivan and Rohde, 2002; Briske et al., 2003).

This also has important implications for management. Whereas 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. As the Ferlo example illustrates, this can strongly reduce the actual impacts of a drought in a particular area. For the Ferlo, improving the resilience would mean partial rehabilitation of the herbaceous cover through reducing the livestock density (note that reduction of the livestock stocking rate also leads to a higher income per hectare for pastoralists, see section 8.5).

8.4 Application of ecosystem services valuation in the context of the pursued modelling approach

In the last decades, substantial advances have been made in the development of valuation techniques (Deacon et al., 1998; Sterner and Van den Bergh, 1998; Turner et al., 2003). As explained in chapter 1, this thesis examines how existing ecosystem valuation techniques can be applied in the context of the pursued ecological-economic modelling approach. Only relatively simple valuation techniques, such as the basic travel cost method, have been used. Ecosystem services valuation has been conducted, in particular, for the De Wieden wetland (chapter 4). For the two production services of De Wieden, reed cutting and fisheries, value estimates were based upon the value added of the sectors involved. The recreation service was valued using the travel cost method, whereas an indication of the minimum value of the nature conservation service was obtained by examining the actual payments of the Dutch public to the NGO managing the site. For the hillside forest ecosystem modelled in chapter 3, and the Ferlo rangeland analysed in chapter 7, fixed respectively variable market prices derived from literature have been used as indicator for the marginal value of the relevant ecosystem services.

The thesis illustrates that, in the context of the Netherlands, the economic value of natural ecosystems can be high compared to the potential value that can be generated by agricultural land-use. For instance, the combined annual economic value generated by the four services provided by the De Wieden wetland is around 830 euro/ha/year (chapter 4). An indication of the annual value generated by nearby agricultural land (in 2001) is provided by the local lease prices of agricultural land: 360 euro/ha/year (LEI and CBS, 2004). The market lease price of agricultural land provides an indication of the net income farmers expect to gain from the land, and hence of the value of the land for the farmer. In order to obtain the economic value of nearby agricultural land, this would need to be corrected for positive and negative externalities, such as contributions to biodiversity conservation by providing breeding opportunities for meadow birds, and negative impacts on water quality through run-off of fertilisers and pesticides. Chapter 4 of the thesis also confirms the importance of non-use values for the overall value of natural and semi-natural ecosystems (cf. Cummings and Harrison, 1995; Ruijgrok, 2002). A first estimate of the minimum non-use value of the De Wieden wetland is around 400 euro/ha/year. Of course, this value strongly depends upon the economic, ecological and social context; in a highly populated, rich country as the Netherlands the non-use value of the remaining (semi-) natural ecosystems can be expected to be relatively high.

The thesis also shows that the value of ecosystem services can vary strongly between stakeholders at different spatial scales, and that this needs to be accounted for in the formulation of spatial and environmental policies (chapter 4). In areas rich in natural capital, local stakeholders experience an abundance of natural resources and biodiversity. They may particularly value the production services of the ecosystem, such as local fisheries or reed cutting in De Wieden. In addition, in the Netherlands, there is often strong pressure from various stakeholders at the municipal level to allow additional residential building in or around protected areas. However, natural capital may be much scarcer when analysed at the national scale and, at this scale, the conservation of biodiversity and landscapes may be the most important services of a protected area. There is a risk that local policy makers do not sufficiently account for national interests in spatial and environmental planning, whereas national policy makers may partly disregard local interests. This discrepancy between the interests of local and national stakeholders needs to be accounted for in setting up decision making structures for spatial and environmental planning. In particular, there may have been insufficient consideration for this discrepancy in the Netherlands' recent spatial planning policy document 'Nota Ruimte' (VROM, 2004). This policy document proposes to shift a substantial part of the decision making responsibilities on land use in protected areas from the national to the municipal level. There is a risk that this leads to sub-optimal land use management from the national perspective, because municipal policy makers can be expected to act primarily according to municipal interests. This means that, contrary to national interests, they may decide to relax the constraints currently in place on land use in protected areas, and let building activities prevail over concerns on biodiversity conservation.

8.5 Optimising the management of complex ecosystems

Three important aspects in the optimisation of ecosystem management are efficiency, sustainability and equity. The previous sections show that the efficiency and sustainability of ecosystem management options can be analysed with an ecological-economic model constructed according to the developed framework. As motivated in chapter 1, equity aspects are not considered in this thesis. In this section, I compare the efficient and the sustainable management option for the three ecosystems studied in this thesis in order to assess how this information provides insight in the optimal management of these ecosystems.

(i) The forest ecosystem. In the hypothetical forest ecosystem (chapter 3), *efficient* management involves a relatively short rotation period, and leads to depletion of the forest resources in the course of decades up to centuries, depending upon the discount rate used. *Sustainable* management requires long-term maintenance of the full forest cover. Hence, there is a substantial discrepancy between the efficient and the sustainable management option. A compromise solution between efficient and sustainable management involves the application of variable rotation periods. Two different types of compromise options have been examined, one applicable to the reversible system, and the other applicable to the irreversible system.

For a *reversible* ecosystem, a period of intensive harvesting followed by a recovery period allows to achieve long-term sustainability at a modest reduction in efficiency, compared to the efficient rotation period. This approach is most suitable if a high discount rate is used. The *irreversible* ecosystem requires maintenance of the minimum sustainable stock levels to avoid an irreversible collapse of the system, and there is less scope to reconcile efficiency and sustainability considerations through variable rotation periods. Maintenance of the minimum sustainable stock levels in an irreversible system provides a compromise between efficiency and sustainability because it leads to a modest reduction in efficiency as well as a modest decline in natural capital stock, and because it allows future generations, at certain costs, to rehabilitate the system. Note that this option requires that the ecosystem manager knows the minimum sustainable stock levels. However, for real-world ecosystems, these levels are often hard to establish due to a lack of information on the precise dynamics of the ecosystem. In this case, a safety margin needs to be applied, with the size of the margin depending upon the accuracy with which the ecosystem manager knows the characteristics of the minimum sustainable stock levels.

(ii) De Wieden. Compared to the natural state, the De Wieden wetland is currently in a degraded, turbid state as a result of eutrophication in the last decades. The most efficient way of achieving clear water in the four main lakes of De Wieden is through reduction of the total-P concentrations in the four lakes from the current 0.10 to 0.09 mg total-P/l, in combination with biomanipulation (chapter 5). Restoration to a clear water state would increase the supply of ecosystem services due to increased opportunities for biodiversity1conservation and recreation. However, these benefits are difficult to quantify. It is shown that the increase in ecosystem services supply needs to be valued at at least 210,000 euro/year in order to justify rehabilitation of the system from an economic perspective. This compares to current expenditures of several millions of euro per year for the management of the area (expenditures of the NGO Natuurmonumenten and of the local Waterboard).

The strong sustainability definition applied in this thesis allows for limited substitution between natural capital (chapter 2). For the De Wieden ecosystem, this means the following. If the sustainability definition is applied at the scale of the ecosystem, there is, strictly speaking, no need to rehabilitate the system as the benchmark is the current state. However, it is more common to define sustainability at a more aggregate level (e.g. Pezzey and Toman, 2002). In this case, the overall decline in natural areas in the Netherlands including wetlands (Bal et al., 2001; CBS and RIVM, 2004) provides an argument to rehabilitate the ecosystem, under the assumption that enhancing biodiversity in De Wieden can partly mitigate the decline in biodiversity elsewhere. Hence, for De Wieden, provided that stakeholders attach a sufficiently high value to the increased opportunities for nature conservation and recreation in clear water, efficiency as well as national sustainability considerations indicate that rehabilitation is the preferred option from the perspective of the Netherlands' society.

(iii) the Ferlo. The impacts of grazing and rainfall variability on the Ferlo rangeland in Northern Senegal are analysed in chapters 6 and 7. It is demonstrated that, in the Ferlo, the impact of high grazing pressures is concentrated in dry years, and that high grazing pressures, therefore, increase the vulnerability of the local population to drought. Furthermore, the thesis shows that important efficiency gains can be reached by reducing the grazing pressure in the Ferlo (chapter 7). The current grazing pressure equals around 0.15 to 0.2 TLU (Tropical Livestock Unit)/ha, whereas maximum per hectare income for livestock owners is reached at a livestock density of around 0.09 TLU/ha. Annual per hectare income for pastoralists is currently on average around 240 CFA[4] compared to the around 460 CFA/ha/year that could be obtained at the efficient stocking density.

With respect to sustainable development, the criterion is that there is no further degradation of the system, but rehabilitation is not required. Hence, efficiency rather than sustainability considerations indicate that, from the perspective of the pastoral society in the Ferlo, it is optimal to reduce grazing pressures by around half, and partially rehabilitate the herbaceous plant cover of the Ferlo. In practise, the open-

[4] 1 euro equals around 620 CFA (August 2004)

access and common property character of the rangeland act as constraints towards achieving efficient management of the ecosystem, and individual pastoralists face no incentive to reduce their livestock herds. The analysis demonstrates that any further government intervention aimed at increasing livestock numbers in the area would be counterproductive, and that, where possible, transformation to (sedentary) livestock systems based upon fewer animals should be promoted in order to reach more efficient management of the Ferlo.

General implications for the optimisation of ecosystem management. The case studies demonstrate that the current stocks of harvestable resources, or the current level of environmental quality may be equal to, below or above the efficient levels. In case the current level of resource stocks or environmental quality *equals* the efficient level, efficient management involves maintaining the stocks at this level (e.g. through ensuring that the harvest levels don't exceed the replenishment levels). In specific cases, sustainability considerations can provide a motivation to increase the natural capital stock, for instance if it concerns a unique ecosystem, and rehabilitation would bring important gains for biodiversity conservation.

For ecosystems in which the current level of natural resource stock or environmental quality is *below* the efficient level, more efficient management requires partial restoration of the ecosystem. This is the case for De Wieden and the Ferlo, and has also been found for many other ecosystems on earth, in particular for open access systems with a long history of human use (McNeely et al., 1995; Balmford et al., 2002). In De Wieden, more efficient management involves restoration of the clear water state of the lake, in the Ferlo it involves reduction of the grazing pressure and partial rehabilitation of the herbaceous cover. For these systems, efficiency and sustainability criteria converge. It is efficient to rehabilitate the ecosystems involved, up to the point where maximum utility is provided, whereas also global sustainability considerations provide an argument to restore these ecosystems (again, assuming that rehabilitation of certain ecosystems may partially compensate for ecosystem losses elsewhere). For systems that are degraded because of market inefficiencies that lead to over-exploitation of the system, management needs to focus on reducing these inefficiencies. This may involve, for instance, incorporating externalities in price mechanisms, attributing property rights, and/or setting up payments mechanisms for ecosystem services - depending upon the economic, ecological and social setting involved (e.g. Baumol and Oates, 1988; Schmid, 1995).

For systems where the current level of environmental quality is *above* the efficient level, efficiency and sustainability criteria do not converge. Efficient management leads to a decrease in the natural capital stock, whereas sustainable management leads to inefficient management of the resource, as in the case of the hypothetical forest ecosystem analysed in chapter 3. In this case, decision making needs to weigh the importance of efficiency, sustainability (and equity) considerations. This requires consideration of the interests and opinions of the various stakeholders in the ecosystem or resource. This may be pursued with, for instance, multi criteria analysis or participatory approaches involving stakeholder dialogues. This has not been studied in this thesis, but further information on multi criteria analysis can be found in, for instance, Nijkamp and Spronk (1979) and Costanza and Folke (1997). Renn et al. (1993) and Beierle (1998) provide more information on the application of participatory approaches for environmental decision making. Of course, following the formulation of the preferred management strategy, the social planner still needs to consider the social and institutional setting, and the presence of market inefficiencies, in order to ensure that the various stakeholders conform to the selected management strategy.

8.6 Recommendations for ecosystem management

Based upon the results of the research conducted for this thesis, a number of considerations and recommendations for ecosystem management are summarised below.

(i) Complex dynamics involve irreversible, non-linear and/or stochastic responses of the ecosystem to management or stress, and they occur in a wide range of ecosystems. Complex dynamics result from non-linear or stochastic ecological processes, and from feedback mechanisms between ecosystem components. To date, complex dynamics have been found in shallow and deep freshwater lakes, semi-arid rangelands, temperate forests, coastal estuaries with seagrass cover, coral reefs, and marine fish stocks.

(ii) Complex dynamics determine the response of the ecosystem to management and need to be accounted for in management plans. The precise implications of complex dynamics for ecosystem management depend upon the type of dynamics involved. In this thesis, I examined: (i) irreversible ecosystem responses; (ii) multiple states and thresholds; and (iii) stochasticity and lag effects.

- *Irreversible responses*. Whereas some ecosystems recover easily from intensive exploitation, for instance because key species return rapidly through the establishment of seedlings, other systems irreversibly change once the system has been degraded up to a certain point. For example, irreversible responses are common in hillside woodlands with highly erodible soils, and they also occur in marine fish stocks. The 'minimum sustainable stock level' expresses the critical ecosystem condition at which point further degradation leads to an irreversible change in the system. Normally, a change in ecosystem state also changes the capacity of the ecosystem to supply services. For instance, following sustained heavy grazing pressures, North American rangelands may irreversibly switch from a system dominated by palatable grasses to one dominated by unpalatable shrubs (Laycock, 1991). Hence, management needs to consider the minimum sustainable stock levels at which point irreversible responses occur, as well as the consequences of these responses for the supply of ecosystem services. Furthermore, as shown in this thesis, the minimum sustainable stock can depend upon sets of ecosystem state indicators instead of one single indicator, in particular if there are strong feedback mechanisms between the different components of the ecosystem. In these cases, information based upon one indicator is not sufficient to assess the state of the ecosystem. For example, when planning logging in an area, it is important to consider, in addition to the forest cover, the status of the soil. If the soil is easily erodible and/or it has been degraded through previous logging activities, the system is likely to respond more strongly to logging compared to a system with a less erodible and/or intact topsoil.

- *Multiple states and thresholds.* Multiple steady states are relatively stable configurations of an ecosystem. The stability is derived from negative feedback mechanisms that tend to maintain the structure and functioning of the ecosystem. At the threshold level, an increase or release in stress can cause a sudden change from one ecosystem state to the next. Multiple states are common in freshwater lakes, coral reefs and coastal estuaries. The response of the ecosystem to management depends upon the conditions of the ecosystem, in particular upon its current state vis-à-vis the threshold level. Whereas the system may initially appear inert, there can be rapid changes in ecosystem state once a threshold is surpassed. Because the supply of ecosystem services is often closely linked to the state of the ecosystem, this can have important consequences for the supply of ecosystem services. Management needs to consider the current state in relation to the threshold levels in the ecosystem, as well as the changes in ecosystem services supply that result from a switch in ecosystem state.

- *Stochasticity and lag effects.* Stochastic effects occur in all ecosystems where randomly occurring events, such as fire or storms, have an important impact on the state of the ecosystem. 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. For instance, this thesis shows that the impacts of high grazing pressures on the productivity of the Ferlo rangeland in Senegal are only noticeable during a drought that occurs, on average, around once in 5 to 10 years. The response of the ecosystem to stochastic effects depends upon its resilience for the particular effect involved. Sustained environmental pressure can lead to a loss of resilience, and increase the impacts of stochastic events. In the Ferlo, high grazing pressures reduce the resilience of the system for drought, aggravating the impacts of drought on livestock and local food security. It is often difficult to quantify changes in ecosystem resilience, as changes in resilience are much less recognisable compared to the direct impacts of major events, such as drought. Nevertheless, because it is often difficult to control stochastic events, it is crucial to maintain or increase the resilience of systems prone to stochastic disruptions in order to partly mitigate the large scale impacts of these events.

Hence, although the impacts of the various types of complex dynamics differ, it is crucial to account for them as they have a large impact on the response of the ecosystem to management. However, whereas the type and general characteristics of the dynamics are, for many ecosystems, known from literature, the precise dynamics of each individual ecosystem are subject to considerable uncertainty. In practical terms, it is recommended that, where funds are available, management considers analysing the dynamics involved, such as minimum sustainable stock levels, locations of thresholds, and/or resiliences to stochastic events, if not in quantitative than at least in qualitative terms (e.g. types of dynamics involved, consequences of sudden changes or stochastic events). Safety margins in ecosystem state vis-à-vis thresholds can be applied to avoid irreversible or sudden changes in ecosystem state. The size of the safety margin to be used depends upon the certainty with which the dynamics are known.

(iii) In environmental costs-benefit analysis, it is necessary to account for complex ecosystem dynamics in order to obtain an accurate assessment of the costs and the benefits of management options. In the framework for ecological-economic assessment developed in this thesis, I distinguish three categories of management options: controlling the harvest/use levels of ecosystem services; limiting pollution loads; and direct interventions in the ecosystem (e.g. through specific mitigation or rehabilitation measures) (figure 2.3). Costs relate to the implementation of the measures, including, for instance, investment and maintenance costs, or the costs of setting and enforcing maximum harvest levels. The benefits of the measures relate to a potential increase in the flows of ecosystem services following implementation of a measure. Environmental cost-benefit analysis needs to account for all relevant ecosystem services and their value. In this thesis, I distinguish three types of ecosystem services: production services (e.g. wood, fish), regulation services (e.g. water purification, carbon sequestration) and cultural services (e.g. recreation, maintenance of biodiversity). For instance, a cost-benefit analysis of an investment in pollution control measures needs to compare the total investment costs (including discounted operation and maintenance costs, etc.) with the benefits stemming from an increased supply of ecosystem services following the cleaning up of the ecosystem. In an aquatic system, these may include, for instance, the provision of increased opportunities for recreation, fisheries and/or biodiversity conservation. Hence, cost-benefit analysis involves two main steps: (i) analysis of the impacts of the measure on the supply of ecosystem services; and (ii) valuation of the changes in ecosystem services supply. Complex dynamics determine the response of the ecosystem to management measures, and need to be accounted for, in particular, in the first step of the analysis.

- *Analysis of the impact of the measure on the supply of ecosystem services.* Accurate assessment of the benefits of ecosystem management options requires analysis of the causal change 'management measure' – 'changes in ecosystem state' – 'changes in the supply of ecosystem services'. In some cases, establishment of the relation between the management measure and changes in the supply of ecosystem services is straightforward, for instance in case of the harvesting of a stock that can be accurately described by a simple logistic growth curve. However, for many ecosystems, there are interactions between different components of the ecosystem, and simple logistic growth curves are not sufficient to obtain an accurate picture of the impact of the measure. In this case, dynamic systems modelling on the basis of the proposed framework allows for a more accurate assessment of the benefits of ecosystem management measures. This requires that different ecosystem components, and their interactions are described by sets of (differential) equations. Ecological literature indicates that many ecosystems are primarily driven by a small set of drivers, which implies that models need to contain only a limited set of processes in order to capture the main dynamics of the ecosystem (Harris, 1999; Holling et al., 2002). General models of ecosystem dynamics can, for many ecosystems, be derived from literature. However, there are large differences in the responses of individual ecosystems to management, and these general models need to be specified for the particular ecosystem involved.

- *Valuation of the changes in ecosystem services supply.* Economic valuation of changes in the supply of ecosystem services is required in order to assess the net benefits of ecosystem management measures. For private ecosystem services, in principle, value estimates can be based upon their market price minus the costs related to producing or accessing the service. For public services, a range of alternative valuation methods has been developed (see chapter 2). Although substantial progress has been made in the development of valuation methodologies in the past two decades, there are still important constraints to ecosystem services valuation. This also constrains the application of the framework developed in this thesis. A main issue, that is particularly relevant for ecosystem management, is the monetary valuation of non-use values using contingent valuation methods (Diamond and Hausman, 1994; Deacon et al., 1998). If monetary valuation of all services is not feasible, there are several alternative approaches that can be used in environmental cost-benefit analysis: (i) using non monetary indicators for selected ecosystem services (see e.g. Strijker et al., 2000); (ii) using participatory approaches to establish value estimates (Renn et al., 1993); or, as applied in this thesis, (iii) perform the cost-benefit assessment for a range of value estimates. This latter approach allows each stakeholder to identify the net present value of ecosystem management options based upon the value he attributes to the ecosystem services involved. It also allows calculating the cut-off values that would justify implementation of the management measure from an economic perspective (see chapter 5 for details).

(iv) Identification of the efficient ecosystem management option requires consideration of the specific dynamics of the involved ecosystems. Efficient ecosystem management involves maximising the net present value of the ecosystem, while accounting for the present and discounted future flows of all ecosystem services. The discount factor to use in natural resources management is still debated, suggestions range from 2 % (Freeman, 1993; Solow, 1993) to 6% (Nordhaus, 1994) - see chapter 2. The efficiency of management options is significantly influenced by complex dynamics. In case of *irreversibility,* there is no possibility to restore the initial state following a modification of the ecosystem. This is particularly relevant if there is a likelihood that the future price (or marginal value) of the supplied ecosystem services will differ from the current (e.g. because of an increasing scarcity of the commodities supplied by the ecosystem). The principal impact of *multiple states and thresholds* is that there are clear point of local maximum and minimum utility that correspond to different ecosystem states. Management needs to weigh, beforehand, the benefits of the ecosystem in each state with the costs of maintaining or bringing the ecosystem in that state. For instance, for the Dutch 'De Wieden' shallow lake ecosystem, the manager needs to compare the costs of eutrophication control measures that result in clear water in the lakes with the benefits obtained from clear water (chapter 5). Based upon this comparison, the manager should chose to pursue clear water, or to maintain the current, turbid water phase. A compromise solution of reducing eutrophication inflow without obtaining a shift in ecosystem state is expensive but yields very little benefits. The principal impact of *stochasticity and lag effects* is that the manager needs to weigh the costs of maintaining the resilience of the ecosystem for stochastic events with the costs of adapting to, or mitigating the impacts of stochastic events. For instance, in the Ferlo rangeland in Northern Senegal, reduction of the livestock density and partial rehabilitation of the grass cover of the system would significantly reduce the impacts of drought in terms of livestock mortality and food shortages (and also increase the pastoralists' income, see chapters 6 and 7).

(v) Analysing sustainability requires careful consideration of the benchmark level, as well as the scale of the analysis. There are a number of different interpretations of sustainable management (Carter, 2001; Pezzey and Toman, 2002). In this thesis, 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'. In the assessment of sustainable use levels, it is particularly important to consider irreversibility, as this reduces the possibilities to rehabilitate the ecosystem following degradation. Other important aspects in defining sustainable management levels are the benchmark used to evaluate sustainability, and the scale at which sustainability is analysed. Application of the sustainability definition provided above (as well as many other sustainability definitions) implies that the current state of the ecosystem is the benchmark to assess the sustainability of future ecosystem use. However, the ecosystem state may be subject to strong interannual fluctuations as a result of stochastic drivers, such as annually varying rainfall. Furthermore, a system may be significantly degraded as a result of past management - which means that, strictly speaking, there is no need to rehabilitate the ecosystem to reach sustainability. It is, therefore, common to analyse sustainability at a more aggregated level, for instance in relation to the natural capital present in a country. The strong sustainability definition applied in this thesis allows for partial compensation of the loss of natural capital by an increase in natural capital elsewhere (see chapter 2). Hence, the sustainability of the management of an ecosystem should be analysed in relation to the overall trends in the natural resource base in a country. If a country is subject to a continuing loss of natural capital, the rehabilitation of one system contributes to more sustainable management at the national scale.

(vi) Optimising ecosystem management requires information on the efficiency and sustainability of management options, and, in case the two consideration do not converge, balancing of these two aspects in decision making. For some ecosystems, efficiency and sustainability criteria converge; more efficient management requires partial restoration of the ecosystem. This is the case in the De Wieden wetland and in the Ferlo rangeland. In De Wieden, the costs of rehabilitation of the lakes to a clear water state are modest in comparison with the current expenditure on the management of the area, and rehabilitation leads to important benefits for nature conservation and recreation. In the Ferlo, the income earned per hectare can be doubled by reducing livestock numbers to around half the current stocking rate. The same has been found for many other ecosystems on earth, in particular for open access systems with a long history of human use (McNeely et al., 1995; Balmford et al., 2002). In case efficiency and sustainability criteria do not converge, decision making needs to consider and weigh the importance of these different aspects. This may be pursued with, for instance, multi criteria analysis (Nijkamp and Spronk, 1979; Costanza and Folke, 1997) or participatory approaches (Renn et al., 1993; Beierle, 1998). Note that a third aspect of optimisation, equity considerations, have not been examined in this thesis. This thesis has taken the perspective of the social-planner, which involves identifying optimal management strategies from the perspective of the society. In practice, in particular in the case of open access and/or common property resources, individual decision makers can face very different incentives regarding the management of the system. This means that the manager also needs to consider strategies that motivate or enforce individuals to comply with the optimum management strategies. This has not been examined in the thesis, but this involves, for instance, incorporating externalities in price mechanisms, attributing property rights, and/or setting up payments mechanisms for ecosystem services - depending upon the economic, ecological, social and institutional setting involved (e.g. Baumol and Oates, 1988; Schmid, 1995).

8.7 Recommendations for further research

The main objective of this thesis was to analyse the implications of complex ecosystem dynamics for the efficient and sustainable management of ecosystems. The thesis resulted in a framework that can be used to support ecosystem management, additional insights in the implications of complex dynamics, and concrete recommendations for the enhanced management of two ecosystems (De Wieden and the Ferlo).

This thesis, of course, only presents one further step in the development of ecological-economic methodologies that can be used to support natural resource management. This section presents a number of recommendations for further research in the field of ecological-economic assessment and modelling. The recommendations are based upon the conclusions and discussion presented in the previous sections of this chapter, as well as the research needs identified in the other chapters of this thesis.

(i) Assessing the spatial heterogeneity of selected ecosystem services. In view of the spatial heterogeneity of several ecosystem services (e.g. the hydrological service), there is a need to further examine the spatial aspects of ecosystem services. This relates both to the spatial variation with which the service is provided, and to the impact of spatial variability on the values of the services. Detailed analysis of these spatial aspects requires the use of GIS in order to model and analyse the relevant services. An interesting topic in this field is to connect the ecological-economic framework developed in this thesis to a GIS in order to develop a methodology for assessing the efficiency and sustainability of management options for ecosystems with strong spatial heterogeneity.

(ii) Dealing with spatial scales of ecosystem services in ecosystem management. Ecosystem processes and services are typically most strongly expressed, or have the strongest impacts at specific spatial scales (Millennium Ecosystem Assessment, 2003). Consequently, there may be large differences in the value attributed to ecosystem services at different spatial scales, as illustrated in this thesis for the services supplied by the De Wieden wetland. This is a major issue in ecosystem management, as stakeholders at local scales often face very different costs and benefits of ecosystem management compared to national or global stakeholders. This issue is particularly relevant for the management of a number of ecosystems of global importance, such as tropical forests and coral reefs. Therefore, for systems that face strong discrepancies between the costs and benefits of ecosystem management, there is a need to further examine how local stakeholder can be compensated for their contribution to the supply of global services. A principal approach for this is the setting up of mechanisms to promote payments for global ecosystem services to local stakeholders, as currently pioneered by, for example, the World Bank Prototype Carbon Sequestration fund (Lecocq, 2004). Further research on this issue is required, for instance in order to examine how local stakeholders can also be compensated for other ecosystem services of global importance (e.g. conservation of biodiversity). In addition, it is necessary to examine how mechanisms can be set up to promote payments for the opportunity costs related to maintaining *existing* ecosystems. For instance, the prototype carbon fund only facilitates payments for projects that lead to additional carbon sequestration (and, hence, not for the protection of existing forests). In addition, it is necessary to examine how potential conflicts between the supply of global services can be accounted for. This relates, in particular, to two of the main global ecosystem services, carbon sequestration and biodiversity conservation, as plantations of fast growing trees tend not to provide a valuable habitat to biodiversity.

(iii) Dealing with the long-term benefits of ecosystems in decision making. Ecosystem services are supplied over a range of temporal scales, varying from the short term impact of the pollination service to the long term impact of carbon sequestration. Assessment of the current value of the services supplied at the long term is often complex, as there may be considerable uncertainty related to the future impacts of the services as well as future preferences for the services. In addition, there is still debate on the discount rate to be used in net present value assessments. Although the use of low discount rates (e.g. 2 to 3%), as recommended by Freeman (1993) and Solow (1993), increases the consideration of the interests of future generations, it still leads to a rapid depreciation of future benefits. This leads to the question of how efficiency and sustainability criteria can be compared in decision making. This has been pursued, for instance, through participatory approaches such as stakeholder dialogues (e.g. Renn et al., 1993). More research is needed on these issues in order to enhance the consideration of long term benefits of ecosystems in decision making processes.

(iv) Incorporation of non-use values in ecosystem services valuation. The thesis illustrates that non-use values can represent a significant part of the total value of an ecosystem. However, this type of value is difficult to quantify, and therefore often not fully considered in cost-benefit analyses. As already indicated by, for example, Deacon et al. (1998) and Hanley (1999), there is a need to further examine how these values can be analysed and quantified, or how they can be included in environmental planning through participatory processes and/or multi-criteria analysis. This relates in particular to global non-use values, for example the non-use value of protected areas harbouring globally important biodiversity. As global stakeholders are often not aware of the characteristics of the biodiversity contained in such parks, it is complicated to obtain value estimates using contingent valuation methods.

(v) Assessment of complex dynamics in different ecosystems. The existence of complex dynamics, such as multiple steady states and irreversible responses, has been demonstrated for a range of ecosystem types. However, it is often complicated to quantify the threshold and minimum sustainable stock levels for specific ecosystems (e.g. Muradian, 2001). In order to enhance the understanding of ecosystem dynamics as well as to support the formulation of ecosystem management strategies, there is a need to further examine complex dynamics. Among others, this involves assessing how complex dynamics, such as thresholds, depend upon system characteristics and management variables of specific ecosystems.

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Summary

Ecosystems supply a wide range of goods and services to mankind. This includes, for example, timber supplied by forests, and animal feed supplied by rangeland systems. In addition, ecosystems supply a range of essential life support services, such as the regulation of climatic and biochemical processes, and they have a non-use value related to their importance for the conservation of global biodiversity. As a consequence of a growing global population and expanding economic activities, human pressure on ecosystems has been progressively increasing. Currently, many ecosystems have been modified by, for instance, pollution or the overharvesting of resources. Hence, in view of the importance of ecosystems for mankind, there is a need to ensure the optimal management of the current stock of ecosystems.

Two key elements in optimal ecosystem management are economic efficiency and sustainability. Economic efficient management can be interpreted as management that maximises the present value of the net present and future benefits derived from ecosystem services. Sustainable management is, in this thesis, interpreted as management that maintains the capacity of ecosystems to provide future generations with the amount and type of ecosystem services at a level at least equal to the current capacity. Both the efficiency and the sustainability of ecosystem management has been studied intensively in the past four decades, in particular in the fields of ecology and environmental and ecological economics.

The traditional models that analysed the responses of ecosystems to human management assumed that ecosystems respond in a gradual, reversible manner to stress. However, increasingly, it is now recognised that many ecosystem show much more complex dynamics, such as irreversible responses to stress, multiple states separated by thresholds, and stochastic behaviour. Such complex dynamics have been observed in, for instance, marine fish stocks, freshwater lakes, woodlands, rangelands, coral reefs and coastal estuaries. These dynamics have important implications for ecosystem management; they determine the response of the ecosystem to human pressures as well as rehabilitation measures. In view of the emerging insights in ecosystem dynamics, there is a need to further analyse the implications of complex dynamics for ecosystem management.

The objective of this thesis is to examine the implications of complex ecosystem dynamics for the efficient and sustainable management of ecosystems. Specifically, it is examined how complex dynamics can be included in ecological-economic modelling, and how efficient and/or sustainable management can be effected for ecosystems with complex dynamics. In chapter 2 of the thesis, a framework is developed that can be used to analyse the economic efficiency and sustainability of management options for ecosystems subject to complex dynamics. The framework includes three main types of interactions between the economic system and ecosystems, and three types of ecosystem management options. Application of the framework requires the modelling of the causal chain 'ecosystem management' – 'ecosystem state' – 'supply of ecosystem services'. The efficiency of ecosystem management options can be derived from a comparison of the costs of the management options and the benefits resulting from changes in the supply of

ecosystem services. The sustainability can be assessed on the basis of the long-term dynamics of the ecosystem. The approach is based upon dynamic systems modelling, using sets of connected (differential) equations. In order to reach an enhanced degree of ecological realism in the models, the models contain different ecosystem components as well as the interactions between these components. The thesis contains three case studies of ecosystems subject to different complex dynamics: (i) a hypothetical forest ecosystem; (ii) the De Wieden wetland (the Netherlands); and (iii) the Ferlo semi-arid rangeland (Senegal).

(i) The forest ecosystem. In chapter 3, the implications of irreversible ecosystem responses for the efficient and sustainable management of a hypothetical forest ecosystem are examined. The forest supplies two services, wood and erosion control, and contains two components: forest cover and topsoil. The management variable is the rotation period applied to harvest wood. The paper first presents a general model, based upon the Faustmann model that has been widely used to examine the efficiency of forest management. Subsequently, it demonstrates the implications of pursuing efficient and sustainable forest management for a forest ecosystem in case of a reversible respectively an irreversible response to intensive harvesting. The paper shows how pursuing efficiency or sustainability can lead to the selection of very different management strategies. The possibilities to combine efficiency and sustainability considerations in management are influenced by the (ir-)reversibility of the ecosystem's response to harvesting.

(ii) The De Wieden wetland. Chapters 4 and 5 focus on the De Wieden wetland. Chapter 4 examines the four main ecosystem services supplied by the De Wieden wetland, and the spatial scales at which they are supplied to stakeholders. These services are reed cutting, fisheries, recreation, and the habitat service. Using different valuation methods, an economic value of the four services is provided, and it is examined how much of this value accrues to stakeholders at the municipal, provincial, national and international scale. The chapter shows that the four services generate a combined annual value of around 830 euro/ha/year. This is high compared to the value generated by surrounding agricultural land, which can be estimated at around 300 to 400 euro/ha/year. The analysis also shows that stakeholders at different spatial scales can have very different interests in ecosystem services. In De Wieden, reed cutting and fisheries are only important at the municipal scale, recreation is most relevant at the municipal and provincial scale, and the habitat service is important at, in particular, the national and international level. Consequently, stakeholders at different scales may have very different interests in the management of the ecosystem, which needs to be considered when valuation of ecosystem services is applied to support the formulation or implementation of ecosystem management plans.

In chapter 5, the costs and benefits of eutrophication control in the four main lakes of De Wieden are analysed. These shallow lakes have two steady states separated by a threshold. The two states are a clear water state and a turbid water state; the switch from one state to the next is sudden once a certain threshold in nutrient concentrations in the lakes is passed. Currently, the lakes are in a turbid water phase, as a consequence of the high nutrient loading in, in particular, the 1960s and 1970s, and the local authorities are considering rehabilitation of the lakes. In chapter 5, an ecological-economic model is developed that describes the response of the lakes to reduced nutrient loading, without and with biomanipulation. The model is used to determine the economic optimal level of eutrophication control – including the costs of the eutrophication control measures and the benefits resulting from an increased supply of ecosystem services following a change to clear water in the lakes. The model shows that an approach with biomanipulation is substantially more cost-effective than trying to reach clear water through reductions in nutrient loading alone. A clear water state can be reached most cost-effectively by reducing the yearly inflow of total-P with 2 ton per year in combination with biomanipulation. The incremental benefits of an increased supply of the recreation and habitat service need to be valued at at least 210,000 euro per year in order to justify, from an economic perspective, rehabilitation of the lakes.

(iii) The Ferlo semi-arid rangeland. The Ferlo in Northern Senegal is studied in chapters 6 and 7. In chapter 6, it is examined how grazing and rainfall variability influence the productivity of the herbaceous layer in the rangeland. Data are based upon a ten years grazing experiment conducted in the Widou-Thiengoly catchment in the Ferlo, Northern Senegal. This experiment examined the impacts of a high (0.15-0.20 tropical livestock units per ha) and a medium (0.10 tropical livestock units per ha) grazing pressure – with the high grazing pressure corresponding to the current grazing pressure in the Ferlo. The paper shows that species composition, biomass production and rain-use efficiency differ markedly for the two grazing regimes – and that this impact is concentrated in years with low rainfall. The impact of high grazing pressures on the productivity of the Ferlo is hardly noticeable in years with normal or above normal rainfall. However, in dry years, both biomass production and rain-use efficiency are significantly reduced in the plots subject to a high grazing pressure. The findings have important implications for the management of rangelands: the experiments indicate that high grazing pressures may increase the vulnerability of rangeland ecosystems and local people to droughts.

In chapter 7, an ecological-economic model of the Ferlo is constructed that includes two drivers for ecosystem change: grazing pressure and stochastic rainfall variability. The impacts of grazing pressure are modelled conform the analyses conducted in chapter 6. The chapter presents an ecological-economic model that captures the impacts of grazing, as well as a case study of the Ferlo rangeland. The theoretical model shows that accounting for the impact of grazing on the herbaceous layer leads to a lower optimal stocking rate compared to a model that does not take this feedback into account. On the basis of the model, the long-term optimal livestock stocking rate for the Ferlo is calculated. The case study for the Ferlo also accounts for fluctuations in livestock prices; livestock prices strongly decrease during a drought when many pastoralists want to sell their livestock because of animal feed shortages. With the constructed ecological-economic model, it is shown that the economic optimal long-term stocking rate for the Ferlo is around half the current stocking rate. Reduction of the stocking rate to 0.09 tropical livestock unit per hectare would increase the per hectare income of the pastoralists from the current 237 CFA/ha/year to around 460 CFA/ha/year. Government actions should be aimed at reducing the livestock density in the area rather than promoting further expansion of the herd sizes.

The thesis demonstrates the importance of considering complex dynamics in cost-benefit and sustainability analysis of ecosystem management options. Complex dynamics determine how the state of the ecosystem, and its capacity to supply ecosystem services, changes following the implementation of management measures.

Application of the developed ecological-economic modelling approach can lead to concrete recommendations to support the formulation of economic efficient and/or sustainable ecosystem management strategies. In the De Wieden wetland, the benefits of eutrophication control measures largely depend upon their capacity to cause a switch from the current turbid water state to a clear water state. The developed ecological-economic modelling approach allows comparison of the costs and the benefits of these measures. In the Ferlo rangeland, the dynamics of the herbaceous layer depend upon the stochastic rainfall conditions in combination with the impact of grazing. With the ecological-economic modelling approach, the optimal livestock stocking density can be calculated. Based upon these results, it is anticipated that the framework and the modelling approach can also be applied to support the formulation of optimal management strategies for other ecosystems subject to complex dynamics.

Résumé en français

Les écosystèmes fournissent une large gamme de biens et services à l'humanité. Cela inclus, par exemple, la production de bois, de produits forestiers non ligneux et de ressources pharmaceutiques produites dans les systèmes forestiers. Les écosystèmes fournissent également un ensemble de services essentiels au maintien de conditions environnementales favorables, comme la régulation du climat et des cycles biochimiques. Ensuite, ils ont une valeur de non-usage reflétant l'importance de conserver leur biodiversité. En raison d'une population mondiale croissante et d'une expansion continue des activités économiques, la pression exercée sur les écosystèmes n'a eu de cesse de croître. La plupart des écosystèmes ont déjà été modifiés, par exemple par la conversion de terres ou la surexploitation des ressources naturelles. Ainsi, si l'on considère l'importance des écosystèmes pour l'humanité, il est essentiel d'assurer une utilisation optimale des ressources naturelles existantes.

Deux éléments clés en ce qui concerne la gestion optimale des écosystèmes sont le rendement économique et la durabilité. L'optimisation du rendement économique peut être interprétée comme étant la maximisation de la valeur actuelle des bénéfices nets, présent et future, fournis par les services des écosystèmes. La gestion durable est définie, pour cette thèse, comme étant la gestion permettant le maintient des capacités d'un écosystème à fournir aux générations futures des services équivalent, en qualité égale, par rapport à ceux fournis actuellement. L'optimisation du rendement économique et la durabilité de la gestion des écosystèmes ont été étudiés de façon intensive, en particulier dans le cadre de l'écologie, de l'économie environnementale et de l'économie de l'écologie.

Les modèles traditionnels permettant l'analyse des impacts de la gestion humaine sur les écosystèmes assument que ceux-ci répondent progressivement et d'une façon réversible aux stress. Cependant, il est maintenant de plus en plus reconnu que la plupart des écosystèmes ont des dynamiques beaucoup plus complexes telles que l'irréversibilité suite à un stress, des changements d'états séparés par des seuils, et des comportements aléatoires. Des dynamiques aussi complexes ont été observées par exemple dans des populations de poissons marins, dans des lacs d'eau fraîche, sur des terrains boisés, des savanes semi-arides, des récifs coralliens et des estuaires. Ces dynamiques ont d'importantes conséquences en ce qui concerne le management des écosystèmes. Elles déterminent les réponses des écosystèmes par rapport aux pressions humaines et par rapport aux mesures de réhabilitation potentielles. Considérant ces idées émergentes de dynamiques complexes des écosystèmes, il est nécessaire d'analyser en détail les implications de ces dynamiques pour la gestion des écosystèmes.

L'objectif de cette thèse est d'examiner les implications des dynamiques écologiques complexes pour la gestion économique efficace et durable des écosystèmes. Afin d'analyser ces implications, une approche de modélisation écologique-économique est développée. Cette approche permet de formuler des stratégies de gestion efficace et/ou durable pour des écosystèmes spécifiques. Dans le chapitre 2 de la thèse, le cadre d'analyse (framework) pour la modélisation est développé. Ce cadre d'analyse inclut trois types d'interactions majeures entre le système économique et les

écosystèmes ainsi que trois types d'options principales pour la gestion des écosystèmes. L'application de ce cadre d'analyse requiert la modélisation de la chaîne d'interactions: 'gestion de l'écosystème', 'état de l'écosystème', 'services fournis par l'écosystème'. L'efficacité économique des options pour la gestion des écosystèmes peut être calculée en comparant leurs coûts (investissement, entretien, etc.) et leurs bénéfices résultant des modifications de services fournis par l'écosystème. La durabilité peut être estimée en se basant sur les dynamiques à longs termes de l'écosystème. Les modèles développés dans la thèse sont du type 'modèles de systèmes dynamiques' (dynamic systems models), basés sur un ensemble d'équations différentielles connectées. Afin d'obtenir un degré avancé de réalisme écologique, les modèles contiennent différentes composantes d'un écosystème, ainsi que les interactions entre ces composantes. La thèse contient trois études de cas sur des écosystèmes sujets à des dynamiques complexes : (i) un écosystème forestier hypothétique, (ii) la zone humide De Wieden (Pays-Bas); (iii) la savane semi-aride de Ferlo (Sénégal).

(i) L'écosystème forestier. Le chapitre 3 examine les conséquences des réponses irréversibles pour la gestion efficace et durable d'un écosystème forestier hypothétique. La forêt fournit deux services, le bois et le contrôle de l'érosion. Elle contient deux composantes : la couverture forestière et le sol. La période de rotation pour la récolte du bois est choisi comme variable de gestion. Tout d'abord, un modèle général est présenté, basé sur les modèles de Faustmann qui ont été largement utilisé pour étudier le rendement économique de la gestion forestière. Par la suite, le chapitre montre les implications économiques et la durabilité de différentes périodes de rotations, dans le cas ou l'écosystème forestier répond de façon réversible à une récolte intensive et dans le cas ou l'écosystème répond de façon irréversible. Il est présenté comment l'efficacité économique ou la gestion durable peut mener à la sélection de stratégies de gestion différentes. Il est aussi démontré que la combinaison de l'efficacité et durabilité, en matière de gestion, est influencée par la notion de réversibilité de la réponse des écosystèmes à leur exploitation.

(ii) La zone humide de De Wieden. Les Chapitres 4 et 5 étudient la zone humide de De Wieden, constituée de lacs d'eau fraîche, de marais et de zones d'agriculture extensive. Le chapitre 4 examine les quatre principaux services fournis par la zone humide de De Wieden. Ces services sont (i) la coupe de roseaux; (ii) la pêche; (iii) les loisirs et le tourisme; et (iv) la provision des habitats pour la biodiversité. En utilisant différentes méthodes d'évaluation, une valeur économique est attribuée à chacun de ces quatre services. Il est ensuite étudié à quel niveau, local, régional, national ou international, les bénéfices économiques de ces services sont reportés sur les principaux acteurs. Le chapitre montre que ces quatre services génèrent une valeur annuelle combinée de 830 euro/ha/an. Cette valeur est importante si l'on compare à la valeur générée par les terres agricoles aux alentours, qui peut être estimée à environ 300-400 euro/ha/an. L'analyse montre également que les acteurs, à différents niveaux, peuvent avoir des intérêts très variés en ce qui concerne les services des écosystèmes. A De Wieden, la coupe de roseaux et la pêche sont essentiellement important à une échelle locale. Les loisirs et le tourisme ont une réelle importance au niveau local et régional, quant au service ''habitat'', il est plus important à l'échelle nationale et internationale. Par conséquent, les acteurs à différents niveaux ont des intérêts très différents quant au management de l'écosystème – ce qui doit être considéré dans la formulation des stratégies de gestion des ressources.

Dans le chapitre 5, les coûts et bénéfices du contrôle de l'eutrophisation dans les quatre principaux lacs de De Wieden sont analysés. Ces lacs peu profonds ont deux états d'équilibres, séparés par un seuil. Ces deux états sont soit une eau claire, soit une eau trouble. Le passage d'un état à un autre est très rapide dès lors qu'un certain seuil de concentration en éléments nutritifs a été atteint dans les lacs. Ils sont actuellement dans un état d'eau trouble en raison de l'augmentation de la quantité d'éléments nutritifs, en particulier au cours des années 1960-70. Les autorités locales envisagent une réhabilitation des lacs. Dans le chapitre 5, un modèle écologique économique est développé afin de décrire la réponse des lacs à une diminution de la quantité d'éléments nutritifs, avec et sans biomanipulation. Le modèle est utilisé pour déterminer le niveau économique optimal pour le contrôle de l'eutrophisation. Le modèle prend en compte les coûts des mesures de contrôle de l'eutrophisation et les bénéfices résultant des services des écosystèmes lorsque la qualité de l'eau augmente. Le modèle montre qu'il est efficace d'utiliser la biomanipulation pour atteindre l'état d'eau claire. Le meilleur rapport coût-efficacité, permettant d'atteindre l'état d'eau claire, est de réduire les apports totaux en élément P de deux tonnes par an en combinant avec la biomanipulation. Les bénéfices progressifs d'une amélioration des services de De Wieden doivent être évalués à au moins 210 000 euro par an afin de justifier, d'un point de vue économique, une réhabilitation des lacs.

(iii) La savane semi-aride de Ferlo. Le Ferlo, situé dans le Nord du Sénégal, est étudié dans les chapitres 6 et 7. Dans le chapitre 6, il est mis en évidence comment le pâturage et la variabilité des précipitations influencent la productivité de la couche herbacée dans la savane. Les données sont basées sur une expérimentation de pâturage de dix ans, conduite dans le bassin de Widou-Thiengoly, dans le centre de la région du Ferlo. Cette expérimentation a examiné les impacts d'un pâturage intensif (0.15-0.20 unité bovin tropical /ha) et moyen (0.10 unité bovin tropical /ha). Les valeurs intensives correspondent au type de pâturage actuel dans la région du Ferlo. Le chapitre montre que la composition des espèces, la production de matière organique, et l'efficacité à utiliser l'eau des précipitations diffèrent largement en fonction du régime de pâturage. Notamment, les impacts sont les plus importants au cours des années moins pluvieuses. Durant ces années sèches, la production de matière organique et l'efficacité à utiliser l'eau des précipitations sont réduits de façon importante sur les parcelles sujettes à un pâturage intensif. Par conséquent, l'impact d'un tel pâturage sur la productivité du Ferlo n'est pas mis clairement en évidence durant les années de précipitations normales ou pluvieuses. Cependant la productivité de ces zones de savane peut être lourdement affectée durant les années sèches. Ces résultats ont d'importantes implications pour la gestion des zones de savane. Les analyses indiquent qu'un pâturage intensif pourrait augmenter la vulnérabilité des écosystèmes et des populations locales pour la sécheresse.

Dans le chapitre 7, un modèle écologique-économique du Ferlo est réalisé incluant deux facteurs influençant les changements de l'écosystème: la pression exercée par le pâturage et la variabilité aléatoire des précipitations. Le chapitre présente un modèle écologique-économique permettant d'estimer le chargement en bétail optimal sur les savanes, ainsi qu'une étude de cas de la savane du Ferlo. Les impacts exercés par le pâturage sont modélisés selon les analyses réalisées dans le chapitre 6. Sur la base du modèle, le chargement optimal en bétail à long terme est calculé pour le Ferlo. L'étude de cas du Ferlo prend également en compte les variations de prix du bétail; les prix diminuent largement durant une période de sécheresse due au fait que les éleveurs souhaitent vendre les animaux qui ne peuvent être nourrit correctement. En utilisant le modèle, il est démontré que le chargement en bétail optimal est une demi-fois égal au chargement actuel du Ferlo. La réduction du pâturage de 0.15-0.20 unité bovin tropical /ha (chargement actuel) à 0.09 unité bovin tropical /ha augmentera les revenus des éleveurs de 239 FCFA/ha/an (revenues actuelles) à environ 460 FCFA/ha/an. Par conséquent, les actions gouvernementales devraient permettre de réduire la densité d'animaux dans la région plutôt que de promouvoir l'expansion de la taille des élevages.

Cette thèse démontre ainsi l'importance de la prise en compte des dynamiques écologiques complexes dans l'analyse de l'efficacité économique et de la durabilité des options de gestion des écosystèmes. Ces dynamiques complexes déterminent l'évolution de l'état d'un écosystème et sa capacité à fournir des services, en fonction de l'application des mesures de gestion. Des modèles écologique-économiques, construis sur la base du cadre d'analyse développé dans cette thèse, peuvent mener à des recommandations concrètes permettant d'améliorer la gestion des ressources naturelles. Dans la zone humide de De Wieden, l'approche d'une modélisation écologique-économique a permis de comparer les coûts et les bénéfices des mesures de contrôle de l'eutrophisation, et d'identifier la stratégie de réhabilitation la plus efficace. Dans la savane semi-aride de Ferlo, les dynamiques des pâtures dépendent des conditions aléatoires des précipitations combinées avec l'impact du pâturage. Avec l'approche d'un modèle écologique-économique, le chargement optimal en bétail peut être calculé. A partir de ces résultats, il est prévu que le cadre d'analyse et la modélisation peuvent être utilisés pour aider à la formulation de stratégies de gestion optimales pour d'autres écosystèmes sujets à des dynamiques complexes.

Nederlandse samenvatting

Ecosystemen leveren de samenleving een groot aantal goederen en diensten, zoals bijvoorbeeld hout en vis, of de mogelijkheid voor recreatie. Hiernaast zijn ecosystemen belangrijk voor het handhaven van een gunstig leefmilieu, middels de regulatie van biochemische en klimaatprocessen, en bezitten zij een intrinsieke waarde gebaseerd op onder andere hun biodiversiteit. Niettemin staan veel ecosystemen onder druk van een groeiende wereldbevolking, in samenhang met een continu toenemende economische activiteit. Veel ecosystemen zijn in meer of mindere mate aangetast, bijvoorbeeld door vervuiling of het niet-duurzaam oogsten van natuurlijke hulpbronnen. Vanwege het belang van ecosystemen voor menselijk welzijn is het van groot belang dat de nu resterende ecosystemen op een optimale manier worden beheerd.

Twee belangrijke elementen in het optimaal beheer van ecosystemen zijn economische efficiëntie en duurzaamheid. Economische efficiëntie is nodig vanwege het belang van ecosystemen voor het levensonderhoud van grote groepen mensen in de wereld, terwijl duurzaamheid betrekking heeft op het garanderen van het voortbestaan van ecosystemen. Economisch efficiënt beheer kan gedefinieerd worden als beheer dat leidt tot een maximale netto contante waarde van de huidige en toekomstige diensten geleverd door het systeem. Duurzaam beheer wordt, in dit proefschrift, geïnterpreteerd als beheer dat toekomstige generaties de beschikking geeft over een hoeveelheid ecosysteem goederen en diensten ten minste gelijk aan de huidige beschikbaarheid. Zowel de efficiëntie als de duurzaamheidaspecten van ecosysteem beheer zijn intensief bestudeerd de afgelopen decennia, met name binnen de vakgebieden milieu-economie, ecologie en milieukunde.

Een belangrijk element binnen de studie van ecosystemen zijn ecosysteem-modellen. De traditionele modellen gingen ervan uit dat veranderingen in ecosystemen zich voltrekken op een geleidelijke, omkeerbare manier. Echter, het wordt steeds duidelijker dat dit vaak niet het geval is. Veel ecosystemen zijn onderhevig aan complexe dynamiek, zoals onomkeerbare veranderingen, niet-lineaire, abrupte veranderingen in het systeem (thresholds), en stochastisch bepaalde veranderingen. Deze complexe dynamiek is inmiddels waargenomen in, onder andere, mariene vispopulaties, zoetwatermeren, verschillende types bossen, savannes, koraalriffen en estuaria. Complexe dynamiek heeft belangrijke consequenties voor de efficiëntie en duurzaamheid van het beheer van ecosystemen: het bepaalt de reactie van het ecosysteem op zowel stress als op rehabilitatie maatregelen. In het licht van de recente inzichten in ecosysteem dynamiek is het nodig om de gevolgen van complexe dynamiek voor het beheer van ecosystemen verder te bestuderen.

De centrale doelstelling van dit proefschrift is daarom 'het onderzoeken van de implicaties van complexe ecosysteem dynamiek voor het efficiënt en duurzaam beheer van ecosystemen'. In het proefschrift is een ecologisch-economische modelleer benadering ontwikkeld die gebruikt kan worden voor de analyse van de efficiëntie en duurzaamheid van beheersmaatregelen voor ecosystemen met complexe dynamiek. Deze benadering is gebaseerd op een raamwerk dat de verschillende interacties tussen de economie en ecosystemen bevat. Toepassing van de benadering vereist het modelleren van de causale keten 'beheersmaatregel' – 'ecosysteem conditie' – 'levering van goederen en diensten'. De efficiëntie van maatregelen kan worden afgeleid van hun impact op de levering van ecosysteem diensten, de duurzaamheid van de maatregelen van hun impact op de lange-termijn condities van het ecosysteem. In het proefschrift is een dynamische simulatie modeleer methode gebruikt, gebaseerd op het gebruik van gekoppelde (differentiaal-)vergelijkingen. Teneinde een zo realistisch mogelijke inschatting te kunnen maken van de dynamiek van het bestudeerde ecosysteem worden in de modellen verschillende ecosysteem componenten, en hun interacties, onderscheiden. Specifiek heb ik gekeken naar drie soorten complexe dynamiek: onomkeerbaarheid van veranderingen in ecosystemen, thresholds, en stochasticiteit. Deze verschillende soorten dynamiek zijn bestudeerd in drie case studies: (i) een hypothetisch bos ecosysteem; (ii) het De Wieden wetland in Overijssel; en (iii) de Ferlo, een savanne ecosysteem in Noord Senegal.

(i) Het bos ecosysteem. In hoofdstuk 3 bestudeer ik de implicaties van onomkeerbaarheid voor het efficiënt en duurzaam beheer van een hypothetisch bos ecosysteem. Het bos levert twee diensten: hout en erosie controle, en het model bevat zowel de vegetatie laag als de bodem, en de interacties tussen deze twee componenten. De beheersvariabele is de rotatie periode die gebruikt wordt voor de oogst van hout. Hoofdstuk 3 presenteert zowel een algemeen model, gebaseerd op de binnen de milieu-economie veel gebruikte Faustmann modellen, als een toepassing van het model. Het hoofdstuk laat zien dat de keuze voor economisch efficiënt of duurzaam beheer ver uit elkaar kan liggen, en dat de mogelijkheden om te komen tot een compromis oplossing sterk worden beïnvloed door het al dan niet omkeerbaar zijn van veranderingen in het ecosysteem.

(ii) De Wieden. Zowel hoofdstuk 4 als 5 zijn gewijd aan De Wieden (Overijssel). In hoofdstuk 4 analyseer ik de diensten geleverd door De Wieden, en de ruimtelijke schaal waarop deze diensten geleverd worden aan verschillende belangengroepen (stakeholders). De onderzochte diensten zijn visserij, rietteelt, recreatie en natuurbescherming. Met behulp van verschillende economische waarderingsmethoden kom ik tot een analyse van de economische waarde van de vier diensten. De totale waarde van deze vier diensten bedraagt rond de 830 euro per hectare per jaar. Dit is hoog in vergelijking met de waarde van omringend landbouwgrond, welke geschat kan worden op tussen de 300 en 400 euro per hectare per jaar. Tevens onderzoek ik hoe belangrijk de economische waarde van de vier diensten is op vier niveaus: gemeente, provincie, nationaal en internationaal. Het blijkt dat visserij en rietteelt alleen van belang zijn op de gemeentelijke schaal, recreatie vooral op de gemeentelijke en provinciale schaal, terwijl natuurbescherming het belangrijkst is op het nationale en internationale schaalniveau. Hierdoor is het te verwachten dat stakeholders op deze vier niveaus verschillende prioriteiten hebben in het beheer van De Wieden. Het is belangrijk dat dit verschil in belangen tussen verschillende schaalniveaus wordt meegenomen bij het opzetten van een beslissingsstructuur voor natuurlijke gebieden (zoals bijvoorbeeld de recente 'Nota Ruime' van VROM).

In hoofdstuk 5 analyseer ik de kosten en baten van het terugdringen van eutrofiering in De Wieden. Net als veel meren in Nederland zijn De Wieden in de afgelopen decennia blootgesteld aan een hoge influx van nutriënten. Bij het bepalen van de impact van rehabilitatie maatregelen is het van belang rekening te houden met de dynamiek van het ecosysteem. Ondiepe meren kunnen zich bij een bepaalde nutriënt concentratie in twee toestanden bevinden: een helder water fase en een troebel water fase. De overgang tussen deze fasen geschiedt middels een threshold. In hoofdstuk 5 ontwikkel ik een ecologisch-economisch model dat de respons van het meer op het terugdringen van de nutriënten toevoer berekent, zowel zonder als met de aanvullende toepassing van algemeen biologisch beheer (dit is het wegvangen van een aanzienlijk deel van de witvis, met name brasem). Met dit model kunnen de kosten en baten van verschillende maatregelen vergeleken worden, teneinde de meest efficiënte beheers-strategie te kunnen selecteren. De kosten die in het model zijn opgenomen zijn de investering- en onderhoudskosten van maatregelen gericht op het verbeteren van de waterkwaliteit in De Wieden. De baten zijn het gevolg van een toename in de levering van ecosysteem-diensten (met name recreatie en natuurbescherming) in het geval de meren overgaan naar een helder water fase. Het model laat zien dat de meest kosten-effectieve manier om helder water te verkrijgen in De Wieden een reductie van de jaarlijkse instroom van totaal-fosfor is met 2 ton/jaar, in combinatie met algemeen biologisch beheer. Tevens blijkt dat, als de waarde van de togenomen mogelijkheden voor recreatie en natuur in het gebied op tenminste 210,000 euro per jaar gewaardeerd worden, het economisch efficiënt is om maatregelen in gang te zetten die leiden tot terugdringen van nutriënten.

(iii) Het Ferlo savanne systeem. De Ferlo (Senegal) wordt zowel in hoofdstuk 6 als in hoofdstuk 7 bestudeerd. In hoofdstuk 6 onderzoek ik hoe begrazing en regenval variatie de productiviteit van de kruid- en graslaag van de Ferlo beïnvloeden. Data zijn gebaseerd op een tienjarig begrazingsexperiment uitgevoerd in het Widou-Thiengoly bassin in de Ferlo. Dit experiment bestudeerde de impact van een hoge (0.15 – 0.20 tropische graaseenheden per ha) en een medium (0.10 tropische graaseenheid per ha) graasdruk. De hoge graasdruk correspondeert met de huidige begrazing in de Ferlo. Het hoofdstuk laat zien dat de graasdruk een sterke invloed heeft op de soortensamenstelling, biomassa productie, en de regen-gebruiks efficiëntie van de vegetatie. Deze laatste factor is een belangrijke indicator voor het functioneren van het ecosysteem en geeft aan hoe efficiënt de vegetatie de gevallen regen omzet in plantaardige biomassa. Uit de analyse blijkt dat in jaren met gemiddelde of hoge neerslag de productiviteit van de savanne nauwelijks beïnvloed wordt door een hoge graasdruk. Echter, tijdens jaren met weinig neerslag is de impact van een hoge graasdruk zeer sterk, met name op de regen-gebruiksefficientie en de productiviteit van de savanne. Dit resultaat is van belang voor het beheer van savannes: een hoge graasdruk kan leiden tot een verhoogde kwetsbaarheid van de savanne, alsmede de lokale bevolking, voor droogte.

In hoofdstuk 7 wordt een ecologisch-economisch model van de Ferlo ontwikkeld, dat gebruikt kan worden voor het berekenen van de optimale graasdruk (veedichtheid) in de Ferlo. In het model hangt de grasproductie, en de groei van de lokale veestapel, af van zowel de stochastisch bepaalde neerslag als de veedichtheid – conform de analyses uitgevoerd in hoofdstuk 6. Het hoofdstuk bevat een breed toepasbaar theoretisch model, alsmede een case studie waarin de optimale graasdruk in de Ferlo is bepaald. In de case studie is ook opgenomen dat de prijzen van vee sterk afhankelijk zijn van de neerslag: tijdens droogte willen veel veehouders verkopen en daalt de veeprijs. Het model laat zien dat het terugkoppelingseffect van begrazing op het ecosysteem zorgt voor een lagere optimale graasdruk in vergelijking met een modelberekening waarin dit feedback effect wordt genegeerd. De case studie laat zien dat de economisch optimale veedichtheid in de Ferlo ongeveer de helft is van de huidige veedichtheid. De veehouders kunnen hun inkomen verdubbelen door het reduceren van de veestapel met zo'n 50%. Een reductie van de graasdruk tot 0.09 tropische graaseenheid per hectare zal leiden tot een toename van het inkomen van de lokale veehouders van de huidige 237 CFA/hectare per jaar tot ongeveer 460 CFA/hectare/jaar. Dit betekent dat regeringsbeleid erop gericht moet zijn om, waar mogelijk, de veestapel in te krimpen, terwijl iedere verdere investering in het uitbreiden van de veestapel economisch contraproductief is.

Samenvattend laat het proefschrift zien dat complexe ecosysteem dynamiek een grote invloed heeft op de kosten en baten, en de duurzaamheid van beheersmaatregelen. Simplificatie van deze dynamiek kan leiden tot de selectie van zowel inefficiënte als niet duurzame beheersmaatregelen. De resultaten van de case studies laten tevens zien dat de in dit proefschrift ontwikkelde ecologisch-economische modelleer-benadering kan leiden tot beter inzicht in, en concrete aanbevelingen voor het formuleren van optimale beheers-strategien. In De Wieden hangen de baten van maatregelen gericht op het terugdringen van eutrofiëring er voornamelijk vanaf of zij een omslag in het ecosysteem naar een helder water fase veroorzaken. Met de in dit proefschrift ontwikkelde modelleerbenadering is het mogelijk de kosten en baten van eutrofiëringmaatregelen te vergelijken. Voor de Ferlo, welke beïnvloed wordt door zowel fluctuerende regenval als de impact van begrazing, is het mogelijk om met de modelleerbenadering de optimale veedichtheid te bepalen. Gebaseerd op deze resultaten concludeer ik dat de ontwikkelde methodiek ook toepasbaar is voor het bepalen van optimale management strategieën voor andere ecosystemen met complexe dynamiek.

Curriculum vitae

Lars Hein was born on 16 September 1969 in Heerlen in the south-eastern part of the Netherlands. He obtained his 'Atheneum' degree at Middelharnis high school in 1987. He first studied Physics at the University of Leiden, where he obtained a 'with distinction' first year degree in 1988. Subsequently, he obtained an MSc in Environmental Sciences at Utrecht University (1994). In addition to completing the natural sciences part of this study, he studied almost two years of economics at Utrecht University and at the Erasmus University, Rotterdam.

Following a traineeship at the Scientific Bureau of the Dutch Ministry of Economic Affairs (CPB) he took up a research position at Utrecht University in 1994. In the 'Science, Technology and Society' group, he worked for almost 3 years on various projects related to the economic aspects of energy conservation, and the analysis and valuation of ecosystem services. In 1997, he joined the FAO-World Bank Collaborative Program in Rome, first as an associate expert, later as a staff member. His responsibilities comprised the formulation of projects and project components in the fields of rural development and natural resource management, and the preparation of environmental impact assessments and environmental economic analyses. This involved extensive travelling, in particular in Asia and Africa.

Since January 2002, he is working for the Environmental Systems Analysis Group at Wageningen University. Besides his PhD research, he has been involved in the implementation of two EU-funded research projects, as well as teaching in the various MSc courses offered by the group. In addition, he has conducted a number of short consultancy assignments for the World Bank and FAO.