Mapping and modelling the effects of land use and land management change on ecosystem services

from local ecosystems and landscapes to global biomes

Katalin Petz

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Table of contents

List of Abbreviations

ACRU	Agricultural Catchments Research Unit
ARIES	Artificial Intelligence for Ecosystem Services
BIP	Biodiversity Indicators Partnership
CBD	Convention on Biological Diversity
CBS	Statistics Netherlands (Centraal Bureau voor de Statistiek in Dutch)
CICES	Common International Classification of Ecosystem Services
CSIR	Council for Scientific and Industrial Research in South Africa
DBS	Dutch Butterfly Conservation
EEA	European Environment Agency
EHS	Dutch Ecological Main Structure (Ecologische Hoofdstructuur in Dutch)
ESP	ecosystem property (Chapter 3)
ES	ecosystem service (Chapter 4 and 5)
ESF	ecosystem function (Chapter 3)
ESS	ecosystem service (Chapter 3)
FAO	Food and Agriculture Organization of the United Nations
GLOBIO3	global biodiversity modelling framework
GIS	geographic information system
GLAM-2	Mapping the Attractiveness of the Dutch Landscape: A GIS-Based Landscape Appreciation Model
GLC	Global Land Cover
GUMBO	Global Unified Model of the BiOsphere
ICSU	International Council for Science
IEEP	Institute for European Environmental Policy
IGBP	International Geosphere-Biosphere Programme
IIASA	International Institute for Applied Systems Analysis
ILCA	International Livestock Centre for Africa
IMAGE	Integrated Model to Assess the Global Environment
InVEST	Integrated Tool to Value Ecosystem Services
IPBES	International science-policy Platform on Biodiversity and Ecosystem Services
ISRIC	International Soil Reference and Information Centre
ISSCAS	Institute of Soil Science – Chinese Academy of Sciences
IUCN	International Union for Conservation of Nature

JRC	Joint Research Centre of the European Comission
LEI	Agricultural Economics Research Institute, part of Wageningen University and Research Centre
LGN	Dutch land use database (Landelijk Grondgebruiksbestand Nederland in Dutch)
LPJ	Lund-Potsdam-Jena managed Land Dynamic Global Vegetation and Water Balance Model
LSU	livestock unit
MA	Millennium Ecosystem Assessment
MIMES	Multi-scale Integrated Models of Ecosystem Services
MODIS	Moderate Resolution Imaging Spectroradiometer
MSA	mean species abundance
NASA	National Aeronautics and Space Administration
NPP	net primary productivity
OECD	Organisation for Economic Co-operation and Development
PBL	Netherlands Environmental Assessment Agency
PRESENCE	Participatory Restoration of Ecosystem SErvices & Natural Capital, Eastern Cape
RUSLE	Revised Universal Soil Loss Equation
SAACA	South African Atlas of Climatology and Agrohydrology
SANBI	South African National Biodiversity Institute
SCOPE	Scientific Committee on Problems with the Environment
SELS	'Speerpunt' Ecosystem and Landscape Services research program
TEEB	the Economics of Ecosystems and Biodiversity
TLU	tropical livestock unit
UNDP	United Nations Development Programme
UNEP	United Nations Environmental Program
UNEP-WCMC	The United Nations Environment Programme's World Conservation Monitoring Centre
UNESO	United Nations Education, Scientific and Culture Organization
UNU	United Nations University
USLE	Universal Soil Loss Equation
VROM	The Ministry of Housing, Spatial Planning and the Environment (Ministerie van Volkshuisvesting, Ruimtelijke Ordening en Milieu in Dutch)
WRI	World Resource Institute
WWF	World Wildlife Fund

Chapter 1

General introduction

1.1 Background

Ecosystems provide numerous benefits to people. These benefits are called ecosystem services and they include, among others, food, fresh water, fertile soils, timber, medicines and recreation opportunities. In order to meet increasing human needs, natural ecosystems have been converted into heavily managed ecosystems, such as cropland and pasture, and their ecosystem services are used exhaustively (De Fries et al., 2004; Foley et al., 2005; Rodríguez et al., 2006). Land conversion and land use intensification are major drivers of ecosystem degradation, biodiversity loss and ecosystem service depletion (Foley et al., 2005; Pereira et al., 2012). More sustainable land use and land management practices could prevent further ecosystem degradation and ensure the continued provision of ecosystem services. To guide sustainable land management strategies, in-depth information about the current and potential impacts of land management on ecosystem services is needed urgently. Substantial efforts to improve the quantification of ecosystems services and to understand ecosystems' contribution to human well-being have been made (Crossman et al., 2013a). Nevertheless, there are still many knowledge gaps about how ecosystems generate services, how to consistently identify and quantify ecosystem services, how these services interact, and how changes in land management affect these services (Carpenter et al., 2009; De Fries et al., 2004; De Groot et al., 2010b; Villamagna et al., 2013). The empirical information about the capacity of ecosystems to provide a number of ecosystem services simultaneously is fragmented, and a solid scientific basis for integrating ecosystem services into land use decisions is still missing (Ehrlich et al., 2012; Nelson and Daily, 2010; Turner and Daily, 2008). This calls for better understanding and quantification of ecosystem services under alternative land management states or systems (Balmford et al., 2008; De Groot et al., 2010a; ICSU et al., 2008) and for further development of mapping and modelling tools that synthesize information to support decision-making with regard to land management (Nelson and Daily, 2010; Vigerstol and Aukema, 2011).

Before the objective of this thesis and the research questions are presented (Section 1.4), the main concepts are introduced and described (Section 1.2) and the relevant literature on ecosystem service mapping and modelling is reviewed (Section 1.3). Finally, the thesis' outline is presented and motivated (Section 1.5).

1.2 Main concepts used in this research

This section describes the main concepts used in this research, namely: ecosystem, landscape, ecosystem service, biodiversity, land management, land cover and land use. These definitions, among others, are also found in the glossary.

An ecosystem is a dynamic complex of plant, animal and microorganism communities and their non-living environment interacting as a functional unit[1]. Ecosystems can be described across the spatial scale, from small patches of, for example, grasslands to global grassland biomes. The ecosystem concept covers natural systems (e.g. forests) as well as ecosystems strongly modified by humans (e.g. agricultural or urban ecosystems) (MA, 2005a). A landscape generally compromises multiple ecosystems and includes the spatial heterogeneity and interactions among these ecosystems. A landscape is therefore defined as a heterogeneous land area composed of a cluster of interacting ecosystems (woods, meadows, marshes, villages etc.) at kilometres wide "human scale" of perception and modification (Forman and Godron, 1986).

The concept of ecosystem services dates as far back as the 1970s, when the economic benefits of natural processes and ecosystems for society were recognized to support nature conservation (De Groot, 1987; Ehrlich and Mooney, 1983; Westman, 1977). The term ecosystem service (or ecological, environmental, nature's or landscape service) has been used implicitly in many studies, but a clear concept of ecosystem services in scientific literature was published only in the 1990s (Daily et al., 1997; De Groot, 1992). A first attempt at economic valuation of ecosystem, services was provided by Costanza et al. (1997). The definition of ecosystem services has changed over time, depending on the emphasis given to ecological basis or economic use (Braat and de Groot, 2012). Some examples are:

- "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life" (Daily et al., 1997);
- "the benefits human populations derive, directly or indirectly, from ecosystem functions" (Costanza et al., 1997);
- "the benefits that people obtain from ecosystems" (MA, 2003);
- "the components of nature, directly enjoyed, consumed, or used to yield human well-being" (Boyd and Banzhaf, 2007);
- "the aspects of ecosystems utilised (actively or passively) to produce human well-being" (Fisher et al., 2009);
- "the direct and indirect contributions of ecosystems to human well-being" (TEEB, 2010); and
- "the direct contributions that ecosystems make to human well-being" (Haines-Young and Potschin, 2011).

The concept has increasingly been used since the publication of the Millennium Ecosystem Assessment (MA, 2003, 2005c). The MA was the first international science-

[1] Convention on Biological Diversity (CBD), 1993. (http://www.cbd.int/convention/text/default.shtml), Accessed last July 20th 2013

policy assessment to provide a comprehensive overview of the consequences of ecosystem change for human well-being. This assessment is nowadays used as a basis for achieving sustainable resource use and nature conservation (Daily and Matson, 2008; Jack et al., 2008; Tallis et al., 2008). A number of regional and sub-global assessments have been published since the original MA. The Southern Africa (Biggs et al., 2004) and Portugal (Pereira et al., 2004) assessments are the most comprehensive ones. The MA distinguished between provisioning services, such as the provision of food and fresh water; regulating services, such as the regulation of climate and air quality; cultural services, such as aesthetic and recreational benefits; and supporting services, such as soil formation and nutrient cycling. Many authors have emphasized the difficulties of including supporting services in decision-making frameworks and valuation schemes (especially regarding double counting) since the MA's appearance (Balmford et al., 2008; Boyd and Banzhaf, 2007; Fisher et al., 2008). How best to define and refine the concept in order to quantify ecosystem services in a consistent manner and use them as a basis for decision-making is still much debated (Fisher et al., 2009; MA, 2005b).

The global study on The Economics of Ecosystems and Biodiversity (TEEB, 2008, 2010) proposed a definition that explicitly acknowledges that services benefit people in multiple, direct and indirect ways. The TEEB study (TEEB, 2008, 2010) provided more in-depth insight in the economic significance of ecosystems. As a result, ecosystem services gained importance at the policy level, which is illustrated by the establishment of the International science-policy Platform on Biodiversity and Ecosystem Services (IPBES), and the incorporation of ecosystem services in the 2020 Aichi targets by the Convention on Biological Diversity (CBD) (Larigauderie and Mooney, 2010; Mace et al., 2010). The TEEB study re-classified ecosystem services into provisioning services (food, fresh water, raw materials, genetic resources, medicinal resources, ornamental resources); regulating services (air quality regulation, climate regulation, moderation of extreme events, regulation of water flow, waste treatment, erosion prevention, maintenance of soil fertility, pollination, biological control); habitat services (maintenance of life cycle of species, maintenance of genetic diversity); and cultural services (aesthetic information, recreation and ecotourism, inspiration for culture, art and design, spiritual experience, information for cognitive development). TEEB does not explicitly recognize supporting services as they are considered part of the underlying structures, processes and functions that characterize ecosystems. This thesis follows the TEEB definition and classification.

In parallel to these international developments, national assessments have also been conducted. The UK National Ecosystem Assessment (2011) is an example of a very comprehensive assessment focussing on mapping and valuation of a wide range of ecosystem services. The assessment of ecosystem services is also at the core of the

European Union 2020 Biodiversity Strategy (Maes et al., 2012). A new standardized classification system, the Common International Classification of Ecosystem Services (CICES) is currently being developed by the European Environmental Agency. The CICES classification aims to better understand how ecosystem services relate to particular economic activities or products and facilitate ecosystem accounts (Haines-Young and Potschin, 2011, 2013). According to CICES, ecosystem services refer to the final outputs or products of ecological systems that are directly consumed or used by people. The CICES classification merges regulating services and habitat services into the 'regulating and maintenance' class and ignores supporting services. The CICES classes can be linked directly to the TEEB classes (Haines-Young and Potschin, 2011, 2013).

An important issue that is still under debate is the position of biodiversity in ecosystems services classifications. Biodiversity is often a motivation for conserving ecosystems and ecosystem services. In some instances, biodiversity is included as a supporting service (following the MA-terminology) (Balmford et al., 2002), in other instances 'providing habitats for biodiversity' is considered as an ecosystem service in its own right (TEEB, 2010). Biological diversity or biodiversity is defined by the CBD as "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems"[2]. There are many ways to measure biodiversity and the various resulting metrics are relevant for different purposes (Butchart et al., 2010). Some of the common metrics are species richness, species abundance, number of threatened species and functional diversity (Díaz and Cabido, 2001), and indices, such as the Mean Species Abundance (Alkemade et al., 2009) and the Living Planet Index (Loh et al., 2005). Biodiversity is important for the delivery of ecosystem services (Naidoo et al., 2008). The relationship between biodiversity and ecosystem services is, however, complex, multi-layered and largely dependent on the characteristics and management of the ecosystem and on the ecosystem services considered (Balvanera et al., 2006; Mace et al., 2012). Biodiversity supports ecosystem processes (e.g. pollination and pest control), affects ecosystem services directly (e.g. crop varieties cultivated for food or medicine) or is valued in its own right (e.g. protected or endangered species) (Balvanera et al., 2006; Mace et al., 2012). Some ecosystem services unquestionably benefit from aspects of biodiversity. For example, high landscape and wildlife diversity stimulates ecotourism (Lindsey et al., 2007). Nevertheless, the complex interactions between biodiversity, ecosystem processes, ecosystem functioning and ecosystem services are poorly understood and are difficult to quantify (Díaz et al., 2006; Mace et al., 2012). In this

[2] Convention on Biological Diversity (CBD), 1993 (http://www.cbd.int/convention/text/default.shtml), Accessed last July 20th 2013

thesis, biodiversity is defined as a combination of ‘habitat for biodiversity’ and ‘abundance of species’ using the Mean Species Abundance index. The contribution of biodiversity to ecosystem services is not studied explicitly.

Land management influences biodiversity, ecosystem functioning and the composition of ecosystem services (Balvanera et al., 2006; Mace et al., 2012; Veldkamp and Fresco, 1996). Land management refers to human activities that affect land cover directly or indirectly and aim to provide specific services (Kremen et al., 2007; Olson and Wäckers, 2007; Verburg et al., 2009). It defines land use and the intensity of use driven by human activities, such as ploughing and irrigating (van Oudenhoven et al., 2012; Verburg et al., 2009). Land management is probably the most important factor influencing the provision of ecosystem services at the landscape level (Ceschia et al., 2010; Fürst et al., 2011; Otieno et al., 2009). For example, activities leading to restoration of vegetation alter ecosystem services by decreasing erosion, stabilizing the water supply, increasing carbon sequestration and providing shelter for wildlife.

Land cover is the physical layer of soil and biomass, including natural vegetation, crops and human structures that cover the land surface (Verburg et al., 2009). Land management affects vegetation, which can degrade as a consequence of intensive use or destructive land management (Reyers et al., 2009). Land use is the purpose for which humans exploit the land cover (e.g. grazing or hay production on grasslands). This purpose is achieved by land management practices (Verburg et al., 2009). Management practices or activities that characterize land use and its intensity include irrigation, pesticide use, livestock management and nature conservation measures (Bennett et al., 2009; Verburg et al., 2009). These management activities define the type and intensity of land use. Land use intensity is characterized by the amount of human input and extraction. Land use intensity ranges from light or extensive with minimal human intervention (i.e. low intensity), to intensive and very intensive management (i.e. high intensity) with many human interventions and conversion of the original ecosystem to permanent human infrastructure or to arable land for food production (De Groot et al., 2010b; Foley et al., 2005). How different land management practices or their consequences, such as agricultural intensity (Temme and Verburg, 2011), vegetation or ecological degradation (Reyers et al., 2009) or restoration measures (Chazdon, 2008), affect ecosystem services is currently better understood (Crossman et al., 2013a). Furthermore, advances have been made in understanding how the management of a certain ecosystem type affects ecosystem services (e.g. Ford et al. (2012), Yang et al. (2012) for grasslands; Chazdon (2008), Başkent et al. (2011) and Ojea et al. (2012) for forests; and Zhang et al. (2007), Swinton et al. (2007) and Sandhu et al. (2010) for agricultural land).

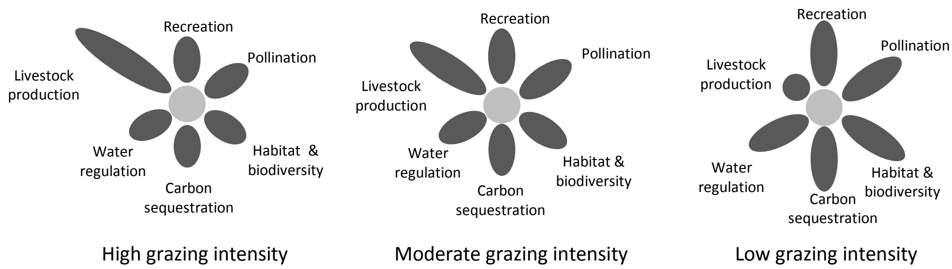

Figure 1.1: Conceptual representation of ecosystem services bundles under varying land use intensities.

Changes in land management practices and land use intensity alter the composition of ecosystem services. It maximizes one or a limited set of services at the cost of others (Foley et al., 2005; Rodríguez et al., 2006). This leads to trade-off between different ecosystem services (Figure 1.1). Minimizing these trade-offs and maximizing the supply of ecosystem services requires sets or bundles of ecosystem services (Raudsepp-Hearne et al., 2010) to be studied and quantified together (Crossman et al., 2013a).

1.3 Mapping and modelling ecosystem services: state of the art

This section defines what maps and models are. It also provides a comprehensive overview of ecosystem service mapping and modelling in terms of current trends in publications, use of spatial scale, ecosystem services studied, and the most common data sources and methods. Finally, a synthesis of mapping and modelling methods is given, including the reasons to choose each method. This leads to the choice of methods used in this PhD research.

1.3.1 What are maps and models?

Maps and models are useful tools to understand, quantify and visualize the spatial distribution of ecosystem services and to communicate this information to decision makers (Crossman et al., 2013b; Kareiva et al., 2011; Martínez-Harms and Balvanera, 2012). Mapping is the process of collection and visualization of geospatial data. A map represents certain features characteristic of an area visually. In this thesis, maps are used to visualize ecosystem properties and the distribution of ecosystem services. The spatial visualization of land management is more difficult, as it involves different activities with temporal as well as spatial component.

A model is an abstract and simplified representation of reality used to understand a certain aspect of that reality. Modelling is the simulation and visualization of biophysical or socio-economic systemic processes by combining certain system elements and parameterizing their behaviour and interactions. How and which elements are combined depends on the purpose of the simulation and visualisation. In this thesis, simple models are developed and applied to estimate the availability of ecosystem services by establishing relationships between land use and ecosystem properties, such as soil type, and the amount of ecosystem services delivered. The models are also used to assess the consequences of different scenarios and to project and compare the effects of potential changes in land management and corresponding land use intensities. Scenarios describe plausible and often simplified future pathways and they are widely used to investigate the effects of socio-economic and environmental changes, and the effects of different policies (MA, 2003).

1.3.2 Trends in ecosystem service mapping and modelling publications

The number of publication on ecosystem service mapping and modelling has increased exponentially over the last two decades, as identified through keyword search in the Scopus database for the period 1992-2012 (http://www.scopus.com) (Figures 1.2 and 1.3). Modelling studies show a stronger increase compared to mapping studies and the number of modelling studies published in 2012 was more than double of mapping studies. Far fewer studies include land management and only about a dozen studies combine land management and ecosystem service modelling or mapping (Figures 1.2 and 1.3). The diverse character of land management activities and related terms may have led to the underrepresentation of studies focusing on land management.

1.3.3 Use of spatial scale in mapping and modelling

Ecological and institutional (i.e. social) phenomena operate at different scales, in space and time (MA, 2003). ‘Scale’ is defined as “both the limit of resolution where a phenomena is discernible and the extent that the phenomena is characterised over space and time” (White and Running, 1994). Ecosystems and ecological processes operate on the spatial scale from plots, ecosystems, landscapes and world regions to the globe; and on the temporal scale from seconds, minutes, hours and days to hundreds and thousands of years. Institutions and the production and use of ecosystem services are present across the spatial scale (Balmford et al., 2008; Fisher et al., 2009; Hein et al., 2006). Land management generally occurs locally, but is constrained by socio-economic factors, such as markets, institutions and governmental policies at national and international levels (Hein et al., 2006). Therefore, the analysis of land management and its effects must be done at different levels of the spatial scale.

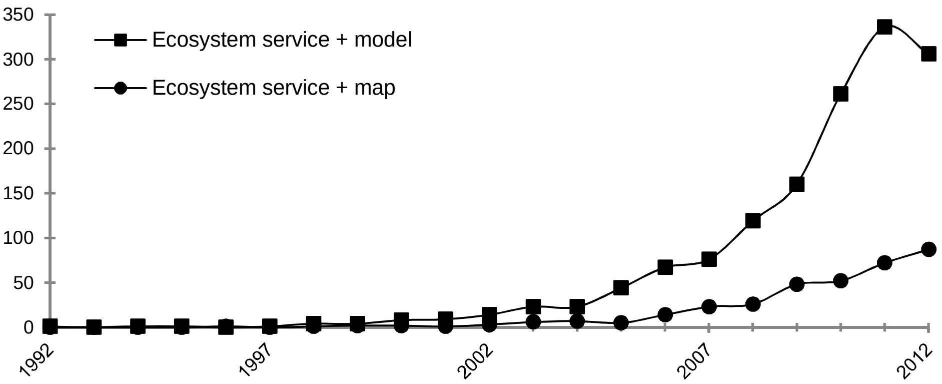

Figure 1.2: Number of publications of ecosystem service mapping and modelling over time (Scopus search 1992-2012, key words in the 'Title, Abstract, Keywords' field: "ecosystem service*" AND "model*"; "ecosystem service*" AND "map*")

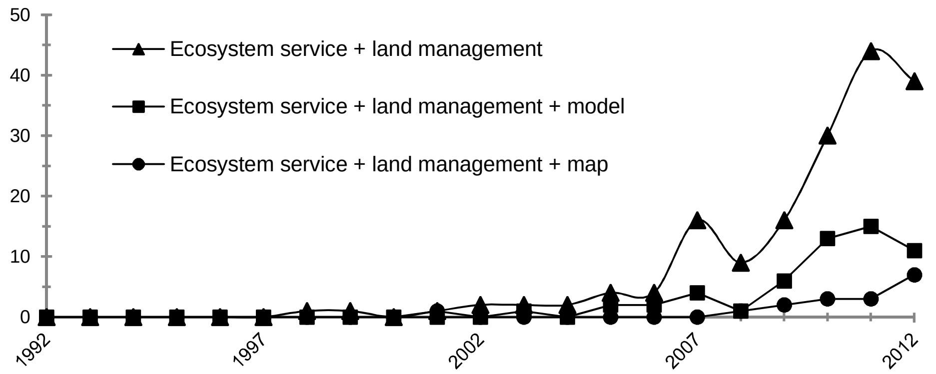

Figure 1.3: Number of publications of land management, ecosystem mapping and modelling over time (Scopus search 1992-2012, key words in the 'Title, Abstract, Keywords' field: "ecosystem service*" AND "land management"; "ecosystem service*" AND "land management" AND "model*"; "ecosystem service*" AND "land management" AND "map*")

Mapping and modelling tools are applied at different spatial and temporal scales, depending on the nature of the problem studied and the scale of the analysis. Several recent studies mapped the supply of multiple ecosystem services at global (Naidoo et al., 2008), continental (Schulp et al., 2012), national (Bateman et al., 2011; Egoh et al., 2008) or sub-national (Nelson et al., 2009; Raudsepp-Hearne et al., 2010; Willemen et al., 2010) levels. This thesis distinguishes ecosystem service studies and models at the landscape and global levels. Landscape level models feed directly into local decision support and spatial planning, whereas global models provide information about global trends and patterns, and can support international policymaking or contribute to international science-policy assessments.

Most studies, however, focus on the local level, on a single landscape or catchment (Egoh et al., 2012; IEEP et al., 2009; Martínez-Harms and Balvanera, 2012). A 'catchment' is an area that forms a comprehensive water drainage system, and includes multiple land uses or landscapes (Allan, 2004). Services dependent on landscape structures and composition, such as pollination, pest control and recreation, operate primarily at landscape scale. The effects of land cover and land use on the spatial distribution of ecosystem services have been widely studied at the landscape-catchment-region level in many parts of the world (e.g. in Europe (Burkhard et al., 2012; Petz et al., 2012; Willemen et al., 2008), in Africa (Egoh et al., 2008; Leh et al., 2013; Swetnam et al., 2011) and China (Bai et al., 2011; Wu et al., 2013)). Landscape-level mapping and modelling approaches generally simulate only a few services and focus on a spatial or temporal scale that is relevant for specific policy questions (Nelson and Daily, 2010).

Only very few studies have mapped or modelled land cover, land use and land management globally (e.g. Ellis and Ramankutty (2008) and van Asselen and Verburg (2012)). Climate regulation, carbon sequestration, water regulation and food provision are the main ecosystem services to have been studied at global level (Naidoo et al., 2008). Pollination, disease regulation and pest control are rarely considered in global ecosystem service studies because they operate locally (IEEP et al., 2009). Another obstacle to incorporating particular services into global models and science-policy assessments is the lack of knowledge about processes. This is the case for disease control and air quality regulation (IEEP et al., 2009). Some global models are able to assess the impacts of economic and environmental factors on natural resources, including ecosystem services (e.g. IMAGE-GLOBIO3 (PBL, 2006), GUMBO (Boumans et al., 2002) and G4M (Kindermann et al., 2006)). The MA (2005c) used already-published, complex individual models to measure potential global change impacts on multiple ecosystem services (Nelson and Daily, 2010). Alcamo et al. (2005) and Naidoo et al. (2008) linked sector-based global models to understand better the interaction between hydrological and other environmental processes and ecosystem services. The Integrated Model to Assess the Global Environment (IMAGE, PBL, 2006) is one of the few global models describing the impacts of socio-economic developments on the environment. IMAGE is used to support international policy formulation in combination with a global biodiversity modelling framework (i.e. GLOBIO3, Alkemade et al., 2009). IMAGE-GLOBIO3 outputs were used in global environmental and biodiversity outlooks (IEEP et al., 2009; Secretariat of the Convention on Biological Diversity, 2010) and for the initial mapping of ecosystem services at the global scale (Schulp et al., 2012).

1.3.4 Ecosystem services included in mapping and modelling studies

The findings of Martínez-Harms and Balvanera (2012), IEEP et al. (2009), Egoh et al. (2012) and Crossman et al. (2013b) show the abundance of studies on specific ecosystem services. Table 1.1 summarizes these findings. On average, only four to five different ecosystem services are mapped and modelled in each individual study. The range of services studied is based either onthe local importance of the service or on the availability of data or expertise (Eppink et al., 2012). Regulating services, including water and climate regulation and carbon sequestration, are the most frequently studied, followed by provisioning, cultural and habitat-supporting services (Crossman et al., 2013b; Egoh et al., 2012; Martínez-Harms and Balvanera, 2012). Medicinal resources, disease control, air quality regulation and traditional knowledge are rarely studied. The services related to freshwater (e.g. flood control and water supply) and to carbon sequestration have received the greatest attention both in scientific and in practical applications (Vigerstol and Aukema, 2011). They are also among the few services mapped globally (Naidoo et al., 2008).

1.3.5 Data sources and mapping and modelling methods

Data sources can be either primary (i.e. measured or sampled field data) or secondary (i.e. literature-based or modelled data). Primary data provide the most accurate estimates of ecosystem services (Eigenbrod et al., 2010). Primary data, especially spatially explicit data, are often not available and this limits ecosystem services research. Hence, data obtained from literature or estimated with remote sensing techniques are commonly used (Eigenbrod et al., 2010; Martínez-Harms and Balvanera, 2012; Seppelt et al., 2012). Primary data are often not readily available for coarser spatial scales, therefore they are used mainly in local and landscape-level studies. Available national or international data mostly cover only provisioning and regulating services (Egoh et al., 2012).

Table 1.1: Frequency of ecosystem services mapped and modelled, based on the findings of Martínez-Harms and Balvanera (2012), IEEP, Alterra et al. (2009), Egoh et al. (2012) and Crossman et al. (2013b)

Most studied	**Often studied**	**Less studied**	**Rarely studied**
Climate regulation	Forage/livestock production	Timber production	Medicinal resources
Carbon storage/sequestration	Soil fertility	Pollination	Disease control
Food provision	Flood regulation	Biofuel provision	Air quality regulation
Recreation	Aesthetic value/Scenic beauty	Erosion control/Soil stability	Natural hazard regulation
Water regulation, supply and quality		Pest control	Waste treatment
		Habitat	Traditional knowledge/ Spiritual and educational value
		Nutrient cycling	

Current assessments and models that study ecosystem service bundles use various approaches (Eppink et al., 2012; Seppelt et al., 2012). This thesis classifies these approaches into four methodological groups based on the reviews of Martínez-Harms and Balvanera (2012), Eigenbrod et al. (2010), IEEP et al. (2009) and Balmford et al. (2008), and describes them using the latest scientific literature. The methodological groups differ in data requirements and level of complexity, use a variety of mathematical techniques, such as regression analysis, dynamic models and geographic information system (GIS), and can be applied to different spatial scales. The four methodological groups are:

- Proxy-based methods or lookup-tables
- Statistical models
- Causal relationships
- Biophysical models

Proxy-based methods, which use literature- or expert-based estimates of ecosystem services linked to particular land cover or land use types, are the most commonly used method to map ecosystems services (Egoh et al., 2012; IEEP et al., 2009; Martínez-Harms and Balvanera, 2012). Examples include Egoh et al. (2008), Burkhard et al. (2009) and Nelson et al. (2010). Land cover-based proxies enable the user to map ecosystem services quickly in regions where primary data are lacking. At the same time, proxies generalize information, reduce spatial accuracy and limit the understanding of ecological processes (Eigenbrod et al., 2010; Rounsevell et al., 2012). Carbon sequestration is often derived simply from land cover or land use, both at the landscape (e.g. Raudsepp-Hearne et al. (2010) and Bai et al. (2011)) and global levels (Naidoo et al., 2008). Other commonly used proxies for ecosystem services are soil, vegetation and nutrient-related indicators (Egoh et al., 2012). Regulating services are often estimated by using databases (e.g. Food and Agriculture Organization of the United Nations, FAO) and topographic and remote-sensed information (Martínez-Harms and Balvanera, 2012). Global livestock, food and timber production estimates are mostly taken from the FAO statistics (IEEP et al., 2009).

Statistical models provide the most direct information about ecosystem services if primary data are available. They use statistical correlation or regression analysis to extrapolate the availability of ecosystem services across space based on sampled field data of various different biophysical and environmental variables (Martínez-Harms and Balvanera, 2012). Willemen et al. (2008), for example, used regression analysis to map tourism and plant habitat. Statistical models can be used to link biophysical processes with social variables, such as perception and expectations, on which cultural services depend (Daniel et al., 2012; Sherrouse et al., 2011). Statistics provide the basis for tracking and quantifying uncertainty in ecosystem service assessments (Smith et al., 2011). Statistical models calculate correlations and not necessarily causality. Using statistical relationships for conditions outside the original data domain may therefore yield unreliable results. Statistical models are rarely used globally, as primary data are scarce at this level.

The SolVES[3] tool, a GIS tool to assess, map, and quantify the perceived social values for ecosystems uses statistical models (Sherrouse et al., 2011). ARIES (Artificial Intelligence for Ecosystem Services[4], Villa et al., 2009), Bayesian Belief Networks (Haines-Young, 2011) and Maxent (Phillips et al., 2006) are other examples. ARIES uses a probabilistic Bayesian network to define relationships between input and ecosystem service values based on data from other similar sites (using a probabilistic benefit transfer approach).

Causal relationships are the other most frequently used method to map ecosystem service (Martínez-Harms and Balvanera, 2012). Here, land cover variables are related to other biophysical variables based on the current understanding of causal relationships to create a proxy for ecosystem services (Eigenbrod et al., 2010; Martínez-Harms and Balvanera, 2012). Examples include recreation (Chan et al., 2006) and erosion prevention (Egoh et al., 2008) at the landscape level, and air quality regulation and tourism globally (Schulp, 2012). Causal relationships can rely both on primary and secondary data. Causal relationships improve ecosystem service estimates when primary data are absent and are easily applicable to other regions or environmental conditions. Therefore, causal relationships are a major improvement over land cover based proxies (Eigenbrod et al., 2010; Martínez-Harms and Balvanera, 2012). However, establishing the causal relationship requires an adequate knowledge of how an ecosystem service is generated. The general knowledge of how biophysical and social variables determine ecosystem services provision remains poor. Uncertainties increase and erroneous conclusions may be drawn if the causal variables are poor predictors of ecosystem services (Eigenbrod et al., 2010; Martínez-Harms and Balvanera, 2012).

Biophysical models are mathematical models that describe certain processes of the biophysical environment or an ecosystem service, using quantitative biophysical functions of the interactions between environmental and human factors that drive environmental and ecosystem service change. Biophysical models often imply high complexity and they are based on either primary or secondary data. If there is an excellent understanding of the system dynamics, causal relationships can be aggregated and generalized into quantitative biophysical models. However, it remains challenging to determine the appropriate modelling complexity and realistic representations of biophysical processes and feedbacks (Rounsevell et al., 2012; Seppelt et al., 2012). Seppelt et al. (2012), for example, showed that mapping with look-up tables is preferred over complex models. Biophysical models

[3] http://solves.cr.usgs.gov/, Accessed last November 20th, 2013
[4] http://www.ariesonline.org/, Accessed last November 20th, 2013

provide a good estimate of an ecosystem service if proper input data are available and if models are appropriately calibrated (Nelson and Daily, 2010). Carbon sequestration (e.g. Naidoo et al. (2008)) and water supply (e.g. Naidoo et al. (2008) and Alcamo et al. (2005)) are often derived from biophysical models that use climate and land cover information (IEEP et al., 2009). Biophysical models may actually be more data-intensive than statistical models (Nelson and Daily, 2010; Vigerstol and Aukema, 2011). Biophysical models used to predict ecosystem services are dynamic (e.g. GUMBO/MIMES[5], Boumans and Costanza, 2007; Boumans et al., 2002), Guo et al. (2000) and Portela and Rademacher (2001)) and often also spatially explicit (e.g. IMAGE, PBL, 2006).

Currently, one of the most commonly used and comprehensive ecosystem service modelling and mapping tools is InVEST (Integrated Tool to Value Ecosystem Services[6], Kareiva et al., 2011). InVEST, an open access GIS-tool collection, includes separate models for multiple ecosystem services to analyse spatial patterns or track changes caused by land cover change using land cover data and other relevant environmental variables (Crossman et al., 2013b). The complexity of these models varies from proxy-based mapping to simple biophysical production equations. InVEST has been used to map and value ecosystem services under different land cover scenarios, among others, in Oregon, the United States (Nelson et al., 2009) and Tanzania (Swetnam et al., 2011). Bai et al (2008) used InVEST to analyse the spatial correlations between biodiversity and ecosystem services in China and Guerry et al (2012) used InVEST to quantify ecosystem services in a Canadian marine case study (Crossman et al., 2013b).

1.3.6 Synthesis and choice of mapping and modelling methods

Many studies, reviews and books focus on ecosystem services quantification, mapping and modelling. These studies vary widely in services studied, the scale of analysis and in the approach used to map and model ecosystem services (Crossman et al., 2013b; Seppelt et al., 2011; Villamagna et al., 2013). Consequently, there is no consensus on what is actually mapped, and on the methods used to map and model ecosystem services. Therefore, it may be difficult to compare studies, even if they describe similar ecosystem services. There is no standardized, broadly accepted way to map or model ecosystem services (Crossman et al., 2013b; Martínez-Harms and Balvanera, 2012). This is also true of the study of land management effects on these ecosystem services. Current mapping and modelling studies on multiple ecosystem services mainly refer only to land cover and land use (Bennett et al., 2009). This knowledge gap is an important shortcoming, since the

[5] http://www.afordablefutures.com/services/mimes, Accessed last November 20th, 2013
[6] http://www.naturalcapitalproject.org/InVEST.html, Accessed last November 20th, 2013

provision of ecosystem service within a land use type varies among the different land management strategies.

All mapping and modelling methods and model types have their role, strengths and shortcomings. The selection of the most adequate method and modelling approaches depends on the purpose of the study and on data, expertise and time constrains (Figure 1.4). Simple models have reduced data requirements, are easier to run, require less expertise, but often provide less accurate results than complex models (Vigerstol and Aukema, 2011). The main challenge of ecosystem service mapping and modelling is to create approaches that are sufficiently complex to represent the system, but also simple enough to be understood and be parameterized with often limited data (Tallis and Polasky, 2011). Crossman et al. (2013a), for example, call for the better linking of biophysical models to high resolution data and the supply of ecosystem services. Biophysical models, causal relationships, proxy-based methods and probabilistic relationship transfers rely at least partly on secondary data. This makes the methods applicable when primary data are scarce. All four methods, except for statistical models, imply causality, making them applicable to understand and extrapolate the effects of land management on ecosystem services. Therefore, this PhD study will use mainly biophysical models, causal relationships and proxy-based methods, consistent with the InVEST approach to map and model ecosystem services (Figure 1.4).

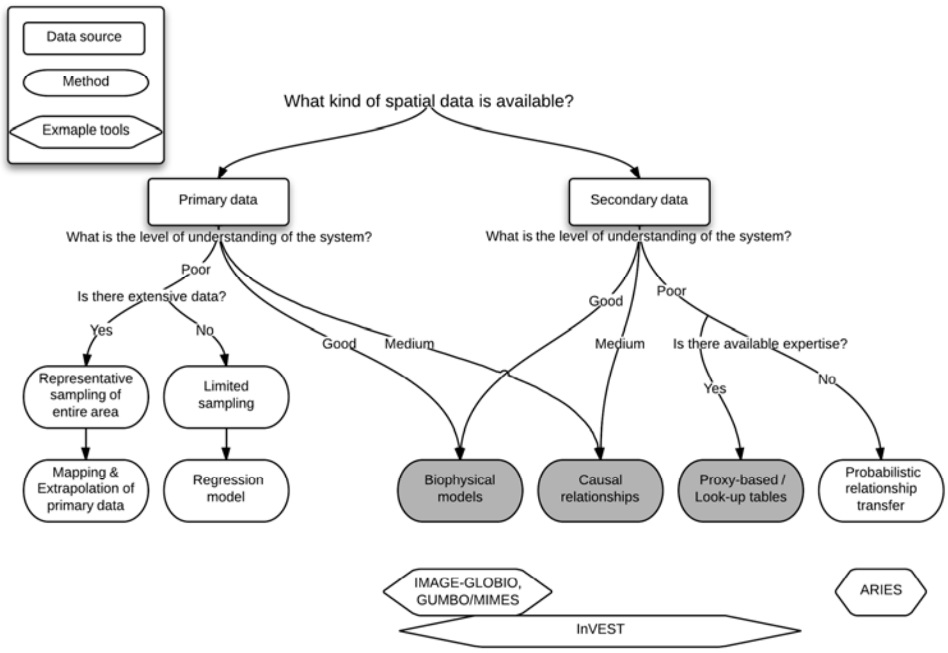

Figure 1.4: Decision tree of mapping and modelling method choice depending on data availability, based on Martínez-Harms and Balvanera (2012) Eigenbrod et al. (2010) and Vigerstol and Aukema (2011). The dark grey boxes indicate the methods applied in this thesis.

1.4 Objectives

In-depth information about the impact of land management change on a wide range of ecosystem services is important to guide land use and land management decisions. However, quantitative and empirical information about the effects of land management on ecosystem services is generally limited. The objective of this thesis research is therefore to develop a methodology to quantify the effect of land management on the spatial distribution of ecosystem services, in order to determine ecosystem service trade-offs caused by land management. The research focusses on scales ranging from local ecosystems and landscapes to global biomes. To achieve this objective six research questions (RQ) are formulated and answered. The first two research questions are methodological and the other four concern the results. The RQs are:

1) How can land management and its effects on bundles of ecosystem services be characterized?
2) How can the effect of land management change on ecosystem services be quantified and modelled across the spatial scale when data are limited?
3) What is the effect of land management on the spatial distribution of bundles of ecosystems services?
4) Which land management option provides most ecosystem services and meets most policy targets?
5) What are the land management-related synergies and trade-offs between ecosystem services?
6) What is the effect of changes in land management on bundles of ecosystem services from landscapes to worldwide ecosystems?

To answer these questions, existing but scattered information about the dependencies between land management and ecosystem service provision are integrated. GIS-based mapping and modelling tools are developed for different scales, from the landscape to the global level. The mapping and modelling tools are applied in combination with spatial analysis and scenario analysis. The development of mapping and modelling methods is a core part of the research, as well as the demonstration of how these methods can be used to assess and evaluate land management effects on ecosystem services when data are scarce.

Selected ecosystem services from the provisioning, regulating, habitat and cultural service categories are mapped and modelled. Ecosystem services that are linked directly to land management are emphasized, in order to assess effectively and efficiently the consequences of different management options. Because of their complex character, ecosystem services are assessed through indicators. Indicators are selected that express most accurately the changes in ecosystem services and on which data are available. A comprehensive but generic framework is developed to support indicator selection, quantification, mapping and modelling (RQ1). The framework is applicable to cases at different scales. Three case studies, ranging from landscape to global levels, are selected to answer the research questions: a small-scale Dutch landscape (the Groene Woud), the Baviaanskloof Catchment in South Africa and natural rangelands across the world. Mapping and modelling is central to all three case studies. Nevertheless, the focus of each case study differs. The Dutch case study characterizes and quantifies ecosystem services (RQs 1 2), and studies their spatial distribution (RQ3). The South African case study places land management in the policy context (RQ4), and the natural rangeland study identifies synergies and trade-offs between ecosystem services (RQ5). The last research question is answered using a synthesis of the results of the three case studies (RQ6).

1.5 Outline of the thesis

This thesis consists of six chapters, including this introduction. Each of the subsequent chapters addresses at least one of the research questions (Figure 1.5).

Chapter 2 presents a framework that links land management to the provision of ecosystem services in a stepwise approach. The chapter demonstrates how the framework is used for systematic indicator selection, quantification and mapping with the example of a Dutch case study, the Groene Woud (RQ 1).

Chapter 3 describes the effect of land management on eight ecosystem services in the Groene Woud case study. Ecosystem services are quantified, mapped and modelled for current land management and a scenario analysis demonstrates the expected effect of different levels of land use intensity on ecosystem services (RQs 2, 3 and 6).

The framework development and its application for indicator selection and ecosystem function and service quantification in the Dutch case study, the Groene Woud, was executed in collaboration with Alexander van Oudenhoven, a fellow PhD candidate (Chapters 2 and 3).

Chapter 4 evaluates alternative land management options through quantifying and mapping multiple ecosystem services in the South African Baviaanskloof Catchment. Seven ecosystem services are studied for three alternative management scenarios developed by local stakeholders. The land management options are evaluated in terms of ecosystem service provision and meeting management targets (RQs 2, 4 and 6).

Chapter 5 quantifies trade-offs and synergies between livestock grazing intensity and ecosystem services on natural rangelands worldwide by using global-scale datasets and models (RQs 2, 5 and 6). This chapter locates areas where grazing and livestock production are unsustainable and where ecosystem services are impaired by livestock grazing.

Finally, Chapter 6 discusses the strength and weaknesses of the mapping and modelling approach and presents a synthesis of the main findings and the conclusions.

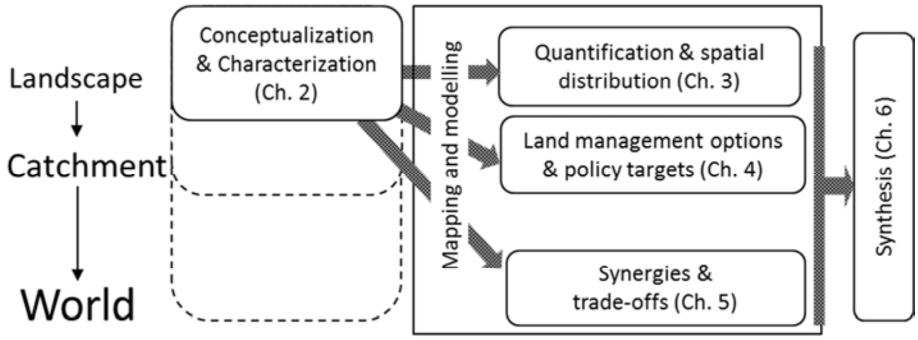

Figure 1.5: Overview of the methodological steps in the different chapters of the thesis; from conceptualization to mapping and modelling across the spatial scale.

Milk cow in the Netherlands

Chapter 2

Framework for indicator selection to assess effects of land management on ecosystem services

Land management is an important factor that affects ecosystem services provision. However, interactions between land management, ecological processes and ecosystem service provision are still not fully understood. Indicators can help to better understand these interactions and provide information for policy-makers to prioritize land management interventions. In this paper, we develop a framework for the systematic selection of indicators, to assess the link between land management and ecosystem services provision in a spatially explicit manner. Our framework distinguishes between ecosystem properties, ecosystem functions, and ecosystem services. We tested the framework in a case study in The Netherlands. For the case study, we identified 12 property indicators, 9 function indicators and 9 service indicators. The indicators were used to examine the effect of land management on food provision, air quality regulation and recreation opportunities. Land management was found to not only affect ecosystem properties, but also ecosystem functions and services directly. Several criteria were used to evaluate the usefulness of the selected indicators, including scalability, sensitivity to land management change, spatial explicitness, and portability. The results show that the proposed framework can be used to determine quantitative links between indicators, so that land management effects on ecosystem services provision can be modelled in a spatially explicit manner.

Keywords: indicators, land management, milk production, air quality regulation, recreation

Based on: A. P.E. van Oudenhoven, K. Petz, R. Alkemade, R. S. de Groot, L. Hein (2012) Framework for systematic indicator selection to assess effects of land management on ecosystem services, Ecological Indicators, Vol. 21, pp.110-122

2.1 Introduction

Ecosystems provide humans with numerous benefits, such as clean water, medicines, food, and opportunities for recreation. The Millennium Ecosystem Assessment (2005c) highlighted the importance of these ecosystem services for sustaining human well-being. The Economics of Ecosystems and Biodiversity study (TEEB, 2010) provided insight in the economic significance of ecosystems. As a result, the ecosystem services concept has now gained importance at the policy level, illustrated by the establishment of the International science-policy Platform on Biodiversity and Ecosystem Services (IPBES), and the incorporation of ecosystem services in the 2020 targets set by the 10th Conference of Parties to the Convention on Biological Diversity (Larigauderie and Mooney, 2010; Mace et al., 2010).

Policy and environmental planning decisions largely influence how land is being managed (Carpenter et al., 2009; Fisher et al., 2008; von Haaren and Albert, 2011). On a regional scale, land management is one of the most important factors that influence the provision of ecosystem services (Ceschia et al., 2010; Fürst et al., 2010b; Otieno et al., 2011). Land management is defined by the presence of human activities that affect land cover directly or indirectly (Kremen et al., 2007; Olson and Wäckers, 2007; Verburg et al., 2009). It comprises ecosystem exploitation, land use management, and includes ecosystem management (Bennett et al., 2009; Brussard et al., 1998). Land management refers to human activities; land cover to the biotic and abiotic components of the landscape, e.g. natural vegetation, forest, cropland, water, and human structures (Verburg et al., 2009). Land use refers to the purpose of human activities to make use of natural resources, thereby impacting ecological processes and functioning (Veldkamp and Fresco, 1996). Land management includes but does not equal ecosystem management, because it refers to managing an area so that ecological services and biological resources are conserved, while sustaining human use (Brussard et al., 1998; MA, 2005c). Examples of land management include irrigation schemes, tillage, pesticide use, nature protection and restoration (Bennett et al., 2009; Blignaut et al., 2010; Carvalho-Ribeiro et al., 2010; Follett, 2001; Ngugi et al., 2011).

The analysis of ecosystem services to support land management decisions faces a number of challenges. They include: (1) identifying comprehensive indicators to measure the capacity of ecosystems to provide services; (2) dealing with the complex dynamics of the link between land management and ecosystem services provision; (3) quantifying and modelling the provision of ecosystem services by linking ecological processes with ecosystem services; and (4) accounting for the multiple spatial and temporal scales of ecological processes and ecosystem services provision (Bastian et al., 2012; Carpenter et al., 2009; De Groot et al., 2010b; Turner and Daily, 2008; van Strien et al., 2009; Villa et al., 2009).

Given these challenges, it is necessary to have a consistent and comprehensive framework for analysing ecosystem services (Ostrom, 2009; Posthumus et al., 2010). A framework provides structure to the research and enables better validation of its outcomes (Bockstaller and Girardin, 2003; Niemi and McDonald, 2004). Furthermore, it is important to formulate a comprehensive set of indicators (Layke et al., 2012; Niemeijer and de Groot, 2008) that enables the assessment of land management effects on ecosystem services provision, at different levels of the spatial scale (Carpenter et al., 2009; De Groot et al., 2010b; van Strien et al., 2009). With indicators, policy-makers and land managers can be provided with information, based upon which interventions can be identified, prioritized and executed (Layke, 2009; OECD, 2001). Finally, there is a need to test how ecosystem services frameworks can be used for the selection of indicators (Nelson et al., 2009).

The objective of our study was, therefore, to systematically select indicators which can be used to analyse the link between land management and the provision of ecosystem services across the spatial scale. To achieve this objective we developed a consistent framework for indicator selection, which builds on existing frameworks, in particular by TEEB (De Groot et al., 2010a) and Haines-Young and Potschin (2010).

We first describe our framework and how it can be used for indicator selection. Then, we apply it to a case study to assess the effect of land management on ecosystem services provision. Characteristics of and interactions between indicators were studied, and all indicators were evaluated based on a selected set of criteria. The case study was done in a multifunctional rural landscape in the southern part of the Netherlands, where multiple ecosystem services are provided across the spatial scale.

2.2 Method

2.2.1 Framework

Consistent and comprehensive frameworks that link human society and economy to biophysical entities, and include impacts of policy decisions, have been developed during the last decades. For the analysis of ecosystem services such a framework was developed in the context the Millennium Ecosystem Assessment (MA, 2003), which was itself based on a Driver, Pressure, State, Impact, Response framework. We adapted the frameworks by TEEB (De Groot et al., 2010a) and Haines-Young and Potschin (2010) for indicator selection. These frameworks are among the most recent and comprehensive ecosystem services assessment frameworks. The TEEB framework explains the link between biodiversity, ecosystem services and human well-being (De Groot et al., 2010a) and builds

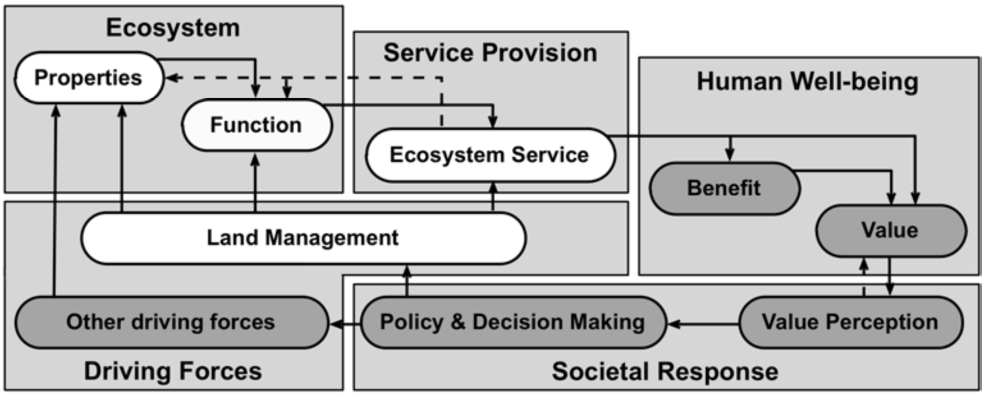

Figure 2.1: Framework for assessing links between land management, ecosystem services provision, and human well-being. Based on Haines-Young and Potschin (2010), Kienast et al. (2009), De Groot et al. (2010a), and Hein (2010). The white boxes indicate the scope of our study. Solid arrows indicate effects; dashed arrows indicate feedbacks.

on several recent studies (Braat et al., 2008; Fisher et al., 2008; Fisher et al., 2009; MA, 2003). The TEEB-study calls for the development of indicators for the economic consequences of biodiversity and land use change (De Groot et al., 2010a; Reyers et al., 2010). The stepwise so-called ‘cascade-model’ by Haines-Young and Potschin (2010) is useful for assessing the provision of ecosystem services in a structured way, linking ecosystem properties to functions and services. Although the importance of land management is acknowledged in (descriptions of) both frameworks, land management is not explicitly included. We therefore adapted the framework by including land management, which enables the selection of indicators for assessing the effects of land management and ecosystem services.

Figure 2.1 shows the main elements of our framework: the driving forces, ecosystem, service provision, human well-being, and societal response. The emphasis of our study is indicated by the white boxes in Figure 2.1: land management, ecosystem properties, function and service. Unless stated otherwise, definitions and relations provided are based on or adapted from the TEEB-study (De Groot et al., 2010a). In the framework, we use the term ‘ecosystem’. We note, however, that the interactions which we describe below can refer to ecosystems at multiple levels of spatial scale, e.g. at landscape, regional or even national (Hein et al., 2006).

Drivers or *driving forces* are natural or human-induced factors which can influence the ecosystem, either directly (e.g. through climate change or environmental pollution) or indirectly (e.g. through changes in demography or economy) (MA, 2005c). Although drivers such as climate change or environmental pollution have also an impact on the ecosystem, we focus in our assessment on the driving force *land management*. As described earlier, land management are the human activities that can affect ecosystem properties and function (Bastian et al., 2012; Chen et al., 2011; Kremen et al., 2007), as well as the ecosystem service that can be provided (Edwards et al., 2011; O'Farrell et al., 2007). *Ecosystem properties* are the set of ecological conditions, processes and structures that determine whether an ecosystem service can be provided. Examples include net primary productivity (NPP), vegetation cover, and soil moisture content (Johnson et al., 2002; Kienast et al., 2009). Ecosystem properties underpin *ecosystem functions,* which are the ecosystem's capacity to provide an ecosystem service (De Groot et al., 2010a). An ecosystem function or potential (Bastian et al., 2012), given by a subset of ecosystem properties, indicates to what extent an ecosystem service can be provided. Examples of ecosystem functions include capturing of aerosols by vegetation (Nowak et al., 2006) and carbon sequestration (Díaz et al., 2009). The *ecosystem service* contributes to human well-being, for example cleaner air and reduced climate change. The *benefit* is the socio-cultural or economical welfare gain provided through the ecosystem service, such as health, employment and income. Finally, actors in society can attach a value to these benefits. Value refers to importance, and it is most commonly defined as the contribution of ecosystem services goals, objectives or conditions that are specified by a user (Costanza, 2000; Farber et al., 2002). The *value perception* can trigger changes in *policy and decision-making,* for instance when certain services or resources are not available or too expensive. Alternatively, value perception can influence the ecosystem service value, for instance through increasing demand for a certain product. Policy and decision-making form preconditions, constraints and incentives for land management and other drivers (Daily et al., 2009; Fisher et al., 2009).

2.2.2 Indicator selection and evaluation

To operationalize the framework for indicator selection, it is important to select indicators that provide accurate information on all main aspects of ecosystem services provision: land management, ecosystem properties, function, and service (Figure 2.1). To be able to evaluate the usefulness of indicators for our purpose, we compiled a set of criteria. First, we assembled general criteria for indicators, based on information from ecological assessments. We found that the selection process of indicators should be *flexible* and *consistent,* and that indicators should be *comprehensive* and understandable to multiple types of end users. A flexible, yet consistent selection process implies that multiple frameworks can be used, depending on the scope and aim of the assessment (Niemeijer and de Groot, 2008). A test for comprehensiveness evaluates whether the whole set of indicators would provide complete and consistent information, which relates to the specific research question (Niemi and McDonald, 2004). Considering that information should be communicated among scientists and other stakeholders, indicators need to be clear and understandable in order to be useful to these multiple end users (Niemeijer and de Groot, 2008; UNEP-WCMC, 2011).

We also looked for criteria that were more specific for indicators for ecosystem services. We found that indicators need to be *sensitive to (changes in) land management*, *temporally and spatially explicit, scalable*, and *quantifiable*. These criteria apply both to individual indicators as well as sets of indicators and ensure that the indicators can be used for quantification and modelling purposes. Furthermore, indicators should provide information about causal relationships between land management and changes in ecosystem properties and function (De Groot et al., 2010b; Riley, 2000). Temporal and spatial explicitness refers to whether trends can be measured and mapped over time, and whether relations between indicators can be linked to specific locations, for instance through mapping and GIS analyses (National Research Council (NRC), 2000). An indicator is considered scalable if it could be aggregated or disaggregated to different scale levels, without losing the sense of the indicator (Hein et al., 2006). Quantifiable indicators ensure that information can be compared easily and objectively (Layke et al., 2012; Schomaker, 1997).

Finally, we considered *data availability*, *credibility*, and *portability* as other criteria. Data availability is especially essential if information are compared among different studies (Layke et al., 2012). Indicators should also provide credible information. This criterion tests whether indicators actually convey reliable information (Layke et al., 2012). Portability refers to the question whether indicators are repeatable and reproducible in other studies, and across different regions (Riley, 2000).

2.2.3 Case study: Indicator selection and evaluation for 'Het Groene Woud', The Netherlands

We applied the framework for the selection of indicators for nine ecosystem services in a rural area in the south of The Netherlands (Box 1). First, we focused on interactions between indicators for ecosystem properties, function and service. Secondly, we assessed the effect of land management on the provision of three ecosystem services. For both steps of the case study, we evaluated the indicators using the criteria as introduced in the previous section.

Figure 2.2: Map of case study area. 'Het Groene Woud' is located in the southern part of The Netherlands (inset), between three large cities, situated north, west and south of the area. Land cover data by de Wit et al. (1999)

Box 1: Study area description

'Het Groene Woud' (~330 km^2) is located in the southern part of The Netherlands (Figure 2.2), amidst three densely populated cities: Eindhoven (216 000 inhabitants), 's-Hertogenbosch (140 000), and Tilburg (200 000) (CBS, 2011). The area comprises extensively managed maize & grassland, rural settlements and patches of forest and heath lands (Figure 2.2). Due to its tranquillity, abundant forest patches and cultural historic elements, Het Groene Woud offers many recreation opportunities to inhabitants of surrounding cities (Het Groene Woud, 2011). Moreover, agriculture has been an important economic activity in the area. A large part of the area is occupied by cropland (20%, mainly corn and wheat) and grassland (43%, dairy production) (De Wit et al., 1999; Kuiper and de Regt, 2007). Finally, an increasing area is part of the Dutch Ecological Main Structure (EHS) and Natura 2000 network (Blom-Zandstra et al., 2010). Therefore, local biodiversity and the connectivity of the natural elements in those segments need to be protected and enhanced (Het Groene Woud, 2011).

Het Groene Woud was declared a Dutch National Landscape in 2005, which resulted in the implementation of new policies to protect the area's unique cultural-historical and natural features (Het Groene Woud, 2011). The main challenge for local policy-makers and managers lies in maintaining agricultural production while protecting biodiversity and increasing recreation opportunities (Petz and van Oudenhoven, 2012).

Indicator selection for ecosystem properties, function and service

We made an inventory of ecosystem services provided in Het Groene Woud, and of the indicators that describe these services or describe relevant properties. For this, we conducted expert interviews and consulted scientific literature, policy documents, reports from local projects and organisations, brochures, and websites. The typology of the TEEB study (De Groot et al., 2010a) was used to categorise the ecosystem services. The selected ecosystem service types are listed below, with the specific service for the study area between parentheses: food provision (dairy production), air quality regulation (fine dust capture), climate regulation (carbon sequestration), regulation of water flows (water retention), biological control (protection from pest insects), opportunities for recreation & tourism (walking), lifecycle maintenance (refuge for migratory birds), aesthetic information (green residential areas), and information for cognitive development (research and education).

We selected individual indicators of ecosystem properties, function and service for each selected ecosystem service, and determined qualitative relations between them. Examples of these qualitative relations include if and how vegetation characteristics affect water storage and fine dust capture, or relations between carbon stored in vegetation and change in atmospheric CO2 concentration. If insufficient information was available on the provision of ecosystem services in the area, we consulted literature on similar services in other case studies. Examples include air quality studies in other areas in The Netherlands (Wesseling et al., 2008) and in the UK, such as Glasgow (Bealey et al., 2007) and East England (Beckett et al., 2000).

Linking indicators for land management and ecosystem services

To analyse the relation between land management and ecosystem services, we studied three services in detail: dairy production, fine dust capture, and opportunities for recreation. For each service, we focused on the role of land management factors as well as on relations (including feedbacks) between ecosystem properties, function and service. These relations were also determined qualitatively. There were several reasons for analysing three instead of all nine services. We considered it important to study an example each of provisioning, regulating and cultural services, to test whether the framework would enable the selection of a proper set of indicators for different ecosystem service categories. Moreover, the three services were identified as key services in the area (Blom-Zandstra et al., 2010; Het Groene Woud, 2011). In addition, fine dust capture by vegetation is an understudied ecosystem service (Nowak et al., 2006), yet considered highly relevant in the Netherlands (Hein, 2011; Velders et al., 2007; Wesseling et al., 2008).

After selecting indicators with management relevance, we studied how these could be linked to indicators for ecosystem properties, function and service. In addition, we looked at the spatial scale and mapped the function indicators in order to visualize spatially the potential of the area for providing the service. We distinguished between landscape element, plot and landscape levels across the spatial scale. We considered landscape elements, such as individual trees, bushes, treelines or other physical structures, of less than 1 km^2 that could be studied in isolation from the landscape (Grashof-Bokdam et al., 2009a; Krewenka et al., 2011). We assumed a plot to correspond with patches of land cover (e.g. forest or grassland) with a size of 1-10 km^2; and the entire study area (350 km^2) was assumed to be representative of a landscape.

2.3 Results

2.3.1 Indicators for provision of multiple ecosystem services

Relevant indicators for the provision of nine ecosystem services in Het Groene Woud were selected. These ecosystem services were: dairy production, fine dust capture, carbon sequestration, water retention, protection from pest insects, refuge for migratory species, green residential areas, opportunities for walking, and research and education. We identified 12 key indicators for ecosystem properties, nine for functions, and nine for service provision. An overview of these indicators is presented in Figure 2.3.

Indicators for ecosystem properties were grouped into five categories, of which three are described as ‘natural properties’ (soil, water, flora and fauna) and two as indicating ‘human presence’ (land cover and landscape structure, and infrastructure). Examples of these human presence indicators include the degree of naturalness (also a measure of urbanisation), noise level (mainly caused by traffic), and number and extent of dairy farms. Function indicators were divided into four categories, in line with the ecosystem functions typology by De Groot et al. (2002) and as also used by Kienast et al. (2009). Function indicators refer to ecosystem’s capacity to provide a service, e.g. amount of water stored in vegetation, fine dust captured by vegetation, and the walking suitability of an area. Service performance indicators were grouped in accordance with the typology of the TEEB-study (De Groot et al., 2010a). These indicators refer to the actual service provision or use from which people benefit. Examples include milk production, change in ground water level, change in atmospheric fine dust concentration, and the number of walkers in an area.

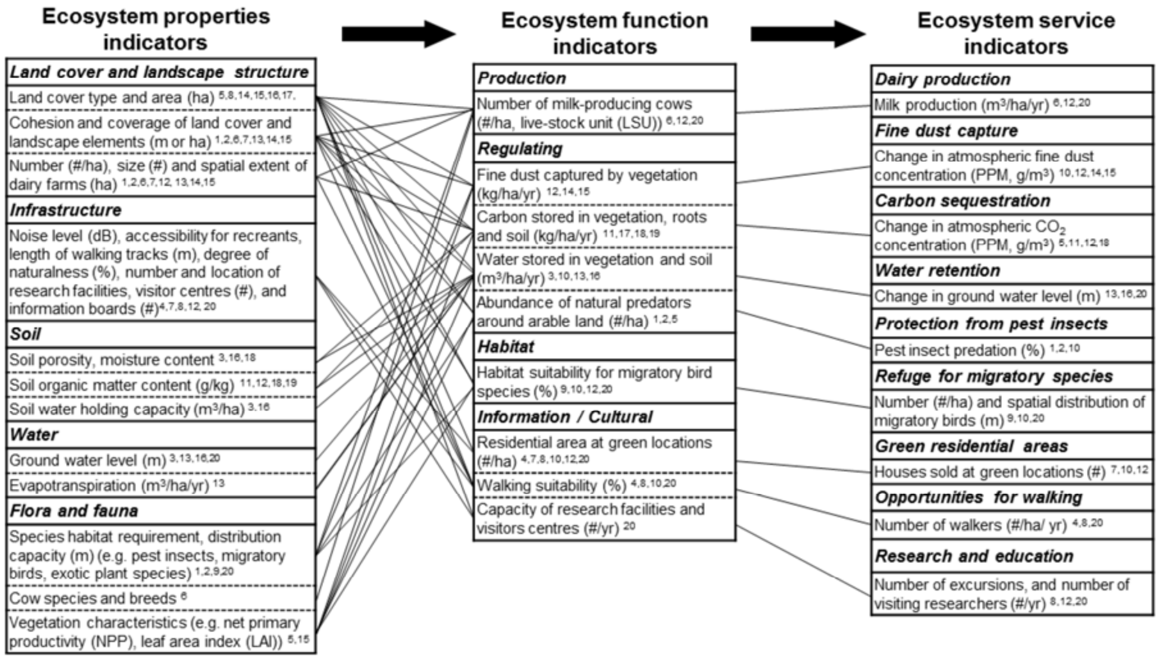

Figure 2.3: Overview of key properties, function and service indicators for nine ecosystem services in Het Groene Woud. Units are given between parentheses. Lines indicate linkages between individual indicators. Typology of indicators is based on De Groot (1992), Kienast et al. (2009) and De Groot et al. (2010a).

Sources: [1] Baveco and Bianchi (2007) [2] Bianchi et al. (2008; 2009); [3] De Vries and Camarasa (2009); [4] De Vries et al. (2007); [5] Foley et al. (2005); [6] Naeff and Smidt (2009); [7] Goossen and Langers (2000); [8] Goossen et al. (1997); [9] Grashof-Bokdam and Langevelde (2005); [10] Kienast et al. (2009); [11] Kuikman et al. (2003); [12] Layke (2009); [13] Mulder, Querner (2008); [14] Oosterbaan et al. (2006); [15] Oosterbaan et al. (2009); [16] Querner et al. (2008); [17] Schulp et al. (2008); [18] Schulp and Verburg (2009); [19] Pulleman et al. (2000); [20] Website 'Groene Woud'. Accessed on January 20th, 2011, URL: www.groenewoud.com.

The number of ecosystem properties indicators was the highest. All functions depend on land cover and landscape structure, whereas vegetation characteristics influence all functions but the information and cultural functions. Indicators for ecosystem functions were found to depend on a large number of ecosystem properties and corresponding indicators. Indicators for regulating and habitat functions could be linked to many ecosystem property indicators: water stored in vegetation to most (eight), followed by carbon stored in vegetation (six), fine dust captured by vegetation (four), and natural predators abundance (four). To each ecosystem function indicator one service indicator was assigned. Therefore, the number of service indicators corresponds with the number of function indicators.

2.3.2 Effect of land management on ecosystem properties, function and service: example for three ecosystem services

Food provision: dairy production

Management for dairy production affects ecosystem properties, function and service provision (Figure 2.4). Application of pesticides and nutrients, the first land management indicator in Figure 2.4, influences several ecosystem properties. For instance, the NPP of grass can be enhanced by applying fertilizers (Batáry et al., 2010; Jangid et al., 2008). Veterinarian measures can influence the cows' milk producing capacity through disease prevention and additional feeding. Mechanisation can affect the area of grassland and farm size that is required for milk production. Moreover, mechanisation can alter the grass properties through mowing; the milk producing capacity of the cows through more efficient feeding; and the milk production through mechanised milking.

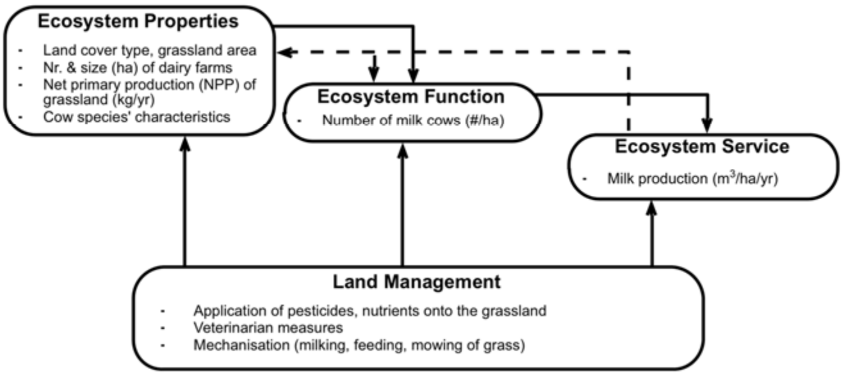

Figure 2.4: Framework with indicators for land management, ecosystem properties, function, and services, for the provisioning service 'milk production'. Arrows indicate direct linkages between the boxes; the dashed line indicates feedback.

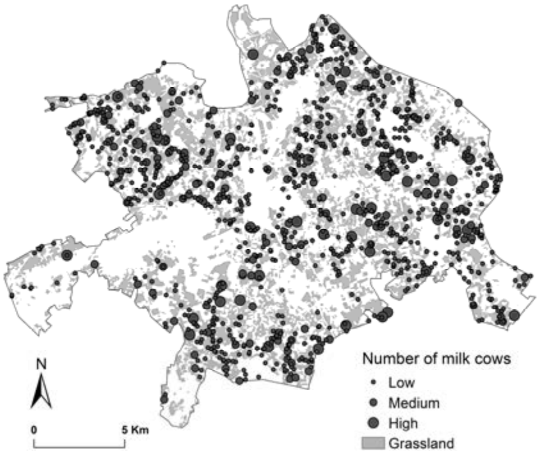

Figure 2.5: Map of Het Groene Woud, indicating where the service 'milk production' can be provided. The service indicator number of milk cows (dots) and function indicator area of grassland (light grey) were mapped. Land cover data by de Wit et al. (1999), milk cow data by Naeff, Schmidt (2009).

The number of milk cows (function indicator) is not only influenced by management, but also by ecosystem properties. The land cover type as well as the size and number of dairy farms influence how many cows can graze on how much land. Milk production is influences by the cows' characteristics and NPP of grass influence, which in turn also determines the required grassland area. The milk production (service indicator) is related directly to the number of cows. However, milk production can also influence the ecosystem function and properties. For instance, if the (targeted) milk production is too high, the number of cows and the area of grassland would have to be altered. This would require either more nutrient application and mechanisation, increasing the number of cows or area of grassland, or lowering the milk production.

The service dairy production is provided on grassland, which covers about 60% of the study area (Figure 2.5). The highest numbers of cows (function indicator) are kept in the northwest, south and east, but generally these numbers are evenly distributed over the area. The actual service performance can be measured on plot (grassland) and landscape (entire area) level, as its spatial pattern follows the allocation of the grassland across the landscape. Only a few parts of the area are not used for dairy production. They include forest patches and urbanized areas.

Air quality regulation: fine dust capture

The key management action that influences the fine dust concentration involves selecting the location and planting (species choice) as well as maintaining forest plots and woody elements (Beckett et al., 2000; McDonald et al., 2007; Oosterbaan et al., 2006).

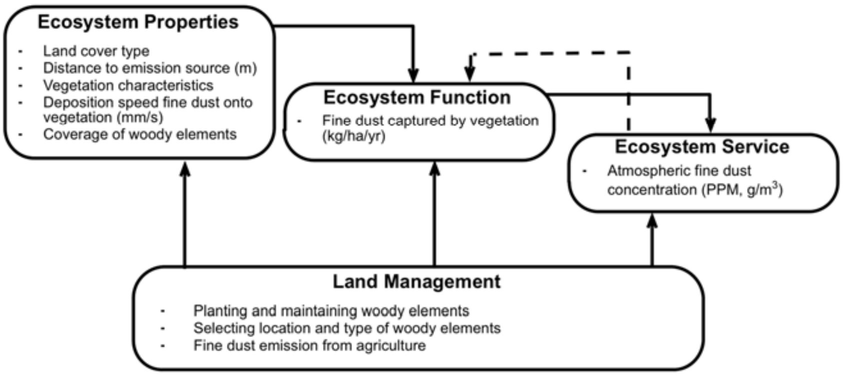

Figure 2.6: Framework with indicators for land management, ecosystem properties, function, and service, for the regulating service 'fine dust capture'. Solid arrows indicate direct linkages between the boxes; the dashed line indicates feedback.

Woody elements are forest patches and tree rows. For example, on a yearly basis coniferous tree species can capture twice as much fine dust as deciduous tree species (Oosterbaan et al., 2009). Vegetation characteristics such as leaf area and hairiness determine the deposition speed onto and therefore the capture of fine dust by vegetation (Beckett et al., 2000; Oosterbaan et al., 2009). Spatial planning is important because the distance between woody elements and fine dust emission sources (such as roads, intensive agriculture, and cities) determines the woody elements' capacity to capture fine dust (function indicator) (Tonneijck and Swaagstra, 2006) (Figure 2.6).

Intensive agriculture together with traffic are the main fine dust emission sources in Het Groene Woud (Oosterbaan et al., 2009). Local emission influences the amount of fine dust that can be captured by vegetation directly (Nowak and Crane, 2000; Nowak et al., 2006), and naturally causes a change in atmospheric fine dust concentration (service indicator). On locations where concentrations are higher, e.g. point sources such as pork stables, vegetation can capture more fine dust than on other locations. The amount of fine dust captured by vegetation (function indicator) results in a change in atmospheric fine dust concentration (service).

There are large differences in capacity of land cover types to capture fine dust, and therefore deciding on the location and extent of land cover can have a large influence on fine dust concentration. Forests and woody elements have a higher capacity to capture fine dust than all other types of land cover. Moreover, adding or maintaining woody elements can further increase the area's total capacity, as is shown in Figure 2.7. Fine dust capture can be measured on landscape element (e.g. treerows), plot (forest patch) and landscape levels (entire area). Figure 2.7 shows the spatial pattern of woody elements and forest plots across the landscape in Het Groene Woud area. All areas except those with urban infrastructure (white on the map) contribute to the capture of fine dust in the area.

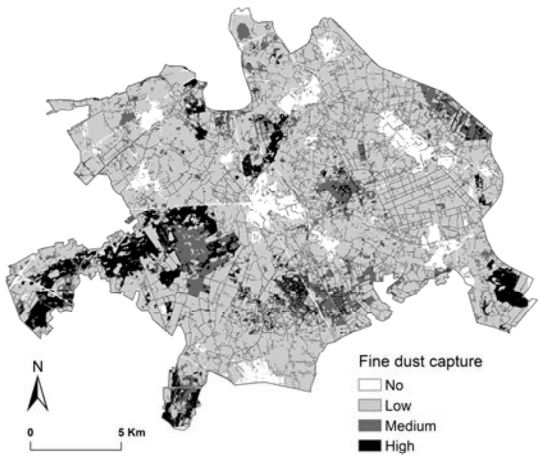

Figure 2.7: Map of Het Groene Woud, indicating where the service 'fine dust capture' can be provided. The function indicator 'fine dust capture' was mapped, based on the capacity of land cover, land use, and woody elements to capture fine dust. Forest areas (black) have a higher capacity to capture fine dust than other types of land cover. Air quality information by Oosterbaan et al. (2009), land cover data by de Wit et al. (1999).

Opportunities for recreation: walking

Managing Het Groene Woud area to improve walking opportunities influences the area's ecosystem properties and functions. Developing and maintaining nature reserves, parks and green areas influence the area's degree of naturalness, can increase the length of walking tracks and accessibility (Goossen and Langers, 2000). Protecting and maintaining historical landscape elements improve the historical distinctiveness of the area (Edwards et al., 2011; Het Groene Woud, 2011). Finally, improving the accessibility of rural landscapes and nature areas determines whether walkers can actually visit the areas (De Vries et al., 2007). Many walkers prefer to visit locations where parking space, route indication, walking routes and information boards are available (De Vries et al., 2007; Goossen and Langers, 2000) (Figure 2.8).

The area's suitability for walking (function indicator) can be improved by designating separate areas for walking. However, the suitability mainly depends on the area's properties, such as land cover preference, accessibility, the length of walking tracks, the naturalness, the noise level and the presence of historic elements in the area (Goossen et al., 1997). Land cover types that are preferred by walkers are forest or heath over arable land, grassland or urban areas (Goossen and Langers, 2000). The diversity of land cover is also highly appreciated by walkers (De Vries et al., 2004; van den Berg et al., 1998).

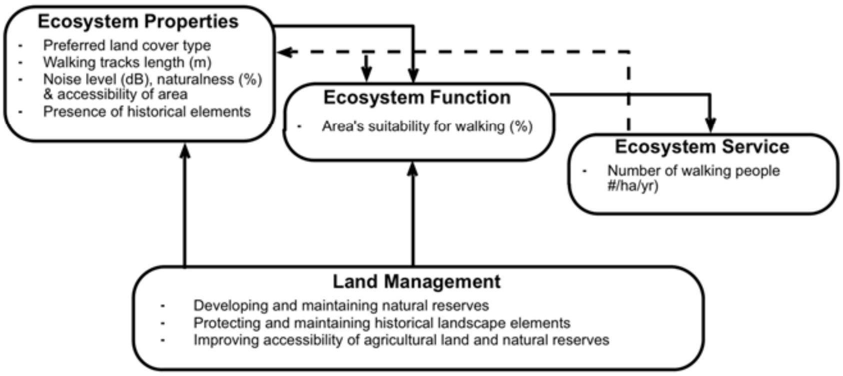

Figure 2.8: Framework with indicators for the land management, ecosystem properties, function, and service boxes, for the cultural service 'opportunities for walking'. Arrows indicate direct linkages between the boxes; dashed lines indicate feedbacks.

The actual service performance can be measured by the number of walkers (service indicator), which is related directly to the walking suitability. Naturally, an area with higher suitability is more likely to attract larger numbers of walkers (De Vries et al., 2004; Goossen and Langers, 2000). At the same time, too many walkers can influence the function and properties, for instance through increased noise level and loss of naturalness (van den Berg et al., 1998). Forest and areas with high land cover diversity are preferred the most for walking (Figure 2.9). This land cover preference (properties indicators) can be measured on plot (e.g. forest patch) and landscape level. The map also indicates the distance from urban areas to potential walking areas. The majority of the area is suitable for walking.

2.4 Discussion

2.4.1 Methods: framework & indicator selection

In this paper, we presented a framework to analyse effects of land management on ecosystem services. The framework elements (driving forces, ecosystem, service provision, human well-being and societal response) basically follow the DPSIR approach (Driving forces, Pressure, State, Impact, Response), which was also used by Braat et al. (2008), Niemeijer and De Groot (2008), Layke et al. (2009), and others. Our framework enables the assessment of how land management can affect ecosystems ('state'), and their services and human well-being ('impact'). These are two subjects of which the ecosystem services assessments face most scientific challenges (Carpenter et al., 2009; ICSU et al., 2008).

To clarify the distinction between 'state' and 'impact', Kienast et al. (2009) adapted the 'cascade model' from Haines-Young and Potschin (2010) and defined the meaning of the terms 'landscape function' and 'ecosystem service'. The stepwise 'cascade-model' was also referred to by Bastian et al. (2012) and De Groot et al. (2010a; 2010b) but to our best

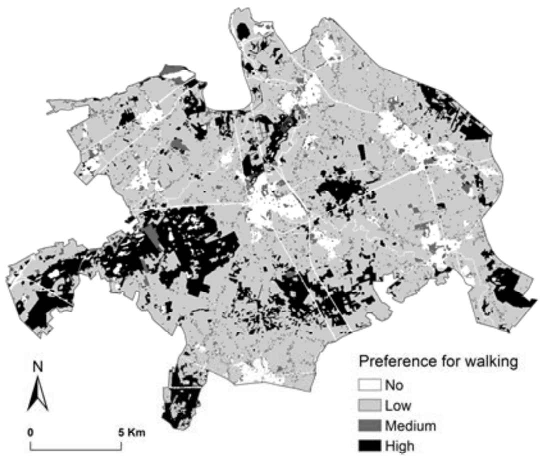

Figure 2.9: Map of Het Groene Woud, indicating where the service 'opportunities for walking' can be provided. The properties indicator preferred land cover type for walking was mapped. Forest areas (dark) are preferred most by walkers, compared to agricultural area (grey) and urban area (white). Recreation preference information by Goossen, Langers (2000), land cover data by de Wit et al. (1999).

knowledge, the framework we present is a first actual application focused on the biophysical aspects and underlying management effects that matter for the provision of ecosystem services. Our framework enables this analysis in a structured and stepwise manner, avoiding the confusion between ecosystem properties, functions and services and thereby also avoiding double-counting (Bateman et al., 2011). This specification is essential to link ecosystem service assessments to valuation studies (Farber et al., 2006). Some remaining challenges are briefly described below.

Flexibility and comprehensiveness

Ecosystem assessment frameworks should be flexible enough to be modified in line with the aim of the assessment (Czúcz et al., 2011; De Bello et al., 2009). Many studies have been carried out on impacts of land use on ecosystem services provision (Barral and Oscar; Fürst et al., 2010a; Richert et al.; Schröter et al., 2005) and on policy and land use planning in relation to ecosystem services (e.g. van Meijl et al. (2006), Fisher and Turner (2008), and Fürst et al. (2011)). Incorporating their findings into the framework would be an important next step to make it more comprehensive. Specifying more detailed relationships between policy and other drivers would also allow for a more complete ecosystem services assessment.

Quantification of indicators

Establishing causal relationships is an import factor, when seeking to improve more accurate quantitative relationships (Lin et al., 2009). Our framework can help to determine quantitative relationships between the various steps of service provisioning, e.g. how does ecosystem functioning depend on ecosystem properties, how do ecosystem functions provide ecosystem services, and how to measure the benefits derived from ecosystem services? Quantified relationships could also provide input for more reliable and accurate mapping and modelling and for determining the value of ecosystem services.

Practical applicability

Indicators are important to understand how ecosystem services are provided, through both qualitative and quantitative links between the different steps. Initiatives like the Biodiversity Indicators Partnership (BIP[7]) and the World Resources Institute (WRI) ecosystem services indicators database (Layke, 2009), as well as studies by Fisher et al. (2009) and others offer examples of frameworks for indicator selection and sets of ecosystem services indicators. However, practical guidelines to select multiple appropriate indicators, that can be used to both quantify and model ecosystem services provision, are still lacking (ICSU et al., 2008; UNEP-WCMC, 2011). A lack of robust procedures and guidelines for selecting indicators could decrease the validity of the information by the indicators (Dale and Beyeler, 2001).

The criteria we used to evaluate indicators for land management and ecosystem services provision can be seen as a first step towards a more streamlined indicator selection procedure for ecosystem services. Many criteria stemmed from ecological studies (Dale and Beyeler, 2001; Lin et al., 2009), but also recent studies focused more strongly on ecosystem services provided us with useful criteria (Layke et al., 2012; UNEP-WCMC, 2011). The twelve criteria could be divided into criteria that help evaluating the indicator selection process, the practical aspects of ecosystem service assessments, the indicators' ability to convey information, and causal links between indicators.

2.4.2 Case study: applying the framework

In the first part of the case study, the complex relationships between ecosystem properties, functions and services were investigated. Each property indicator could be linked to several ecosystem functions, which shows the fundamental role of ecosystem properties in the provision of multiple ecosystem services. The indicators provided a

[7] www.twentyten.net., Accessed last August 6^{th} 2011

comprehensive overview of the biophysical state and structural characteristics of the study area.

Function indicators proved to be a subset or combination of ecosystem properties indicators, as was earlier suggested by Kienast et al. (2009). Function indicators were more specific than properties indicators and corresponded to only one specific service indicator. Although function indicators generally provide information about service potentials, they were rarely similar to service indicators. However, they often had corresponding units. Properties and function indicators, also called state indicators, provide information on how much of a service an ecosystem can potentially provide in a sustainable manner (De Groot et al., 2010b; Layke, 2009). Service indicators, also called performance indicators, provide information on how much of the service is actually provided and/or used (De Groot et al., 2010b; Fisher and Turner, 2008; Layke, 2009). For ecosystem services assessments, be it quantitative, mapping or modelling studies, it would be commendable to select at least one state and one performance indicator per studied ecosystem service (UNEP-WCMC, 2011). It is also important to make the distinction between indicators for ecosystem function and for service.

Applying the framework to three different services (i.e. food provision, air quality regulation and recreation) illustrated that the linkages, including feedbacks, differ per ecosystem service. Indicators for land management related to land cover, nature protection, application of pesticides and mechanisation, among others. Interestingly enough, they also included indicators that go beyond “traditional” ecosystem management (Grumbine, 1994). Results showed that land management can affect ecosystem services directly (food provision and air quality regulation) or indirectly through ecosystem properties and functions (air quality regulation and recreation). This underlines the importance of management (input) and the smaller contribution of nature’s capacity in the case of production of food. Moreover, management aimed at a certain function or service could have feedbacks on the properties that are fundamental for the provision of other services. Applying the framework and mapping of functions enabled us to see at which levels of the spatial scale services were provided and, additionally, land management could affect the provision of these services. The consideration of spatial scale is important not only because service provision can occur across the spatial scale, but also because the level of service provisioning and decision-making might differ (Daily et al., 2009; Hein et al., 2006; Seppelt et al., 2012). The selected indicators could be linked to landscape element, plot, and landscape levels. Results showed that properties indicators and some function indicators could be linked to all three levels of the spatial scale, whereas some function and all service indicators could only be linked to plot and landscape levels.

Table 2.1: Evaluation of indicators that were identified in the case study. Indicators for ecosystem properties, functions and services (vertical) were evaluated using eight criteria. When it could not be reliably established if indicators met certain criteria, it was indicated by ‘unclear’.

Criteria **Indicator type**	**Flexible selection process**	**Consistency**	**Comprehensive**	**Sensitive to changes in land management**	**Temporarily explicit**	**Spatially explicit**	**Scalable**	**Credibility**
Ecosystem properties indicators								
Land cover and landscape structure	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Infrastructure	Yes	Yes	Yes	Yes	No	Yes	Yes	Yes
Soil	Yes	Unclear	Unclear	Unclear	Unclear	Yes	Yes	Unclear
Water	Yes	Unclear	Unclear	Yes	Yes	Unclear	Unclear	Unclear
Flora and fauna	Unclear	Unclear	Unclear	Unclear	Unclear	Unclear	Yes	Yes
Ecosystem function indicators								
Production	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Regulating	Yes	Yes	Yes	Yes	Yes	Unclear	Yes	Yes
Habitat	Yes	Yes	Unclear	Unclear	Yes	Yes	Yes	Yes
Information / Cultural	Yes	Unclear	Yes	Unclear	Yes	Yes	Yes	Unclear
Ecosystem service indicators								
Milk production	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Fine dust capture	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Carbon sequestration	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Water retention	Yes	Unclear	Unclear	Unclear	Unclear	Yes	Yes	Unclear
Protection from pest insects	Yes	Unclear	Yes	Yes	Yes	Yes	Yes	Unclear
Refuge from migratory species	Yes	Yes	Unclear	Unclear	Yes	Yes	Yes	Yes
Green residential areas	Unclear	Unclear	Unclear	Unclear	Yes	Yes	Yes	Unclear
Opportunities for walking	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Research and education	Yes	Unclear	Unclear	Unclear	Yes	Yes	Yes	Unclear

Our criteria can be used as guidelines to select and evaluate indicators. The evaluation of the indicators can be seen in Table 2.1. Although we did not test the indicators for usefulness to multiple end-users, quantification and modelling, and portability, we conclude that the selection procedure was sufficiently flexible and allowed for the selection of a consistent set of comprehensive indicators. Although some indicators (e.g. refuge for migratory species) were difficult to link to land management, the large majority was sensitive to changes in land management. All function indicators were or could be made temporally and spatially explicit, and many could be linked to one or more levels of the spatial scale. The amount of available literature and other information indicates that the indicators are credible, i.e. provide reliable information. In general, indicators for ecosystem properties were found to be most difficult to fully comprehend and utilize, because fewer criteria were met. Especially habitat and cultural functions met only a few criteria. It can be expected that such indicators, which meet only a few criteria, will be difficult to utilize in ecosystem service assessments, and mapping and modelling exercises.

Perhaps an important criterion to further develop would be one that focuses on evaluating whether an indicator would be suitable as a property, function or service indicator. The set of indicators presented here, as well as the maps, could provide local decision-makers with useful information when developing regional management plans. Although the case study yielded indicators that could be relevant for other ecosystem services assessments, we point out that the indicators we found were specific to the area's policy needs, socio-economic situation and spatial configuration.

2.5 Conclusion

This paper describes a framework to select indicators to assess effects of land management on the provision of ecosystem services. The framework was tested in Het Groene Woud area, a multi-functional landscape in the Netherlands. Our framework explicitly connects land management to ecosystem properties, functions and services. For the nine studied ecosystem services, we identified twelve key ecosystem properties, nine function and nine service indicators. Indicators for ecosystem properties that could be linked to each function were land use, land cover and landscape structure. Indicators for regulating and habitat functions could be linked to most ecosystem properties indicators. Furthermore, land management was found to affect ecosystem properties and functions, as was the case for three key ecosystem services in the study area: milk production, fine dust capture, and recreation. In the case of food provision and air quality regulation, ecosystem services were also found to be affected directly by land management.

We conclude that the framework enables the flexible selection of indicators to analyse land management effects on ecosystem services at multiple scales. The criteria we used to evaluate the selected indicators can be seen as a step towards practical guidelines for indicator selection. We recommend that future ecosystem service assessments follow an equally structured methodology, and select at least one state and performance indicator per ecosystem service. The framework we presented in this paper is useful to better understand and quantify the interactions between land management, ecological processes and the provision of ecosystem services. Therefore, the framework can be used to determine quantitative links between indicators, so that land management effects on ecosystem services provision can be modelled in a spatially explicit manner.

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Anthocharis cardamines (Orange Tip) butterfly

Chapter 3

Modelling land management effects on ecosystem functions and services: a case study in the Netherlands (Groene Woud)

Knowledge about the effect of land management on ecosystem services (ESSs) is essential for making decisions on land management. Current modelling approaches that aim to assist decision making generally do not distinguish between ecosystem functions (ESFs) and ESSs, or include land management effects. Our objective was to model the effect of land management on multiple ESSs in 'Het Groene Woud', the Netherlands. Based on quantitative and spatial relationships, we mapped and modelled eight ESFs and ESSs. Next, three ESSs were analysed under two quantitative management scenarios. Natural areas and green landscape elements proved crucial for providing recreation and regulating services. Agricultural areas mainly provide milk and fodder but few other services. We conclude that land use type and green landscape elements are suitable variables for modelling land management effects. Our study underlines that the stepwise analysis of ESSs is essential to understand the interactions between services. The generic relationships we established enable the application of the method for other areas, either inside or outside the Netherlands. The ESF and ESS maps can be used for regional management, because they provide location-specific quantitative information on ecosystems' capacity to provide services as well as on the service provision itself.

Keywords: ecosystem services; land management; mapping; landscape; land use; scenario; the Netherlands; GIS

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Corrigendum (2012). International Journal of Biodiversity Science, Ecosystem Services & Management 8, 286-286.

3.1 Introduction

Human activities have resulted in the conversion of natural forests, grasslands and other ecosystems into cropland and pastures, to provide an increasing world population with food, water, fuel wood, and construction material (Foley et al., 2005; Rodríguez et al., 2006). These changes have impaired the ecosystems' capacity to sustain food production and provide fresh water to humans; provide a healthy habitat and shelter for animal and plant species; regulate climate and air quality; and prevent crops and humans to suffer from infectious diseases (Díaz et al., 2009; Foley et al., 2005; ICSU et al., 2008; WRI et al., 2008). The contributions to human well-being by ecosystems are defined as ecosystem services (ESSs) (De Groot et al., 2010a). Over the years, evidence has mounted on the extent and value of ESSs provided globally (Costanza et al., 2008; TEEB, 2010; WRI et al., 2008), as well as on their decline as a result of land management change and other drivers (ICSU et al., 2008; Kremen et al., 2007; MA, 2005a). We defined land management as the presence of human activities that are affecting land directly or indirectly (van Oudenhoven et al., 2012). Land management can influence land cover, land use and the provision of ESSs (Foley et al., 2005; Verburg et al., 2009).

To develop policies on sustainable land use options or to make adjustments in land management systems, it is essential to have information on the impact of land management change on the bundle of ESSs (ICSU et al., 2008; Nelson et al., 2009). However, quantitative empirical information on the capacity of a given ecosystem to provide a multitude of services is still lacking (ICSU et al., 2008). The biophysical characterization of ESSs is still not well established (Chan et al., 2006; Villa et al., 2009). One of the main challenges for current ESSs research is assessing the bundles of ESSs provided through alternative land management systems (De Groot et al., 2010b; ICSU et al., 2008).

Mapping and modelling of ESS are tools that can help to better understand the interactions between land management and the provision of ESSs (Daily et al., 2009; De Groot et al., 2010a). Among others, Reyers et al. (2009), Egoh et al. (2011), Chan et al. (2006) and Bai et al. (2011) have mapped and modelled ESSs in biophysical quantities. These studies focus mainly on water, carbon sequestration, pollination, biodiversity, and recreation (or tourism) services. They do not distinguish explicitly between the capacity to provide the ESS (ecosystem function, ESF) and its contribution to human well-being (ESS) (De Groot et al., 2010b). Often ESFs rather than ESSs have been quantified and mapped (Kienast et al., 2009; Lamarque et al., 2011), such as Willemen et al. (2008). In several mapping and modelling studies, ESFs and ESSs are reduced to indicators with limited management and policy relevance (Maes et al., 2011; Raudsepp-Hearne et al., 2010; Willemen et al., 2008). Land management may cause changes in land use and landscape structure and can thus alter the processes and structure of an ecosystem, i.e. ecosystem properties (ESPs) (De Groot et al., 2010a). Consequently, the ESFs and ESSs are also influenced by land management (van Oudenhoven et al., 2012; Verburg et al., 2009). The fact that land management can also influence ESSs that are not targeted by this management is often neglected (Fagerholm et al., 2012; Fisher and Turner, 2008; Hein, 2010; Reyers et al., 2009). This underlines that the interconnection between land management, ESPs, ESFs and ESSs is still poorly understood (De Groot et al., 2010b; van Oudenhoven et al., 2012).

Therefore, our study focused on the interactions between land management, ESPs, ESFs and ESSs. Our objective was to model the effect of land management on multiple ESSs. Based on a stepwise framework (van Oudenhoven et al., 2012) we developed generic models in an ArcGIS (ESRI, 1993) spatial modelling environment and applied the models in 'Het Groene Woud', a rural area of 350 km² in the south of The Netherlands (Figure 3.1). We analysed ESSs provided in this Dutch landscape, where different land use types and landscape elements are present. We used multiple indicators per service to quantify, map and model ESSs at this landscape scale. These indicators were related to land management variables such as land use types and intensities, landscape pattern and green and blue landscape elements. Green and blue landscape elements are the hedgerows, tree patches, brooks and fens that intersect the landscape (Kuiper and de Regt, 2007). Finally, we quantified the effect of land management on the provision of ESSs under two simple management scenarios.

3.2 Methods

3.2.1 Study area: Dutch National Landscape 'Het Groene Woud'

The 'Groene Woud' area (350 km^2) is located in the southern part of the Netherlands in the province of Noord-Brabant, amidst three densely populated towns: Eindhoven,'s-Hertogenbosch, and Tilburg. The cities account for 80% of the population of the region (roughly 650 000) (CBS, 2011). The Groene Woud is characterized by a mosaic landscape of cropland, grassland, semi-natural forests, small sand dunes, heath lands, rural settlements and small landscape elements (Figure 3.1). The main targeted sectors of the regional policy are agriculture, tourism/recreation, and nature, which has to be maintained, increased and conserved, respectively (Het Groene Woud, 2011; Streekraad Het Groene Woud en De Meierij, 2008).

In 2005, the area was declared as a Dutch National Landscape (Ministries of VROM (Housing Spatial Planning and the Environment), 2006). This meant that new policies and initiatives have to contribute to conserving the area's unique cultural-historical, natural and landscape features while not compromising local economic activities (Kuiper and de Regt, 2007; Ministries of VROM (Housing Spatial Planning and the Environment), 2006). Improved landscape heterogeneity, multi-functionality and connectivity of green and blue landscape elements are the aims of the regional management strategy (Blom-Zandstra et al., 2010; Kuiper and de Regt, 2007; Opdam et al., 2009). Regional policy and management are closely linked through the local council (‘streekraad’), which translates policy options into management plans (Het Groene Woud, 2011; Streekraad Het Groene Woud en De Meierij, 2008). Large segments are included in the Dutch Ecological Main Structure (EHS) and European Natura 2000 networks (Blom-Zandstra et al., 2010). Nature areas are connected by ecological linkage zones, to preserve habitat and biodiversity through sustainable ecological and economic management (Bredenoord et al., 2011; European Comission, 2011). A biodiversity hotspot and important recreation area is the Kampina Nature Reserve (Figure 3.1). We selected the case study area because of the link between policy and regional management, and the big role that green and blue landscape elements play in the policy and management plans. The area has been used also as a case study location by Speerpunt Ecosystem & Landscape Services (www.ecosystemservices.nl), a research program of Wageningen University and Research Centre (UR).

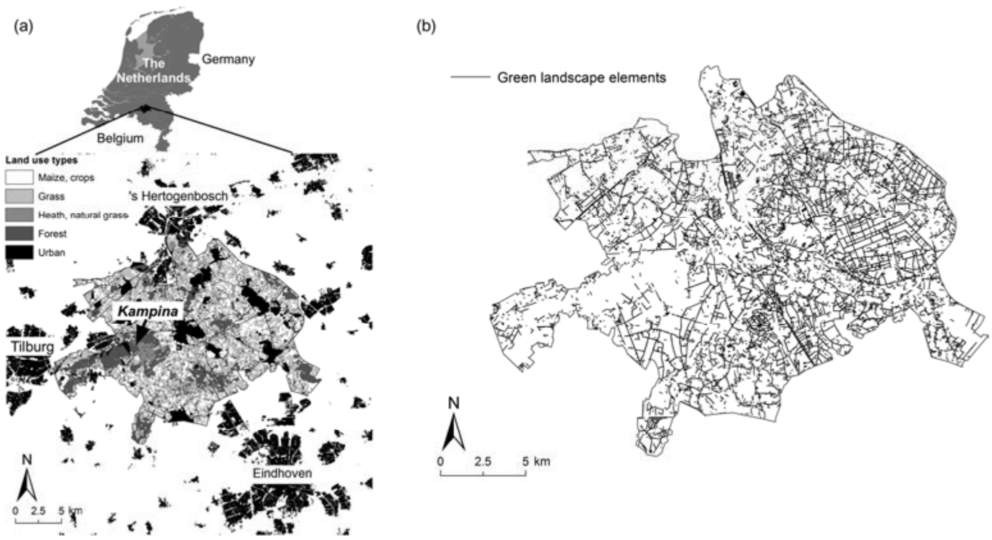

Figure 3.1: The two maps indicate the main land use types (a) and location of green landscape elements (b) in the study area. The land use legend refers to the study area map. Data source: De Wit et al. (1999) and Grashof-Bokdam et al. (2009a).

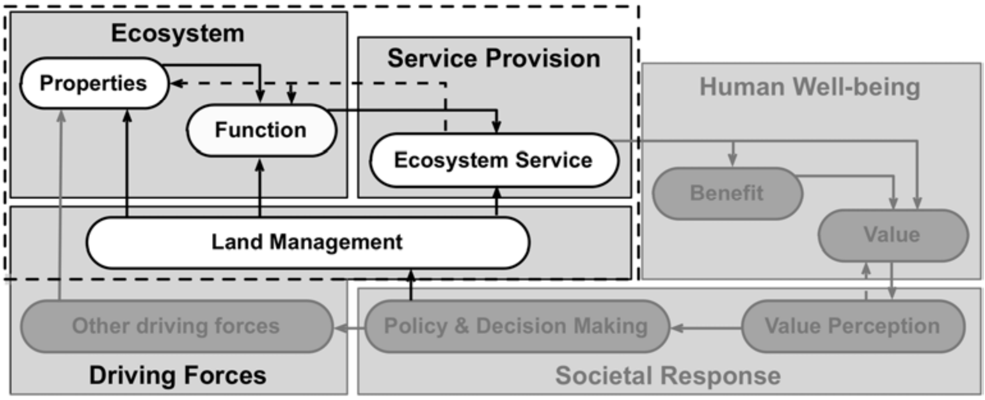

Figure 3.2: Framework for linking land management to ecosystem services. The white boxes in the dotted cadre indicate the focus of our research. Solid arrows indicate effects; dashed arrows indicate feedbacks. Source: Adapted from Van Oudenhoven et al. (2012).

3.2.2 Methodology

We used the following steps to quantify and model ESFs and ESSs: (1) ESSs selection; (2) indicator selection and quantification of ESFs and ESSs; and (3) ESF and ESS modelling. Finally, we also analysed how ESSs would change under alternative management scenarios. Our approach follows the stepwise framework of Van Oudenhoven et al. (2012) (Figure 3.2).

Ecosystem services selection

ESSs were selected, because they had been mentioned by local sources (websites and brochures), stakeholders (regional council members, scientists and farmers) or in scientific literature and reports (Bianchi et al., 2008; Blom-Zandstra et al., 2010; Grashof-Bokdam et al., 2009a; Oosterbaan et al., 2009). Thus, each studied ESS was important to policy-makers, regional management, local inhabitants and/or visitors of the area. We selected food production (milk), production of raw materials (fodder), air quality regulation, climate regulation, pollination, biological control, lifecycle maintenance and opportunities for recreation. We followed the ESSs typology presented in The Economics of Ecosystems and Biodiversity study (www.teebweb.org) as introduced by de Groot et al. (2010a). The selected services represent all four ESSs categories (provisioning, regulating, habitat and cultural) and reflect the three main sectors that are targeted by regional policy.

Indicator selection and quantification of ecosystem functions and services

For each selected ESS, we identified ESP, ESF and ESS steps as well as corresponding indicators. Important criteria for indicator selection were flexibility and data availability. In addition, each indicator needed to be spatially explicit, portable, credible and sensitive to changes in land management (De Groot et al., 2010b; Niemeijer and de Groot, 2008; Reyers et al., 2010). Examples of relevant land management components include land use type, landscape pattern, crop type, noise level and others. Information on indicators and data were collected from scientific and grey literature. Below we provide an overview of the studied ESFs and ESSs, as well as their assumed relationships to management and ESPs. A complete overview of all indicators and relationships can be found in Appendix 1. In Sections 'ESF and ESS modelling' and 'Scenario analysis; shift to extensive and intensive land management' we describe how the selected indicators were used for modelling.

Food production (milk). About 43% of the area is grassland that is used for grazing and milk production (De Wit et al., 1999; Kuiper and de Regt, 2007). The amount of milk that can be produced (ESS) is dependent on the grassland area in combination with the number of milk-producing cows (ESF). Milk production is also influenced by other external inputs, such as nutrient application, veterinarian measures, labour and mechanisation (van Oudenhoven et al., 2012). We did not quantify these external inputs as contributions to the ESS provision. To calculate the amount of milk that can be potentially produced, we assumed that all milk cows feed on grass (no pens) and all grasslands are used for grazing. An average number of 150 cows graze on 100 ha in Noord-Brabant, which means that about 0.66 ha is available per cow (LEI and CBS, 2010). Currently, about one-third of the cows are kept as milk cows in the area (Naeff and Smidt 2009). Based on national statistics (LEI and CBS, 2010) we calculated the number of cows that could graze and the amount of milk that could be produced, thereby comparing organically and conventionally kept cows.

Production of raw materials (fodder). About 16% of the area is under maize cultivation (De Wit et al., 1999; Kuiper and de Regt, 2007). The maize is utilized as fodder and manure resulting from dairy farming is used to enhance maize production (Naeff and Smidt 2009). Manure application, mechanisation and other external inputs enhance maize production. We did not quantify these inputs, but assumed that the area on which maize is cultivated (ESF) determines the amount of maize that can be produced (ESS). We used data on maize production from the Dutch Agricultural Database (LEI and CBS, 2010).

Air quality regulation. Vegetation plays a role in air quality regulation, for instance by capturing volatile organic compounds, ozone and fine dust (Hiemstra et al., 2008; McDonald et al., 2007). PM10 is particulate matter with a diameter of 10 µm or less (Bealey et al., 2007; Beckett et al., 1998). Local agriculture and traffic account for 8% of the total PM10 emission (444 t/year) in the Groene Woud, while the rest originates from outside the area (Bleeker et al., 2008). A way to calculate the potential service is by calculating the difference between PM10 emission and potential PM10 capture in the area

(Oosterbaan et al., 2006). The amount of PM10 (kg/ha/year) captured by vegetation (ESF) leads to a decrease in atmospheric PM10 concentration (ESS) both on a local and a (sub-)national level (Bealey et al., 2007; Beckett et al., 1998; McDonald et al., 2007). We used the capture of vertically deposited PM10 as an ESF indicator, because of high uncertainties and lack of data that exist for horizontal deposition (Oosterbaan et al., 2009). Data on estimated PM10 capture per land cover/land use type by Oosterbaan et al. (2006; 2009) were used. We adjusted this to the average PM10 concentration of 26 µg/m³ in the area (Velders et al., 2007). We interpolated PM10 capture data for additional land use types (e.g. heath and natural grass) and for green landscape elements. The amount of PM10 captured by green landscape elements and all land use types was added up. As a next step we estimated the local atmospheric PM10 emission reduction (ESS) by forest, heathland, natural grass and green landscape elements, based on studies conducted near highways and roads in the Netherlands (Weijers et al., 2000; Wesseling et al., 2008) and in urban and rural areas in the United Kingdom (Bealey et al., 2007; Beckett et al., 1998). The decrease in local atmospheric fine dust concentration is thought to be proportional to the percentage of vegetation cover: 25 % vegetation cover can maximally reduce the PM10 concentration by 15 % (Bealey et al., 2007; Stewart et al., 2002; Tonneijck and Swaagstra, 2006). The atmospheric PM10 concentration varies considerably with increasing distance to emission sources (Janssen et al., 2008), but little is known about the relation between distance-to-source and atmospheric concentration reduction. Therefore, we did not consider the distance to emission sources. Note that we did not relate data on PM10 capture (ESF) to local PM10 concentration reduction (ESS), because no studies could be found that linked these two aspects of air quality regulation.

Climate regulation. Forest and other vegetation types play a role in climate regulation (Baveco and Bianchi, 2007; Brandes et al., 2007; European Environmental Agency, 2009). In the Netherlands, forests sequester about 2.5 Mt CO_2, whereas agricultural grasslands emit 4.2 Mt CO_2 and urban areas emit 0.2 Mt CO_2 annually (Brandes et al., 2007; Schulp et al., 2008). The amount of carbon sequestered (ESF) leads to a decreasing atmospheric CO_2 concentration (ESS) (Adair et al., 2009; Schulp et al., 2008). We used country-level carbon sequestration data (tC/ha/year) for grassland, cropland and forest to map carbon sequestration or emission (Kuikman et al., 2003; Schulp et al., 2008). We assumed the sequestration rate of forest also for heath and natural grass (Ruijgrok, 2006). The carbon pool of urban areas is highly variable (Lorenz and Lal, 2009) and urban carbon exchange is estimated to be low in comparison with other land use types in the Netherlands (Brandes et al., 2007). Therefore, we considered urban areas as carbon neutral. The carbon emitted by transport and infrastructure (e.g. heating) was excluded. Furthermore, carbon sequestration by green landscape elements was not considered, because the country-level input data did not include applicable sequestration rates. The sequestered carbon multiplied by CO_2-equivalency constant (3.67) gives the CO_2-equivalent of the carbon sequestrated or emitted; a proxy for changes in atmospheric CO_2 concentration (Environmental Protection Agency, 2005; Gohar and Shine, 2007)

Pollination. Several crops, such as beets and various vegetables, are dependent on natural pollinators in the Groene Woud (De Wit et al., 1999). Pollination by wild bees is of great economic importance to farmers cultivating pollinator-dependent fruits and vegetables (Gallai et al., 2009; Priess et al., 2007). The abundance of pollinators (ESF) within a given proximity of croplands affects crop yield (ESS) (Klein et al., 2007). We used fruit set, the percentage of flowers that develop into fruits, as a proxy for the pollinator wild bees` abundance (ESF) and adopted the fruit set-distance curve from Steffan-Dewenter and Tscharntke (1999). The maximum fruit set is 60%, which tends to drop to about 20% with increasing distance from nature i.e. forest, heathland and natural grass (Steffan-Dewenter and Tscharntke, 1999). The positive effect of forest and natural grass on crop pollination diminishes beyond approximately 1200 – 1500 m (effective distance) (Priess et al., 2007; Steffan-Dewenter and Tscharntke, 1999). The service itself, the crop yield can be provided only in areas with pollination-dependent crops. We assumed that the ESS follows the pollinator abundance, which means that at the maximum fruit set of 60% the yield is 100%.

Biological control. Many crops, such as wheat, maize and various vegetables, that are grown in the Groene Woud can be severely affected by pests, mainly insects (Bianchi et al., 2006; Gurr et al., 2003). We considered biological control the predation of insect pests by natural predators. The abundance of natural predators (ESF) can decrease the numbers of pests (ESS) and thereby can decrease damage to crops (Clough et al., 2007; Foster et al., 2004; Oelbermann and Scheu, 2009). Forests and hedgerows provide a habitat for the natural predators of pests such as aphids attacking cereals and moths attacking vegetables (Foster et al., 2004; Roschewitz et al., 2005). We used egg predation of crop pest as the ESS indicator for biological control. Bianchi et al. (2008; 2006) and Levie et al. (2005) proved an increase in predation on insect pests as a result of green landscape elements. We used information from studies in the Netherlands on the relation between landscape configuration, green and blue landscape elements and predation on two moth species occurring in cabbage and sprout fields: the diamondback moth (*Plutella xylostella*) (Baveco and Bianchi, 2007; Bianchi et al., 2005) and cabbage moth (*Mamestra brassicae*) (Bianchi et al., 2008). Bianchi et al (2008) showed that egg predation rates increase with increasing area of forest edges within a 1000 m distance. We mapped the density of forest and green landscape elements to determine the natural predation rate. The service is provided in areas that can be affected by agricultural pests: orchards, beets, maize, cereals and non-cereal crops.

Lifecycle maintenance. The Groene Woud area plays an important role in providing habitats for migrating and local animal and plant species. We selected the habitat provided for butterflies to measure lifecycle maintenance. The habitat suitability (ESF) is related to the occurrence of species (ESS). We used butterflies occurring in closed connected woody habitat (forest and forest patches) as indicator species. Butterflies are generally more mobile in continuous landscape (Baguette et al., 2003) and their occurrence and species richness increases with higher amounts of deciduous forest (Bergman et al., 2004). Therefore, we mapped the density of forest and green landscape elements within the species' dispersal distance, taken as 1750 m, to obtain habitat suitability (%) (Grashof-Bokdam et al., 2009a). We also assessed the effect of fragmentation and nature protection. Landscape fragmentation has a negative effect on butterfly mobility (Baguette et al., 2003), which we translated as exponentially decreasing habitat suitability within a 1000 m buffer of roads and railways, similar to Tallis et al. (2011). Nature protection, as a result of Natura 2000 and EHS networks is beneficial for species (Blom-Zandstra et al., 2010; Bredenoord et al., 2011; European Comission, 2011). Therefore, we assumed 30% and 20% habitat suitability increase for Natura 2000 and EHS areas, respectively. We assumed that butterfly species occur in areas with a minimum of 50% suitability, with suitability ranging between 0% and 100%.

Opportunities for recreation. We used the activity walking to measure recreation. Walking is the most popular recreation activity in the Netherlands; 60% of the population walk regularly for pleasure, whereas 50% cycle (CBS, 2010). The suitability of an area for walking (ESF) largely determines how many people can walk (ESS). Walking suitability is based on properties such as the land use type, noise level and diversity of landscape, all in relation to people's preferences (De Vries et al., 2007; Goossen and Langers, 2000; van den Berg et al., 1998). We used a combination of the most influential indicators from countrywide studies by Goossen and Langers (2000) and De Vries et al. (2007). Interview-based data from Goossen and Langer (2000), were used to map most preferred land use types for walking. We added the effect of noise level and landscape diversity. The national noise maps (obtained for roads and railways from www.rijkswaterstaat.nl and www.prorail.nl, respectively) indicate increased noise level within a 500 m buffer of roads and 400 m buffer of railways. A noisy environment is not preferred for walking (Goossen and Langers, 2000) and we assumed that noisy locations decrease walking suitability by up to 80%. A diverse landscape was found to be attractive for recreants (van den Berg et al., 1998). We measured landscape diversity as the proximity of green landscape elements. We assumed that within the 100-200 m distance of green landscape elements walking suitability increases by 30-10%. The number and distribution of people that walk depends on the walking suitability, the percentage of residents that walk (60%) and the number of residents (650 000 people) (CBS, 2010, 2011). We assumed that people walk (ESS) in areas with a minimum of 60% walking suitability.

Ecosystem function and service modelling

The above-described relationships served as a base for modelling each ESF and ESS:

- Ecosystem properties = F (Land use, Green landscape elements, Other management variables)
- Ecosystem function = F (Ecosystem properties, Other management variables)
- Ecosystem service = F (Ecosystem function, Other management variables)

Figure 3.3 shows the schematic overview of the climate regulation model, as an example. Data from the LGN3+ (Dutch land use database, Landelijk Grondgebruiksbestand Nederland in Dutch) land use map (De Wit et al., 1999) and green landscape elements map (Grashof-Bokdam et al., 2009b) were the main data input for the model (Figure 3.1); quantified ESF and ESS map the output. The resolution of all maps was 25 x 25 m. For on overview of the relationships per ESS, see Appendix 1. For each ESS model a graphical representations built in ArcGIS 9.3 can be seen in Appendix 2.

Scenario analysis: shift to extensive or intensive land management

To further analyse the effect of land management on the provision of ESSs, we developed two scenarios: (1) *Intensive agriculture* and (2) *Functional nature protection*. We quantified the services food production (milk), air quality regulation and opportunities for recreation – which are examples of, respectively, a provisioning, regulating, and cultural service –under the two scenarios. We selected these services, because they feature in the sectors targeted by regional policy. Moreover, the services can also be quantified and aggregated for the entire study area. Our scenarios were based on the ‘Suitable Nature’(or ‘Tailored Nature’) and ‘Functional Nature’ scenarios developed by PBL (2011) as part of the Dutch Nature Outlook (‘Natuurverkenning’). The delineation of the two scenarios was based on the main land use types (cropland, grassland and forest), sectors mainly target by regional policy (agriculture, tourism and recreation and nature conservation) and agricultural production intensities (intensive and organic) in the area. The scenarios were translated into changes of land management-related variables, namely land use change, land cover change (green landscape elements) and local PM10 emission (Table 3.1).

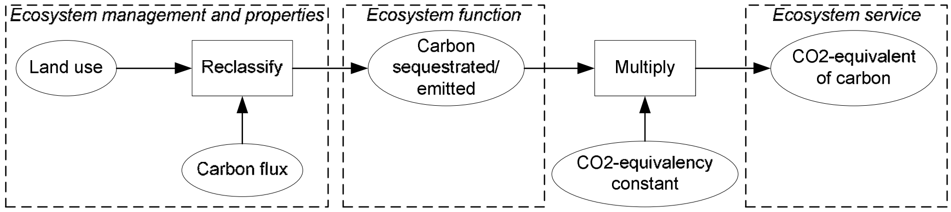

Figure 3.3: Schematic overview of the climate regulation model. Round boxes indicate inputs/outputs, and square boxes indicate processes or tools used in ArcGIS 9.3 to derive outputs.

Under the *Intensive agriculture* scenario, a shift towards large-scale mono-functional agricultural production is assumed. This is in line with the 'Suitable Nature' scenario, which assumes limited intervention by national governments and more trust in market functioning. Nature is utilised mainly for the provision of services with a direct market value, such as agriculture and recreation (PBL, 2011). This is illustrated by the random conversion of 57% of deciduous forest and forest patches into grassland and the clearance of green landscape elements. This would result in a 15% increase of grassland area and, consequently, increased land on which milk cows could graze.

Under the *Functional nature protection* scenario a shift towards organic food production, with no changes in the location and extent of small-scale land use was assumed. The increased focus on nature and biodiversity conservation would be realized through ecological corridors, protection and environmental sound management. We illustrated this by the maintenance, but no further expansion of the existing green landscape elements. This is in line with the 'Functional Nature' scenario, which assumes increased involvement of local stakeholders in decision making and increased awareness of an attention to the benefits of nature, both in financial and non-monetary terms (PBL, 2011). We therefore assumed no changes in PM10 emissions, in the total area of different land use types, and in the coverage of green landscape elements.

Table 3.1: Land management characteristics under the two scenarios: (1) *Intensive agriculture* and (2) *Functional nature protection*. Vegetation cover refers to the area of forest, heath, natural grassland and green landscape elements.

1. Intensive agriculture	**2. Functional nature protection**
Forest patches (3 100 ha) converted into grassland (16 500 ha in total)	No changes in land use areas (14 400 ha grass, 5 400 ha forest)
Conventional milk production (→ 8 000 L milk/cow/year)	Switch to organic milk production (→ 6 600 L milk/cow/year)
20% increase of PM10 emission by agriculture (533 t/year dust emission)	No changes in PM10 emission by agriculture (444 t/year dust emission)
Clearance of green elements 6% vegetation cover	No change in green elements coverage (~5 100 ha) 31% vegetation cover

We quantified the three services under the two land management scenarios (Table 3.1) and using the relationships specified in Sections 'Indicator selection and quantification of ESFs and ESSs' and 'ESF and ESS modelling'.

3.3 Results

3.3.1 Modelled ecosystem functions and services

In this section, numbers and maps are shown for eight quantified and modelled ESF and ESS. Note that we only provided separate maps of ESF and ESS if the spatial pattern of the function and service maps were different.

Food production (milk).

The 14 400 ha of grassland provide grazing area for 7 200 milk cows (Figure 3.4a). A conventional cow can produce 8 000 L of milk per year (LEI and CBS, 2010) and an organic cow 6 600 L (LEI and CBS, 2010). Based on that we calculated that roughly 57 600 kL of non-organic milk or 47 520 kL organic milk can be produced yearly from the milk cows that feed on grass.

Production of raw materials (fodder)

Maize is cultivated on 5 500 ha (Figure 3.4b). The average silage maize yield in 2010 was 45 t/ha (CBS, 2011), resulting in 250 000 t/year maize production in the Groene Woud.

Air quality regulation

Coniferous forests can capture 94 kg PM10/ha/year (high); deciduous forests 54 kg PM10/ha/year; and heathland, natural grass and green elements 27 kg PM10/ha/year. The rest of the land use types can capture less than 15 kg/ha/year (low) and we assumed that urban areas capture no fine dust (Figure 3.4c). In total, 644t PM10 can be captured by the vegetation annually, which means that the total amount of PM10 emitted within the area (444t) can be captured by vegetation. The 31% vegetation cover (forest, heath, natural grassland and green elements) in the Groene Woud is estimated to contribute to a 10-15% reduction of the local PM10 concentration.

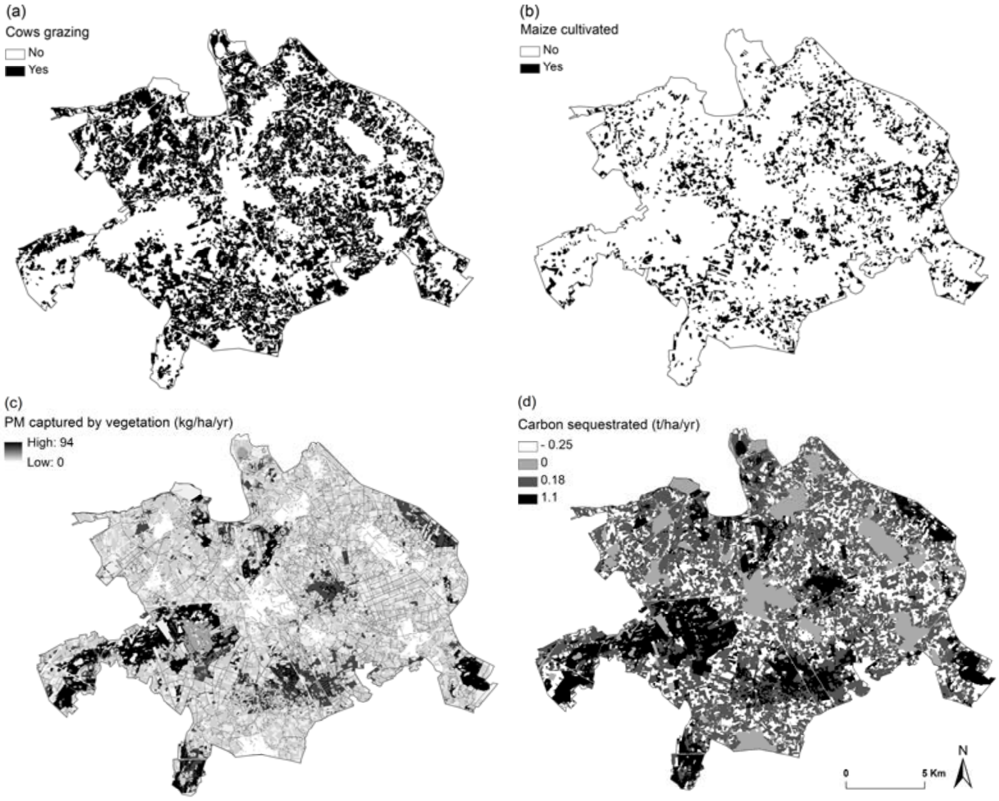

Figure 3.4: Ecosystem function maps for milk production (a), fodder production (b), air quality regulation (c) and climate regulation (d). Ecosystem service maps show similar spatial pattern to the function maps and are therefore not provided.

Climate regulation

Carbon sequestration rates are the lowest on cropland (-0.25 tC/ha/year) and urban area (0 tC/ha/year), followed by grassland (0.18 tC/ha/year), and are the highest on forest, heath and natural grass areas (1.1 tC/ha/year) (Figure 3.4d). Negative numbers indicate carbon emission. The corresponding CO_2-equivalents of the carbon sequestrated or emitted were, -0.92, 0.00, 0.66 and 4.04 tCO_2-equivalents, respectively.

Pollination

Fruit set varies between 32% (low) and 60% (high), and high fruit set occurs near green elements and nature (Figure 3.5a). The service is only provided in cropland areas that depend on natural pollination, thus the service map differs from the function map. The change in crop yield follows the trend in fruit set curve and ranges between 72% and 100% (high) on pollination-dependent crop fields and is 0% (low) in other areas, which do not benefit from natural pollination (Figure 3.5b).

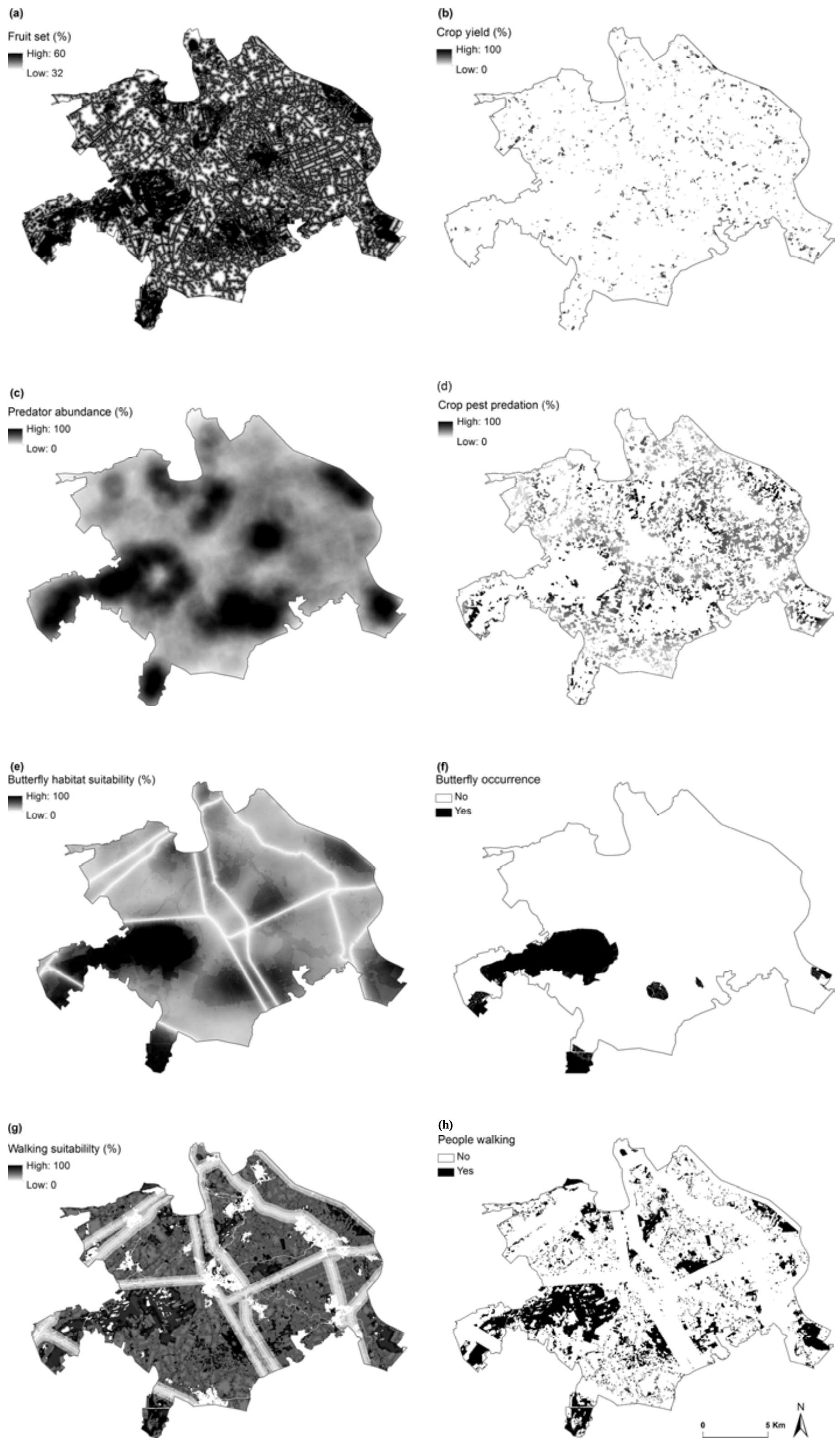

Figure 3.5: Maps of pollination function (a) and service (b); biological control function (c) and service (d); lifecycle maintenance function (e) and service (f); opportunities for recreation function (g) and service (h).

Biological control

The pest predation at crop areas follows the curve of abundance of natural predators of insect pests, with highest predation possible on croplands near forests and green elements (Figures 3.5c and 3.5d). The service is only provided in areas that can be affected by agricultural pests, thus the service map covers only a part of the function map.

Lifecycle maintenance

Butterflies occurring in closed woody habitats live primarily in non-fragmented forests. Therefore, the most suitable habitats are protected forest areas (100%, i.e. highest suitability) and least suitable areas occur near roads and railways (0%, i.e. lowest suitability). The Kampina Nature Reserve is a large area with the highest habitat suitability (Figure 3.5e). The service is provided in areas with at least 50% suitability, which equals 10% of the total area. Therefore, the service map covers only a part of the function map and it mainly comprises the Kampina Nature Reserve (Figure 3.5f).

Opportunities for recreation

The combination of forest and heathlands with low noise levels provides the highest suitability for walking (100%), whereas noisy areas along roads and railways are the least suitable for walking (0%) (Figure 3.5g). About 60% of the area's residents walk regularly, which is about 390 000 people. Assuming that people walk only in areas with at least 60% suitability, walking would occur at 19% of the area (6 265 ha). This leads to a walkers' density of 62.2/ha. Therefore, the service map covers only a part of the function map and mainly comprises the Kampina Nature Reserve and some other small patches of the Groene Woud (Figure 3.5h).

3.3.2 Scenario analysis: shift to extensive or intensive land management

The outcome of the (1) *Intensive agriculture* and (2) *Functional nature protection* scenarios was quantified for milk production, air quality regulation and opportunities for recreation functions and services (Table 3.2). Under the *Intensive agriculture* scenario, more milk could be produced (66 ML/year) as compared with the *Functional nature protection* scenario (47.55 ML/year). This is the result of the increase in grassland area (15%) as well as the larger number and higher productivity of conventionally kept cows (8 250) compared with organically kept cows (7 200). More PM10 could be captured (644 vs. 359 t/year) and the area with high walking suitability (above 60%) is largest in *Functional nature protection* (6 362 vs. 4 360 ha). The higher PM10 capture in *Functional nature* *protection* is caused by the fact that coverage by green elements and forest area are maintained and PM10 emissions remain constant. All locally emitted PM10 (444 t/year) could be captured. Assuming a 10-15% decrease of local PM10 concentration can be achieved by 25% vegetation cover (Bealey et al., 2007; Stewart et al., 2002), *Functional nature protection* (31% vegetation cover) could lead to more and *Intensive agriculture* (6% vegetation cover) to less than 10-15% decrease. Similarly to air quality regulation, better opportunities for recreation in *Functional nature protection* are a result of maintained coverage of green landscape elements and forest. The fact that *Functional nature protection* would result in larger area with high walking suitability than *Intensive agriculture* has consequences for the potential number of walkers per hectare. With the same number of people that can walk in the area (390 000 in each scenario), the walkers density *in Intensive agriculture* is 89.4/ha and in *Functional nature protection* is 61.3/ha. To sum up milk production is highest in *Intensive agriculture*, whereas recreation and air quality regulation have highest values in *Functional nature protection*.

Table 3.2: Quantified results of two scenarios: (1) *Intensive agriculture* and (2) *Functional nature protection* for three ecosystem functions and services.

		Scenario	
ESS	**Function/ service**	**1. Intensive agriculture**	**2. Functional nature**
Milk production	Function	8250 milk cows (conventional)	7200 milk cows (organic)
	Service	66 ML milk/year	47.5 ML litre milk/year
Air quality regulation	Function	396 t/year PM10 captured	644 t/year PM10 captured
	Service	74% of emitted PM10 captured Max. 5% reduction of PM10 concentration	All emitted PM10 captured Max. 15% reduction of PM10 concentration
Recreation	Function	13% of the area (4360ha) is above 60% walking suitability	19% of the area (6365ha) is above 60% walking suitability
	Service	390000 walkers 89.4 walkers/ha	390000 walkers 61.3 walkers/ha

3.4 Discussion

3.4.1 Modelling the effect of land management on ecosystem services

Method

Each ESS was studied through a combination of “simplifying” indicators and generalized relationships between indicators for ESPs, ESFs and ESSs. The relationships were established based on the assessment of multiple sources for each service. Many indicators, mostly at the ESPs level could be used for multiple services, indicating a possible step towards the assessment of ESSs in bundles. All services and function were modelled in the same ArcGIS modelling environment and on the same level of the spatial scale (i.e. landscape), which enabled a quantitative and spatial comparison of ESSs. Previous studies focused on multiple services which were mainly related to water, carbon sequestration, pollination, and recreation (or tourism) (Bai et al., 2011; Chan et al., 2006; Egoh et al., 2011; Reyers et al., 2009), but services such as biological control or air quality regulation were hardly analysed in combination with other services. Therefore, we attempted to assess a wide range of services. We also established explicit links between ecosystem ESPs, ESFs and ESSs. The difference between ‘what the landscape offers’ (ESF) and ‘what is or can be used by people’ (ESS) informs us on the potential of the system to provide a service as well as on the sustainable use of the service (Haines-Young and Potschin, 2010; Kakembo and Rowntree, 2003; Kienast et al., 2009). In the case of pollination and biological control, the function covers a larger area than the service, which means that not all the capacity is used and there is potential for the increased use of the service (Figure 3.5a-d).

Similarly to Lamarque et al. (2011) and Reyers et al. (2009) we linked fodder and milk production to yield and animal numbers, respectively. Information on land use and agricultural statistics was combined into a set of simple but reliable relationships. We also used land use-based indicators for air quality and climate regulation. A consequence of this method is that results are spatially explicit and land use-specific, but lack the dynamic biophysical and management aspects (e.g. nutrient application and tree extraction rate) of the service provision.

Bai et al. (2011), Reyers et al. (2009) and Swetnam et al. (2011), among others, mapped carbon sequestration by vegetation or land use type, but did not relate it to climate change directly. It must be noted that relationship between carbon sequestered and the change in atmospheric CO_2 concentration is complex and uncertain. We used the widely used CO_2-equivalent to estimate changes in atmospheric CO_2 concentration.

Models that simulate PM10 capture by vegetation (Bealey et al., 2007; McDonald et al., 2007; Tiwary et al., 2009) usually do not relate ESF to ESS indicators, nor do they link the air quality service to other ESSs. We could not link data on fine dust capture capacity of vegetation to changes in atmospheric PM10 concentration directly. Although it is known that vegetation has a positive effect on atmospheric fine dust concentration, little is known about the actual quantitative relations. Air quality can also be influenced and measured by concentrations of other components, such as NO_2, NH_3 and O_3 (Nowak et al., 2006). Oosterbaan et al. (2006; 2009) studied both PM10 and NH_3 in the Groene Woud and claimed that NH_3 proved to be an uncertain component to be modelled at landscape scale, as a result of heavily fluctuating concentrations and fluxes. Horizontal PM10 capture is more difficult to estimate than vertical, therefore we used vertical capture based on deposition velocity influenced by vegetation characteristics, as has been commonly done by others (Beckett et al., 1998; Nowak and Crane, 2000; Oosterbaan et al., 2006). Vertical PM10 deposition has been estimated to account for 60-80 % of the total dust captured (Oosterbaan et al., 2006), but due to uncertainties we did not use this information.

Pollination and biological control were modelled before with agent-based models (Kareiva et al., 2011; Kremen et al., 2007; Lonsdorf et al., 2009), with the focus on animal behaviour. Pollination was also mapped and modelled spatially (Chan et al., 2006; Kareiva et al., 2011), but with no clear distinction between function and service. We generalized and applied prior established spatial relationships to model pollination, biological control and lifecycle maintenance. Studies conducted on the spatial effect of forest on crop pollination in other regions showed similar numbers on effective distance and underlined the positive effect of forest on crop pollination, but showed different numbers on fruit sets (60-85%) (Priess et al., 2007). The generalized value of fruit set percentages should be treated with caution, because studies show that fruit set percentages are crop- specific.

Lifecycle maintenance can be measured and modelled through species number (Chan et al., 2006), mean species abundance (Alkemade et al., 2009), habitat rarity and habitat integrity (also referred to as fragmentation) (Tallis et al., 2011), among others. Similar to Tallis et al. (2011) we established quantified and distance relationships between land management and ESPs related to habitat suitability to map lifecycle maintenance. Scientific literature only supports the positive effect of nature protection on species (habitat). The assumed 20-30% habitat suitability increase, as a result of nature protection, was an assumption used for this case study. The choice of indicator also determines output maps; location of forest patches, for instance, influenced the lifecycle maintenance function map and spatial pattern of green elements influenced the pollination function map (Figures 3.5e and 3.5a). Choosing different indicators may lead to different results, which means that the indicator choice involves uncertainty.

Recreation was measured and modelled before by including factors such as proximity to roads, level of public access, amount of natural land cover (Chan et al., 2006) and view shed (Reyers et al., 2009). We used walking as an indicator for recreation, due to the popularity of the activity. We studied recreation rather than tourism, as walking trips would be regarded as touristic activities if a night was spent in an accommodation in the area (CBS, 2010; Henkens et al., 2005). Therefore, motives and indicators for tourism could be different. A diverse landscape has a positive effect on recreation (van den Berg et al., 1998). Nevertheless, the 10-30% walking suitability increase as a result of landscape diversity was an assumption made for this case study. Furthermore, there are also other aspects of landscape diversity (such as topography and waterways) that we did not consider.

Results

The function and service maps provide location-specific information about the effect of land management on the provision of ESSs. The reliability and accuracy of the ESS models and uncertainty of the results depend on the quality of the input data and on the model relationships. For example, information on fodder production was derived directly from statistics of maize production. We used national aggregated, yearly updated statistics, which give a rough indication of the fodder production. Using regional, location-specific data might lead to results that are more accurate. Similarly, the climate regulation function map is derived directly from country-level land use-specific carbon sequestration data. The carbon sequestered (ESF) by different land use types shows a similar trend with the results of studies conducted in other parts of world (Chan et al., 2006; Swetnam et al., 2011), namely, that deciduous forests sequester the highest amount of carbon. For milk production we compared the modelled number of cows (7 200) with results from the agricultural database (10 020) (Naeff and Smidt 2009). The lower model result can be attributed to the fact that cows might have a smaller area in the Groene Woud than the provincial average we used and, therefore, more cows can be kept in reality. Although the 165 t/km^2 average milk production in the Groene Woud (calculated as non-organic milk produced/total area) is relatively low, it falls within the 100-500 t/km^2 range indicated on the national milk production map (in 2008) (Oostenbrugge et al., 2010).

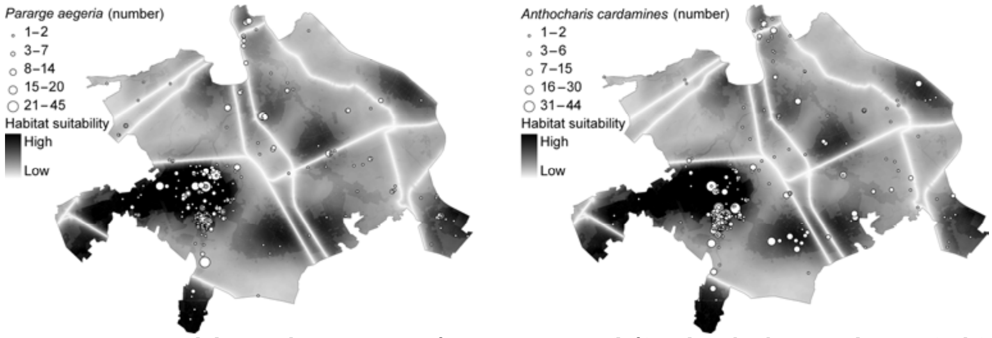

Figure 3.1: Empirical data on the occurrence of *Pararge aegeria* (left) and *Anthocharis cardamines* (right) overlaid with the modelled habitat suitability. The majority of butterflies of both species (64% and 58%, respectively) were found in areas with suitability higher than 50% (black).

For air quality regulation and climate regulation, we mapped ESFs by using land use-specific data of PM10 capture and carbon sequestration. The reliability and accuracy of these results depend on the quality of input data. As discussed above the estimation of the PM10 capture involves uncertainties. Furthermore, the actual contribution of PM10 capture to a lower PM10 concentration and the actual contribution of carbon sequestration to a lower CO2 concentration were difficult to estimate. In other words, it proved to be difficult to make the link to the service itself. That is why studies often describe either the PM10 capture or the modelled decreasing concentration. To our knowledge, Bealey et al. (2007) were the only ones to have modelled both aspects, and they studied a location that was comparable to the Groene Woud (densely populated urban environment in the United Kingdom), which is why we used their assumptions and averaged results for our model.

We tested and validated the modelled relationships and assumptions by comparing and backing up them with other studies. No studies on pollination have been conducted in the Netherlands (Van Rijn and Wäckers, 2007). We made use of a number of studies from different locations to derive information on pollination, which we discussed above. Furthermore, the importance of green landscape elements for pollination, biological control, lifecycle maintenance has also been backed up by literature.

For lifecycle maintenance, we compared the habitat suitability map with empirical observation data on the occurrence of two closed woody habitat butterfly species (1993-2010): *Pararge aegeria* and *Anthocharis cardamines* (DBC, 2011). We found that about 64% of *P. aegeria* and 58 % *A. cardamines* butterflies occur in areas with modelled habitat suitability higher than 50% (Figure 3.6). Hence, the actual butterfly density is higher at areas with higher modelled habitat suitability.

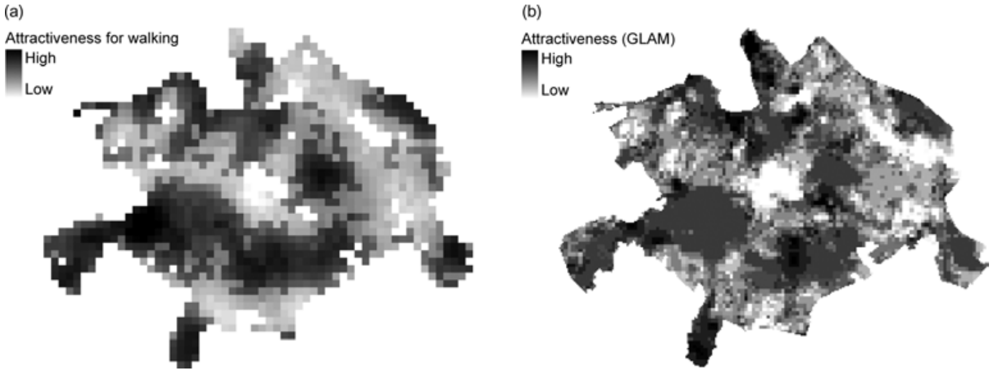

Figure 3.2: Attractiveness for walking (a) based on accessibility, land use preference, social aggression, tranquillity and crowding (Goossen et al., 1997; Goossen and Langers, 2000); and attractiveness of Dutch landscape (b) based on naturalness, relief, urbanization, skyline disturbance, historical distinctiveness and noise level (GLAM-2 model) (De Vries et al., 2007).

We compared the walking suitability map with a national map on attractiveness for walking (Goossen et al., 1997; Goossen and Langers, 2000) and a general attractiveness map of Dutch landscapes simulated with the GLAM-2 (GIS-based landscape appreciation model, version 2) (De Vries et al., 2007). The Kampina Nature Reserve scores the best in all the three studies. On our walking suitability map the negative effect of roads and railways is much more visible (Figures 3.5g and 3.7). These similarities and differences can be attributed to the assumptions used in our methodology as well as the indicator choice. Common indicators were land use preference (Goossen and Langers, 2000) and noise level (De Vries et al., 2007; Goossen and Langers, 2000). However, we also used additional assumptions and data, such noise level maps, thereby assuming that noise along roads and railways decreases walking suitability by 60-80%. The added value of our map is that it provides more detailed information on landscape scale. This is also underlined by the higher resolution of our map (25 x 25 m against 250 x 250 m of GLAM-2 model (De Vries et al., 2007) and 1000 x 1000 m of attractiveness for walking (Goossen and Langers, 2000)). About 75% of all walking trips take place within a range of 20 km from dwelling places (CBS, 1997). The whole Groene Woud area is located within 20 km distance from the three surrounding cities, which makes the whole area attractive for walking.

We have shown that the partial validation of the results could be done through performing additional Geographic Information System (GIS) analyses or comparison with other models, maps and quantification studies. In general, it is difficult to perform a uniform uncertainty assessment on all services, because the methods to assess validity and uncertainty may differ per service.

3.4.2 Scenario analysis

The scenario analysis can be considered a first step towards incorporating the ESS models into decision-making on land management. The *Functional nature protection* resembles the current situation most, since there is a lot of attention on the role of green and blue landscape elements in the Groene Woud. Protection of or even increasing the extent of green landscape elements seems very plausible considering it is a core focus of the current regional policy. This is partly due to the fact that the area's current landscape configuration is the result of a local, bottom-up initiative: nature managers, farmers, and municipalities already started working together to connect several nature areas through the addition of green and blue elements to croplands, roadsides and waterways (c.f. Green Blue Cadre, (Noord-Brabant, 2011)). The complete switch to organic milk production might be not realistic because of the currently low (but increasing) demand for organic milk (LEI and CBS, 2010). However, our analysis shows that still large amounts of milk could be produced in the area.

The *Intensive agriculture* scenario naturally does arrive at high milk production, but at the cost of recreation and air quality regulation. A high recreants' density in a limited area suitable for walking would be highly undesired for local stakeholders as well as walkers (Goossen and Langers, 2000). Moreover, only a fraction of the locally emitted PM10 would be captured by the remaining vegetation. All in all, the *Functional nature protection* scenario seems most realistic and yields beneficial results for the area's inhabitants and policymakers.

Our scenario analysis was quantitative, but lacked spatial explicitness. With a spatially explicit analysis, targeted areas could be identified and modelled separately, in order to arrive at a more precise and relevant outcome. Furthermore, it would also enable the analysis of services that cannot be aggregated in quantitative terms, meaning that they are not cumulative, but depend mainly on the landscape structure. Examples of these services are pollination and biological control. For us, the scenario analysis served the purpose of testing the influence of land management-related variables for the three ESSs, and consequently illustrating how this stepwise modelling approach can facilitate making decisions on land management. We showed that land management for the optimization of one service has an effect on multiple services, because management often targets and alters ESPs (e.g. green landscape elements) that contribute to the provision of multiple services. This underlines the importance of stepwise investigation of ESSs and need for defining and quantifying ESFs and ESSs first in order to enable service quantification. Further steps for the scenario analysis would be the assessment of more services, as well as incorporation of economic and social valuation of the services too.

3.4.3 Societal relevance

Our study in the Groene Woud is useful and relevant regarding the current policy and management of the region. Researchers, local farmers and managers were consulted to learn about the local policy, management and their link to ESSs. Improved multifunctionality, connectivity of green landscape elements and the full implementation of the EHS network are target points of the regional management strategy (Blom-Zandstra et al., 2010; Kuiper and de Regt, 2007; Opdam et al., 2009). Furthermore, a recent policy instrument 'Green Blue Cadre' stimulates farmers to improve and diversify ESSs, for example, to place green and blue landscape elements and establish walking paths on field edges (Noord-Brabant, 2011). Our study confirms that the green landscape elements play an important role in the provision of multiple ESSs. Therefore, a 10% increase of green elements (which could be done if the local council agrees) could contribute to increase landscape multi-functionality and ESS provision in the Groene Woud.

3.5 Conclusion

The ESF and ESS maps show a clear trade-off between ESSs provided by the natural and agricultural land use and land cover types. Natural areas score higher in the provision of regulating and cultural functions and services, whereas agricultural areas score higher in the provision of production-oriented services, such as milk and fodder. In addition, we showed that the presence of green elements is beneficial for multiple services, either directly (regulating and recreation services) or indirectly (pollination and biological control enhancing agricultural production). Therefore, land use type and green landscape elements are suitable variables for modelling land management effects in this area. The ArcGIS modelling environment enabled a quantitative and spatial comparison of ESSs, whereas the use of generic relationships enabled the application of the method also for other areas either in or outside of the Netherlands. We conclude that stepwise modelling of ESFs and ESSs is essential to understand better the effects of land management on the provision of ESSs and is a first step towards bundling services. Our scenario analysis offered a preview of how this can be done in a simple way, with still yielding useful results. The societal relevance of the study lies in its implication in regional management and policy. The maps provide location-specific information about the effect of land management on the provision of ESSs at the landscape scale. Further research in the Groene Woud and similar areas should focus on the assessment of more dynamic services, for instance by studying water and nutrient (nitrogen, phosphorus and carbon) dynamics. This is relevant for regulating services such as water retention, water purification, water provision, soil quality maintenance and climate regulation. Cultural services, such as aesthetic information and cognitive development require a qualitative approach, which enables the synthesis of soft and hard information. Therefore, we suggest to combine the stepwise approach we applied with more dynamic and qualitative approaches to get a more complete overview of the bundle of ESSs that can be provided.

Acknowledgements

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The Baviaanskloof Catchment, South Africa

Land management implications for ecosystem services in a South African rangeland (Baviaanskloof)

In South Africa, restoration and sustainable management of historically overgrazed and degraded rangelands are promoted to increase biodiversity and ecosystem service provision. This study evaluates different land management scenarios in terms of ecosystem services on a South African rangeland. As measured data were limited, we used simple models to quantify and map the effect of the different combination of agricultural, nature conservation and restoration practices on multiple ecosystem services. The land management scenarios were evaluated against management targets set for individual ecosystem services. Results highlight how the provision of ecosystem services is related to land management as unmanaged, pristine ecosystems provide a different mix of ecosystem services than ecosystems recently restored or managed as grazing lands. Results also indicate that historically overgrazed lands provide no forage, may retain 40% less soil against erosion and have 38% lower biodiversity, while providing 60% more fuel wood and supplying two and half times more water (i.e. retaining less water), than pristine or restored lands. We conclude that a combination of light grazing, low input agriculture, nature conservation and restoration is the best for the sufficient provision of multiple ecosystem services. Applying such mixed management would improve biodiversity, ecotourism and maintain forage production and regulating services on farmers' land. This management option also fits into and further optimizes local decision-makers' vision regarding the future management of the area.

Keywords: GIS, mapping, ecosystem degradation, thicket restoration, scenario, environmental decision-making

K. Petz, J. Glenday, R. Alkemade (submitted) Land management implications for ecosystem services in a South African rangeland

4.1 Introduction

Land conversion and intensification are major drivers of ecosystem degradation, biodiversity loss and ecosystem services (ESs) depletion (Nelson and Daily, 2010; Pereira et al., 2010). The increasing international concern about biodiversity loss and ESs depletion resulted in the inclusion of ESs in the 2020 Aichi targets set by the Convention on Biological Diversity (Larigauderie et al., 2012). In South Africa, land conversion and overgrazing related to pastoralism impaired biodiversity and ESs, such as long-term forage production and water supply (Le Maitre et al., 2007; Palmer et al., 2006; Van Jaarsveld et al., 2005). Recently targeted governmental environmental programs have been established to support ecosystem restoration, sustainable land management and livelihood improvement (Milton et al., 2003). The Baviaanskloof Catchment was chosen as a watershed-scale example of how policy and management changes could impact ES provision.

As much of Southern Africa, the Baviaanskloof Catchment is a relatively data-poor environment. A few plot-scale studies have been performed on the quantitative effects of vegetation degradation on hydrological and ecological processes in the larger region (e.g. van Luijk et al. (2013), Mills and Cowling (2006) and Lechmere-Oertel et al. (2005a)), but there has been little quantitative monitoring of most ecosystem processes and functions. In such a setting, information about ESs derived from maps and models can improve land management decision-making. In South Africa some ESs have been mapped and modelled using proxies that relate to land cover and land use (e.g. Egoh et al. (2010) and Reyers et al. (2009)). The combination of different land management practices, their impacts on the resulting land cover and ESs, and the effect of potential future management changes have been less studied in the region. In general, the consequences of alternative land use and land management options for a broader range of ESs are poorly quantified (Carpenter et al., 2009; De Groot et al., 2010b) and the integration of multiple ESs into land use and management decisions is still missing (Ehrlich et al., 2012).

This study aims to evaluate alternative land management scenarios by mapping and modelling multiple ESs in the South African Baviaanskloof Catchment. Land management in the area is a combination of multiple agricultural, nature conservation and thicket restoration practices. Ecosystem restoration and conservation are land use options to increase biodiversity and the provision of a wide range of ESs (Benayas et al., 2009), whereas agricultural land use targets food production. These land uses can be managed with varying intensity, depending on management practices. Land management refers to human activities that affect land cover directly or indirectly (van Oudenhoven et al., 2012). Land management affects also vegetation, which can degrade as a consequence of intensive use or destructive land management (Reyers et al., 2009).

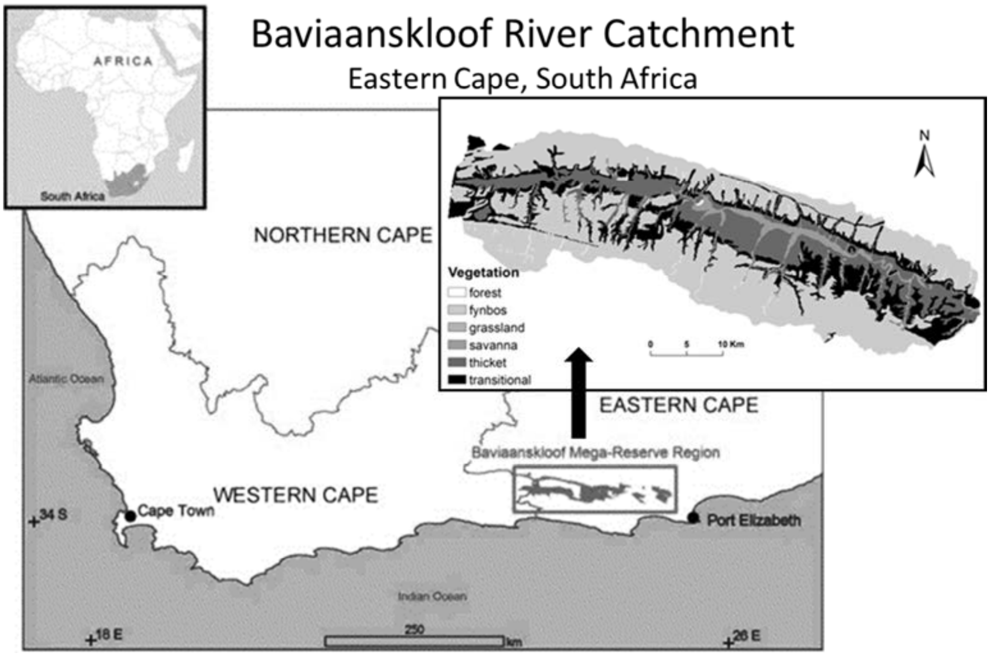

Figure 4.1: Location and vegetation of the Baviaanskloof Catchment, Eastern Cape, South Africa. (Source: map adopted from Crane (2006), vegetation map insert from Euston-Brown (2006))

Prior to this work, future land management of the Baviaanskloof Catchment was explored through stakeholder consultation. We build upon this, and apply scenarios to compare three alternative land management options that reflect stakeholders' preferences. Our study visualizes the spatial distribution of ESs, evaluates land management scenarios against targets set for these ES, and verifies whether the land management scenario preferred by stakeholders is also the most optimal in terms of ESs provision. Therefore, the results may help to strengthen local decision-making regarding the future management of the area.

4.2 Methods

4.2.1 Study area: the Baviaanskloof Catchement

Geography

The Baviaanskloof Catchment (ca. 123 000 ha) is located in Eastern Cape, in South Africa (Figure 4.1). The semi-arid catchment receives low and erratic precipitation in two annual rainfall peaks. Water is scarce and the recurring droughts are followed by flood events (Jansen, 2008). The Baviaanskloof River runs west to east between two parallel mountain ranges. It feeds the Kouga Dam and supplies water to downstream cities, including Port Elizabeth (van Eck et al., 2010). An unpaved road along the river provides access to the area. The catchment is home to seven of South Africa's eight biomes (Fynbos, Subtropical Thicket, Nama-karroo, Succulent Karoo, Grassland, Savanna and Forest), and is part of one of the Earth's biodiversity hotspots, the Cape Floral Kingdom (Boshoff, 2005; Crane, 2006). Savanna and grassland vegetation cover the valley-bottom and thicket shrubland and transitional vegetation cover the lower slopes (Figure 4.1). The catchment has a high diversity of Albany subtropical thicket dominated by the succulent *Portulacaria afra* ('Spekboom') (Boshoff, 2005). The montane vegetation is composed of fynbos, evergreen small-leafed shrub vegetation (Figure 4.1). This vegetation is (nearly) pristine. Most of the fynbos and parts of the thicket and grassland are protected under the Baviaanskloof Nature Reserve (van Eck et al., 2010). The catchment is home to protected (endemic) animal species (e.g. Cape mountain zebra, Black rhino, Cape leopard) (Boshoff, 2005). This highly diverse catchment is facing pressures of land conversion and degradation. On historically overgrazed areas vegetation cover and species diversity are degraded, soil is eroded and carbon stocks, and soil and water quality have declined (Lechmere-Oertel et al., 2005b; Mills et al., 2005). Vegetation, particularly thicket, is most degraded in the valley-bottom and on the lower slopes. Conservation interests emphasize sustainable utilisation of biodiversity and thicket restoration, since the area became an UNESCO World Heritage Site (2004) (van Eck et al., 2010). Governmental land management programmes and some local stakeholders aim to facilitate thicket restoration and livelihood improvement (van Eck et al., 2010).

Stakeholders

Stakeholders include local communities, farmers, non-governmental (e.g. Living Lands[1]) and governmental organizations (Eastern Cape Parks and Tourism Agency[2]), and scientists. About 62% of the area belongs to the government and 36% of the area belongs to a few large-scale farmers. Local communities share the remaining land (Powell and Mander, 2009). Governmental lands form the Baviaanskloof Nature Reserve, managed by the Eastern Cape Parks and Tourism Agency. The reserve is located on the higher slopes and mountaintops. Farmed lands are located on the valley-bottom and lower slopes. In these areas, vegetation is mostly degraded and is partly converted to cropland. Farmers' main income is derived from animal and crop production and from tourism (Crane, 2006). Local communities live in three villages and share small patches of communal lands in the valley-bottom. They depend highly on local natural resources (wild food, fuel wood, medicinal plants, construction material etc.), but both their resource access and income sources are

1 http://www.earthcollective.net/livinglands/, Accessed last November 20th 2013
2 http://www.ectourism.co.za/, Accessed last November 20th 2013

limited. About 95% of the local households extract or collect natural resources, even if it is mostly restricted or prohibited (Rhodes University Consortium, 2007). The unemployment rate is high and many inhabitants obtain social security grants (Crane, 2006). The number of permanent residents is estimated as 1000 people (Crane, 2006) in 463 households (CSIR Satellite Application Centre, 2010).

Land management

The main land uses in the Baviaanskloof catchment are agriculture, nature conservation, and thicket restoration. A part of the land is abandoned and not managed. The intensity of land use is related to crop choice, irrigation, animal choice, animal density and touristic infrastructure. Farmers set up hiking trails and tourist accommodations on their private lands to improve tourism. Agriculture, land abandonment and thicket restoration occur on farmers' private land. Management aimed at nature conservation occurs on all governmental lands and on some private lands.

Agriculture includes crop, livestock and game farming. Crops vary from farming maize as an annual crop in intensively used irrigated fields to perennial crops in non-irrigated orchards (olives, nuts) (Jansen, 2008). Livestock grazing is conventional with goat, sheep, cattle and ostrich production in fenced areas. Game farming is the raising of indigenous wildlife species, such as kudu (*Tragelaphus strepsiceros*), for tourism, sale or hunting.

Unmanaged lands are degraded private lands formerly used for agriculture, but not farmed any more.

Most formal nature conservation takes place in the Baviaanskloof Nature Reserve. Herbivores (Cape mountain zebra, Black rhino and Buffalo) were reintroduced in the reserve as part of conservation management (Powell and Mander, 2009). The (illegal) extraction of wood and other plant materials is a pressure to conservation (Rhodes University Consortium, 2007). Conservation on private land means adopting wildlife-friendly management and removing fences for economic incentives. A voluntary agreement between Eastern Cape Parks and Tourism Agency and farmers can assure this formally (Crane, 2006).

Thicket restoration is a transitional land use on farmers' land. It ideally involves a shift from a degraded, abandoned or low grazing capacity, state to a nearly pristine state. Restoration is done by re-planting the pioneer *P. afra* (van Eck et al., 2010). This creates a monoculture first, but stimulates ecosystem's restoration, carbon sequestration, species diversity, soil fertility, erosion prevention and water quality on a long term (Lechmere-Oertel et al., 2005a; Mills and Cowling, 2006).

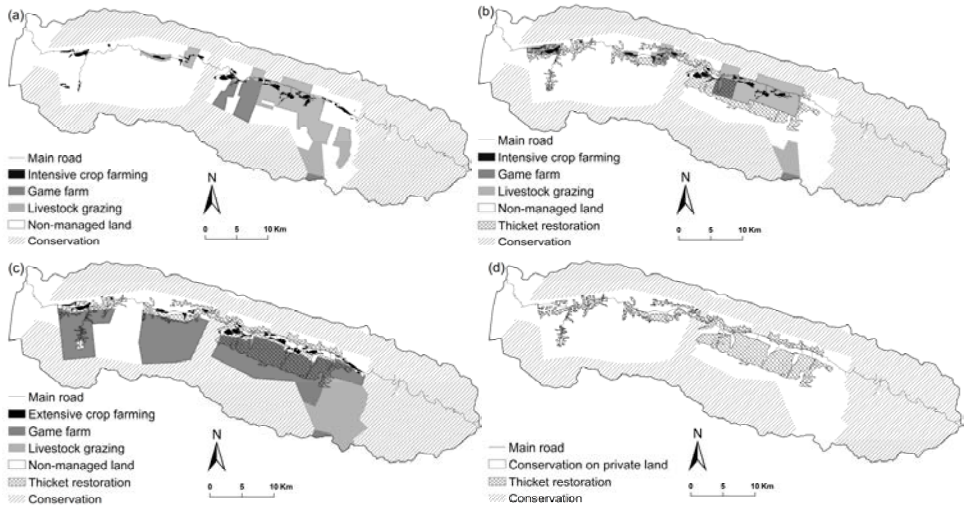

Figure 4.2: Land management under current situation (a), and for future scenarios of Diversity of Farming (b), Living with Nature (c) and Room for Nature (d)

Currently, the area is dominated by livestock, game and annual crop farming, unmanaged land and formal nature conservation. The other land management types are marginal, but may become important in the future (Figure 4.2).

4.2.2 Land management mapping and scenario development

Prior to this study, stakeholders explored future land use and management possibilities. The Living Lands organization facilitated interviews and workshops with the farmers, representatives of the Eastern Cape Parks and Tourism Agency and scientists (Stokhof de Jong, 2013). These stakeholders developed three land management scenarios, the Diversity of Farming (DoF), Living with Nature (LwN) and Room for Nature (RfN), with a vision for the year 2040. The scenarios were restricted to farmers' lands and no changes were expected on governmental lands.

The DoF scenario reflects the farmers' preferences for agriculture and related tourism. It is characterized by livestock, game and annual crop farming and partial thicket restoration. Large parts of the land remain unmanaged. The LwN scenario is a compromise between agriculture, restoration and nature conservation. It is dominated by game and perennial crop farming. Thicket is completely restored. Livestock grazing and unmanaged lands are reduced. The RfN scenario reflects the Eastern Cape Parks and Tourism Agency's preference for extended restoration and nature conservation. Under this scenario, nature conservation dominates, thicket is completely restored and agricultural practices are abandoned. Hiking trails and tourist accommodation expand the least under DoF and to the most under RfN scenarios. In a workshop organized by Living Lands (November 2011) stakeholders chose LwN as their preferred way forward as a reasonable compromise between different interests.

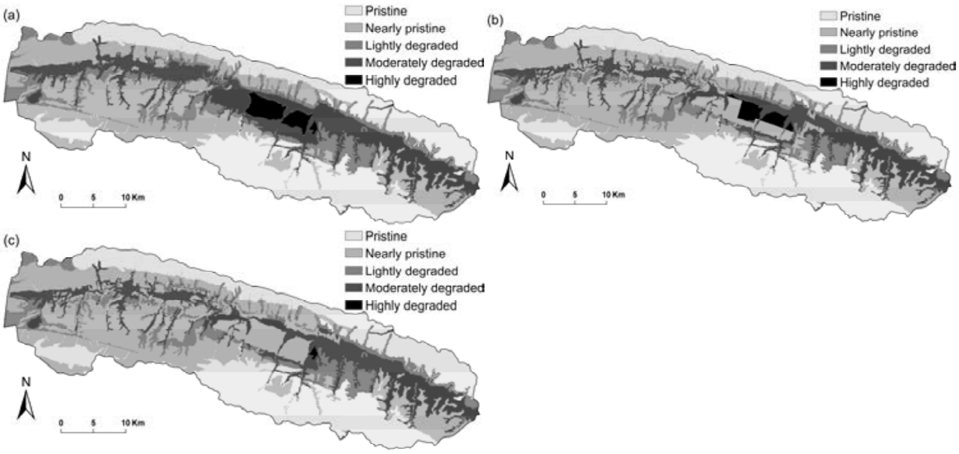

Figure 4.3: Vegetation degradation level under current situation (a), the future scenarios of Diversity of Farming (b), Living with Nature and Room for Nature (c) (Source: Euston-Brown (2006))

The three scenarios and corresponding maps were the starting point for our land management mapping and scenario analysis. We combined the land management maps covering the farmers' lands with a map of formal conservation areas (SANBI, 2003) in order to cover the whole Baviaanskloof Catchment (Figure 4.2). Land management has also consequences for the vegetation and degradation level of the area. Original vegetation type is replaced by crops on cultivated areas and is regained when crop cultivation is abandoned. Vegetation degradation varies from pristine and nearly pristine to lightly, moderately and severely degraded states (Euston-Brown, 2006). Thicket restoration activities reduce the highly degraded area marginally in the DoF scenario and substantially in the other two scenarios (Figure 4.3). We assumed restored thicket to have the ecological and hydrological characteristics of nearly pristine thicket and store as much carbon as pristine thicket (Mills and Cowling, 2006) (Figure 4.3). The land management maps (Figure 4.2), vegetation type (Figure 4.1) and vegetation degradation (Figure 4.3) maps were used as main inputs to calculate ESs.

4.2.3 Mapping and modelling ecosystem services (potentials)

A range of ESs were selected, considering current and future demand for ESs, relevance for stakeholders, data availability, and mapping and modelling possibilities. We studied forage production, fuel wood provision, water supply (provisioning services), erosion prevention, carbon sequestration (regulating services), ecotourism (cultural service), and biodiversity (habitat service).

Table 4.1: Overview of data used for ecosystem service mapping and modelling. SAACA stands for South African Atlas of Climatology and Agrohydrology

Data description	Unit	Format	Source	Ecosystem Service used for
Vegetation	Classes	GIS polygon shape	PRESENCE GIS database (Euston-Brown, 2006)	water supply, erosion prevention, carbon sequestration, biodiversity
Land use/management	Classes	GIS polygon shape	PRESENCE GIS database (Stokhof de Jong, 2013)	all
Vegetation degradation	Classes	GIS polygon shape	PRESENCE GIS database (Euston-Brown, 2006)	all, except for ecotourism
Veld condition as a percentage of benchmark	%	Table	Department of Agriculture, Environmental Affairs & Rural Development, Province of Kwazulu-Natal (Veld management, http://agriculture.kzntl.gov.za/)	forage production
Mean annual precipitation	mm	GIS raster	SAACA (Lynch and Schulze, 2007)	forage production, water supply
Reference evapotranspiration	mm	GIS raster	SAACA (Schulze and Maharaj, 2007)	water supply
Soil depth	mm	GIS polygon shape	SAACA (Schulze and Horan, 2007)	water supply
Soil available water fraction	mm/mm	GIS polygon shape	SAACA (Schulze and Horan, 2007)	water supply
Watershed and sub-watershed	---	GIS polygon shape	SAACA (Schulze et al., 2007)	water supply, erosion prevention
Plant evapotranspiration coefficient	%	Table	University of KwaZulu-Natal (ACRU model parameters)	water supply, erosion
Average root depth	mm	Table	InVEST 2.0 User's Guide, p. 242. (Tallis et al., 2011)	water supply
Rainfall intensity/erosivity	MJ*mm / (ha*h*year)	GIS polygon shape	SAACA (Schulze, 2007)	erosion prevention
Soil erodibility	T*ha*h/ (ha*MJ*mm)	GIS polygon shape	SAACA (Schulze and Horan, 2007)	erosion prevention
Digital Elevation Model	m	GIS raster	http://www.ngi.gov.za/	erosion prevention
Baviaanskloof Nature Reserve	---	GIS polygon shape	South African National Biodiversity Institute (SANBI, 2003)	fuel wood provision, ecotourism
Road	---	GIS polyline shape	http://www.ngi.gov.za/	ecotourism, biodiversity
Walking routes	---	GIS polyline shape	PRESENCE GIS database	ecotourism
Settlement	---	GIS polygon shape	http://www.ngi.gov.za/	ecotourism, biodiversity
Accommodation sites	---	GIS point	PRESENCE GIS database	ecotourism

We used simple (proxy) models to quantify and map the potential or/and actual ES for the current situation and for the three future land management scenarios, expected to be put in place by the year 2040. Applied mapping and modelling methods ranged from single indicator mapping (carbon sequestration) to models linking indicators to environmental variables (fuel wood provision, ecotourism and biodiversity) and biophysical production functions (forage production, water supply and erosion prevention). A field visit and informal consultations with stakeholders supported the data collection and ES estimation. The analysis was carried out in ArcGIS 10 environment (ESRI, 2011) .

Forage production

Forage production is defined as the provision of forage on areas used for livestock or game production. We used the grazing capacity model of Danckwerts (Danckwerts, 1989; Schmidt et al., 1995) to estimate forage production for a potential number of livestock (Livestock Unit(LSU)/ha). The model was developed for herbaceous sweet grassland on Eastern Cape False Thornveld to estimate grazing capacity and it also gives a first order approximation of grazing capacity for thicket (Schmidt et al., 1995). The model inputs are veld condition and mean annual rainfall (Table 4.1). Veld condition refers to the state of natural vegetation in relation to its long-term potential for livestock production (Tainton et al., 1999). We related veld condition scores (0-100, %) to degradation levels using pre-established veld condition categories (Table 4.2). Forage production for an actual number of livestock was taken as two third of the grazing capacity (i.e. forage production for a potential number of livestock).

Table 4.2: Reclassification of degradation levels (Figure 4.3) to veld condition scores based on pre-established veld condition categories and corresponding veld condition scores (for data source see Table 4.1).

Pre-established veld condition categories and corresponding veld condition scores (%)		**Reclassification of degradation levels to veld condition scores**	
Critical	0-25%	Severely degraded	10%
Poor	25-50%	Moderately degraded	30%
		Lightly degraded	50%
Reasonable	50-75%	Nearly pristine or restored	70%
Good	75-100%	Pristine	90%

Fuel wood provision

In the Baviaanskloof Catchment, 90% of the local households collect and use fuel wood (Rhodes University Consortium, 2007). Contrary to livestock and game, fuel wood is not traded. Sweet Thorn (*Acacia karroo)* is one of the popular fuel wood sources in Eastern Cape (Pote et al., 2006) and in the Baviaanskloof (Rhodes University Consortium, 2007). *A. karroo* is a pioneer and dominant species of the valley-bottom savanna and thicket (Puttick et al., 2011).

Fuel wood provision is defined as the annual biomass production (yield) of *A. karroo* (kg/ha) available for collection. We adopted methods of Masera et al. (2003) to local conditions and mapped fuel wood provision as the annual *A. karroo* biomass production corrected for vegetation and land use types, topography and legal accessibility. People prefer wood stems with a certain size for collection (Pote et al., 2006). Annual yield is a product of the biomass (kg/ha) of wood stems with preferred size and the annual increment. As no local data were available, measured biomass data were adopted from a nearby thicket-dominated region, from Pote at al. (2006). Annual increment was taken as 4% after Banks et al. (1996).

A. karroo can dominate the overgrazed and degraded forms of thicket (Puttick et al., 2011). Therefore, we assigned higher *A. karroo* biomass stock to degraded than to non-degraded lands (pristine = 400 kg/ha, nearly pristine = 700, lightly degraded = 1000, moderately degraded = 1200, severely degraded = 1600). *A. karroo* growth decreases on slopes (Pote et al., 2006). Therefore, we related negatively the biomass stock range (400-1600 kg/ha) to the slope. We averaged the biomass stock values adjusted to degradation and slope, for each cell. On formal nature conservation, no wood collection is allowed and *A. karroo* does not grow on cultivated areas. These areas were therefore excluded.

Water supply

Water supplied by the Baviaanskloof Catchment is important to meet the growing downstream irrigation, domestic and industrial water needs (van Eck et al., 2010). A payment system for water-related ESs is a considered option for the larger region (Mander et al., 2010b). Water supply was estimated using the long-term average annual water yield (m^3) as an indicator. We used the InVEST tool (Kareiva et al., 2011) to quantify and map water yield using vegetation and hydrological data. Water yield is calculated as the difference between precipitation and actual evapotranspiration as an annual average. Degradation and grazing reduces biomass and hence evapotranspiration and increases runoff (Asner et al., 2004). Restoration has an opposite effect. Data on vegetation characteristics (root depth, plant evaporation coefficient) were obtained from a hydrological modelling database (ACRU) updated for the Baviaanskloof for vegetation types and degradation levels (Mander et al., 2010b). We assumed moderate grazing, under which 40% of the forage biomass is grazed (Holechek et al., 1999; Palmer et al., 2006). A linear relationship between grazed biomass and the evapotranspiration coefficient was assumed. Due to the lack of quantitative information, we assumed no difference between livestock farming and game farming. We assigned a higher evapotranspiration coefficient and a smaller root depth value to irrigated maize than to perennial orchards. Root depth was interpolated from biome-specific data of Tallis et al. (2011) (for fynbos, grassland and savannah 2600 mm, for transitional and thicket 5100 mm, for forest 7000 mm was taken). For detailed modelling methodology see Tallis et al. (2011) and for input data on vegetation and hydrology see Table 4.1.

Erosion prevention

Erosion prevention provided by the Baviaanskloof Catchment is important to reduce soil and vegetation loss and downstream sedimentation (Mander et al., 2010b). Erosion prevention is defined as the annual sediment retention by vegetation (t/ha). We used the InVEST tool (Kareiva et al., 2011) to quantify and map sediment retention. The model is built upon the Universal Soil Loss Equation, and it calculates sediment retention as the difference between the soil loss under current vegetation cover and an estimate for bare soil. For detailed modelling methodology see Tallis et al. (2011) and for input data on rainfall, soil, topography and vegetation see Table 4.1. Degradation and grazing reduce soil retention and restoration has an opposite effect (Asner et al., 2004). Data on the effect of the crop and management practices for different vegetation types and degradation levels on erosion rates were obtained from the ACRU model database adapted for the Baviaanskloof (Mander et al., 2010b). No additional management practices were considered to reduce erosion. Parameters for sediment retention factor were derived from Tallis et al. (2011) ((nearly) pristine natural vegetation=100, lightly and moderately degraded=50, severely degraded=10, cultivated =60). Furthermore, similarly to the evaporation coefficient, we assumed a linear relationship between the grazed biomass and the sediment retention capacity of vegetation. Thus, we considered a 40 % decrease in sediment retention capacity under grazing.

Carbon sequestration

The Baviaanskloof represents a significant potential for carbon sequestration through thicket restoration (Mills and Cowling, 2006). Carbon sequestration considered here is that sequestered by *P. afra*-dominated thicket. Although other local vegetation types also store carbon (Mills et al., 2012; Reyers et al., 2009), *P. afra*-dominated thicket is the main vegetation type, which undergoes restoration, and for which local data are available. Similarly to Egoh et al. (2010) and Reyers et al. (2009) we calculated carbon storage in the vegetation, including in above and belowground biomass, soil and litter. The estimated change in carbon stocks between the current and the future situation was considered as the carbon sequestration or depletion. A thicket-specific degradation map was used (Powell et al., 2011) and locally measured carbon stock (Powell, 2009) was extrapolated and mapped for moderately-severely degraded (30.50±2.05 t/ha), and (nearly) pristine-lightly degraded (87.73±6.51 t/ha) areas. Total carbon storage is the product of the mean carbon storage corresponding to the degradation level and the area, summed for the catchment. Completely restored thicket stores as much carbon as pristine thicket (Mills and Cowling, 2006). Degraded thicket that does not undergo restoration was assumed not to sequester any net additional carbon (Lechmere-Oertel et al., 2005a).

Ecotourism

The Baviaanskloof Nature Reserve receives about 45 600 and the rest of the catchment receives a further 10 000-12 000 tourists annually (Powell and Mander, 2009). The area is popular for wildlife watching and scenery, watched from the road. Attractive scenery, high accessibility and high diversity of wildlife are among the strongest motives for tourists to visit an area in South Africa (Lindsey et al., 2007; Milton et al., 2003; Reyers et al., 2009). We measured ecotourism, by combining the visibility of the scenery, accessibility and wildlife diversity in an 'ecotouristic suitability' index (0-100, %). Scenery was mapped as areas visible from roads, hiking trails and tourist accommodation sites by creating a viewshed as described by Reyers et al. (2009) and O'Farrell et al. (2010). The highest value (90) was attributed to areas visible both from roads/hiking trails and accommodation sites, a medium value (50) was attributed to areas only visible from roads/hiking trails and the lowest value (10) was attributed to non-visible areas. Accessibility was calculated by taking a buffer of 1,000 m along the road and settlements. Within this buffer, a linear increase in accessibility with decreasing distance to the road or settlements was calculated after Chan et al. (2006). The number of wild animals and hence the suitability of ecotourism increases with habitat protection (Lindsey et al., 2007; Milton et al., 2003). Therefore, we increased the value of ecotouristic suitability slightly more on conservation areas (value * 1.5) than on game farms (value * 1.2). Thus, ecotouristic suitability was calculated by combining the scenery map with the accessibility map and weighing the results with the wildlife value map.

Biodiversity

The Baviaanskloof has a relatively high numbers of plant and animal species. Although land conversion, livestock grazing and vegetation degradation have large negative impact on biodiversity (Biggs et al., 2008; Lechmere-Oertel et al., 2005b; Scholes and Biggs, 2005), local data on species occurrence and abundance are rarely available outside the protected areas (e.g. Subtropical Thicket Ecosystem Project[3]). Therefore we used the GLOBIO3 global biodiversity modelling framework (Alkemade et al., 2009) to quantify the changes biodiversity as a result of changing land management. GLOBIO3 uses the Mean Species Abundance index (MSA, 0-1, remaining original species abundance relative to pristine ecosystem) in relation to multiple human pressures (Alkemade et al (2012; 2009) and http://www.globio.info/). We calculated impacts of four land management-related pressures: 1) land cover/use change 2) proximity of roads, 3) proximity of croplands and villages, and 4) fragmentation. Land cover/use change effects were based on Alkemade et al (2012; 2009) and were extended to vegetation degradation levels (Table 4.3). Background information of road, agriculture and urban impacts is described in Benítez-López et al (2010) and distance impacts were adjusted to local conditions (for the impact zone of roads, annual crop farming and villages 1 000 m was taken and of perennial crop farming 500 m was taken). Fragmentation effect is based on the minimum area requirement of animal species (Alkemade et al., 2009). The biodiversity map was created by overlaying (multiplying values) the four pressure maps.

[3] http://www.bgis.sanbi.org/STEP/project.asp, Accessed last November 12th 2011

Table 4.3: Mean Species Abundance (MSA, 0-1) estimated for land use types and intensities and degradation levels in the Baviaanskloof Catchment. Values are based on MSA values for land use classes from Alkemade et al. (2012; 2009). For unmanaged lands, vegetation types and corresponding degradation levels were used.

MSA values Baviaanskloof	**Land use intensities according to GLOBIO3 (MSA value)**	**Vegetation and land use type**	**Vegetation degradation level (according to Euston-Brown (2006))**
1.0	primary forest (1.0)	Forest	(nearly) pristine
1.0	primary grass- or scrublands (1.0)	Fynbos	(nearly) pristine - lightly degraded
1.0	natural rangeland (1.0)	Grassland	(nearly) pristine
1.0	primary grass- or scrublands (1.0)	Savanna	lightly degraded
0.7	ungrazed abandoned rangelands (0.7)		moderately degraded
1.0	natural rangeland (1.0)	Thicket	(nearly) pristine
0.7	ungrazed abandoned rangelands (0.7)		moderately-severely degraded
1.0	primary grass- or scrublands (1.0)	Transitional	(nearly) pristine - lightly degraded
0.7			moderately degraded
0.1	intensive irrigated agriculture (0.1)	Irrigated agriculture	---
0.6	moderately used rangeland (0.6)	Game farm	(nearly) pristine - lightly degraded
0.5	intensively used rangeland (0.5)		moderately/severely degraded
0.6	moderately used rangeland (0.6)	Livestock grazing	(nearly) pristine
0.5	intensively used rangeland (0.5)		moderately/severely degraded
0.3	low input agriculture (0.3)	Non-irrigated orchard	---
1.0	natural rangeland (1.0)	Restored thicket	(nearly) pristine
1.0	natural rangeland (1.0)	Private conservation land	Restored
0.7	ungrazed abandoned rangelands (0.7)		non-restored

4.2.4 Ecosystem services quantification and scenario analysis

First, ESs were quantified for land management practices. We calculated mean per ha ES value for each land management type and corresponding vegetation degradation level. For this, current land management, vegetation degradation and derived ES maps were overlaid. Since perennial crop farming, thicket restoration and conservation on private are not present under the current situation, for these land management types the LwN and RfN scenario maps and corresponding ESs maps were used.

Next, we compared future scenarios to current management in terms of ESs provision in the whole catchment. Each ES was aggregated for the catchment, except for ecotourism and biodiversity indices. For these mean values were calculated.

Finally, ESs were compared to management targets to evaluate the scenarios and stakeholders' land management choice. We used the catchment management plan compiled by Living Lands (PRESENCE, unpublished report 2011) to identify target for each ES (Table 4.4). If an ES target is met, the scenario is considered to provide an ES sufficiently. For fuel wood provision no target was found therefore we used local demand. It is a product of household numbers (463, (CSIR Satellite Application Centre, 2010)), percentage of households collecting fuel wood (90%) and annual fuel wood consumption per household (5 362 kg, (Rhodes University Consortium, 2007)).

Table 4.4: Management targets based on the catchment management plan (PRESENCE, unpublished report 2011) used for ES evaluation under scenarios.

Ecosystem service	**Management target to evaluate scenarios (year 2040)**
Forage production	Sustainable agriculture translated to min. 500 LSU on (nearly) pristine/restored grazing land*
Fuel wood provision	Meeting local fuel wood demand: About 2 200t/year
Water supply	Increased water supply compared to current situation
Erosion prevention	Erosion reduction compared to current situation
Carbon seq.	Carbon sequestration through 'Spekboom' planting
Ecotourism	Increased ecotourism (facilities) compared to current situation
Biodiversity	Increased biodiversity compared to current situation

* meaning minimal 10 000 ha (nearly) pristine or restored non-fragmented grazing land with a grazing capacity of min. 0.05 LSU/ha (GIS Unit Department of Agriculture, 2004)

4.3 Results

4.3.1 Distribution of ecosystem services under current land management

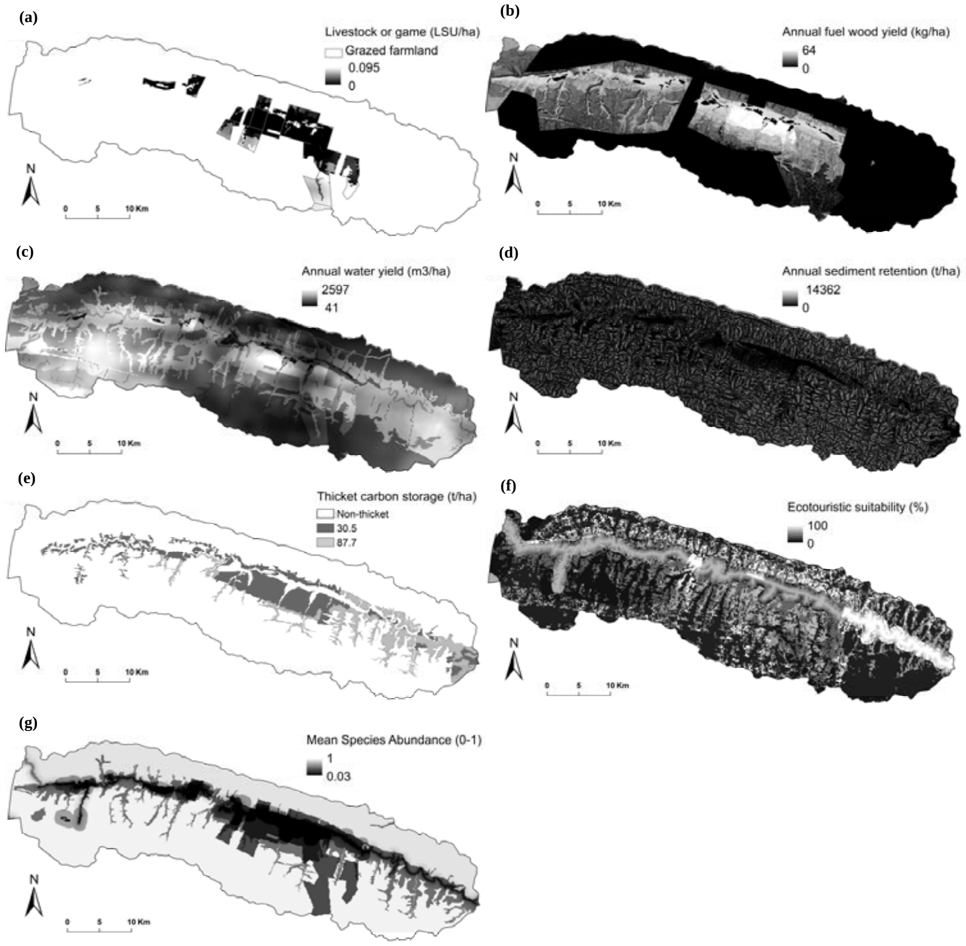

Figure 4.4: Ecosystem services (potentials) under current land management: forage production (a), fuel wood provision (b), water supply (c), erosion prevention (d), thicket carbon sequestration (e), ecotourism (f), and biodiversity (g).

Results show that, in general, valley-bottom and the lower slopes (i.e. farmers' lands) provide ESs at higher rate compared to the mountaintops (i.e. governmental lands), except for biodiversity (Figure 4.4). Slopes provide more forage than the valley-bottom, for fuel wood it is the opposite. The potential for ecotourism and carbon storage was estimated to be the highest in the valley, on the mountain slopes, and in conservation areas. The whole catchment supplies water and retain sediment erosion. Most water is provided at higher altitudes with greatest rainfall and on grazed lands. The pattern of sediment retention follows the topography. Modelled biodiversity intactness is lowest on degraded or annual crop farming lands at lower altitudes; and highest on (nearly) pristine conservation areas at higher altitudes.

4.3.2 Ecosystem services quantified for land management types

ES values vary across land management types and vegetation degradation levels. Historically overgrazed (i.e. moderately and severely degraded) areas may provide approximately two and half times more water and 60% more fuel wood; while retaining 40% less sediment and having 38% lower biodiversity, than pristine lands (Table 4.5). In historically overgrazed areas, where vegetation is degraded, the provision of fuel wood, water supply and ecotourism is higher, while the provision of other ESs is lower than in pristine areas. This is the direct result of the vegetation degradation or the indirect result of the spatial distribution of degraded lands. There is a trade-off between forage production and fuel wood provision, as the *A. karroo*-dominated thicket supports hardly any livestock. Because of their vicinity to roads and touristic infrastructure, the historically overgrazed lands have a higher potential for ecotourism (suitability 0.42 on average) than the remote pristine areas (suitability 0.24 on average).

Grazed lands were predicted to supply more water, than cultivated and non-managed areas. This is a function of lower rainfall at the valley-bottom and lower vegetation density and consequently a higher runoff in grazed areas. Crop cultivation areas provide little water and retain little soil, since they are located in the valley-bottom. Ecotourism and biodiversity appear to be higher in the perennial crop farming than under annual crop farming areas.

Private conservation lands, restored thicket and perennial crop farming areas were predicted to be the best for tourism. This is due to the increased numbers of wild animals, tourist facilities or high accessibility. Non-grazed, restored, or (nearly) pristine lands provide highest biodiversity and forage, prevent most erosion and store most carbon. Formal and private nature conservation areas and unmanaged lands have the highest MSA (0.89 on average), and crop farms have lowest MSA value (0.17 on average), followed by grazed lands (0.53 on average). Unmanaged, degraded lands provide no forage, but provide fuel wood, water and biodiversity, reduce erosion, and are potential areas for ecotourism. ESs do not depend only on land management type and degradation, but also on other biophysical, geographical and management aspects (e.g. topography and infrastructure). This is reflected in the results.

Table 4.5: Mean ecosystem service values for each land management type and corresponding vegetation degradation rates.

Land management type	Ecosystem Services	Forage production (mean LSU/ha)	Fuel wood provision (mean annual yield, kg/ha)	Water supply (mean annual yield, m3/ha)	Erosion prevention (mean sediment retention, t/ha)	Carbon Sequestration (mean carbon stored in thicket, t/ha)	Ecotourism (mean suitability %)	Biodiversity (mean MSA)
Formal nature conservation	Pristine	---	---	581	320		25	0.96
	Nearly pristine	---	---	1023	278	87.7	27	0.95
	Lightly degr.	---	---	1169	286		36	0.93
	Moderately degr.	---	---	1323	284	30.5	45	0.65
Livestock grazing	Pristine	0.07	35	536	336		24	0.61
	Nearly pristine	0.03	42	682	297	87.7	31	0.60
	Lightly degr.	0.01	46	1770	341		33	0.58
	Moderately degr.	0.00	55	943	149	30.5	39	0.37
	Severely degr.	0.00	62	1432	97		36	0.42
Game farming	Pristine	0.07	35	758	384		20	0.60
	Nearly pristine	0.04	42	1035	243	87.7	30	0.61
	Lightly degr.	0.01	48	1326	177		39	0.61
	Moderately degr.	0.00	53	1263	210	30.5	35	0.47
	Severely degr.	0.00	62	1605	119		40	0.46
Unmanaged lands	Pristine	---	35	401	323		24	0.96
	Nearly pristine	---	42	846	221	87.7	21	0.93
	Lightly degr.	---	48	1255	220		27	0.91
	Moderately degr.	---	52	1122	242	30.5	33	0.61
Crop farming	Annual irrigated maize	---	---	346	48	---	43	0.13
	Perennial orchard	---	---	334	34	---	56	0.20

Land management type	Ecosystem Services	Forage production (mean LSU/ha)	Fuel wood provision (mean annual yield, kg/ha)	Water supply (mean annual yield, m3/ha)	Erosion prevention (mean sediment retention, t/ha)	Carbon Sequestration (mean carbon stored in thicket, t/ha)	Ecotourism (mean suitability %)	Biodiversity (mean MSA)
Thicket restoration	Grazed	0.04	45	1372	212	87.7	77	0.66
	Non-grazed	---	43	886	203		69	0.74
Private conservation	Pristine	---	35	371	323	87.7	48	0.97
	Nearly pristine	---	42	1110	234		52	0.94
	Lightly degr.	---	48	1180	233		57	0.95
	Moderately degr.	---	53	1014	195	30.5	64	0.64

4.3.3 Scenario analysis and evaluation

In all scenarios, model results indicate that additional carbon is sequestered and slightly less fuel wood is provided (Table 4.6), as a result of the projected thicket restoration. Most forage and water is provided under LwN scenario due to a decrease of unmanaged land and the increase in game farming. These large grazed areas likely retain less water and impair biodiversity. The LwN scenario supports more provisioning ESs, than the other scenarios. Water supply slightly decreases under the DoF and RfN scenarios (Table 4.6). This is because the thicket restoration and reduced grazing decreases runoff. Ecotourism and biodiversity are provided best under the RfN scenario. Under RfN scenario, no forage is produced, but slightly more sediment is retained and biodiversity and the potential for ecotourism highly increase (Table 4.6). This is the result of increased vegetation restoration and conservation, expansion of hiking trail network and abandonment of agriculture. The vast areas of unmanaged land under DoF and the conservation on private land under RfN are beneficial for biodiversity.

Table 4.6: Ecosystem services in the Baviaanskloof Catchment, for current situation and three scenarios. Results of scenario evaluation: dark grey = meets target; light grey = does not meet target. For targets, see Table 4.4.

Ecosystem service	**Management targets to evaluate scenarios**	**Land management**			
		Current	**DoF**	**LwN**	**RfN**
Forage production (Total Livestock Unit, LSU)	Is management sustainable (i.e. min. 500 LSU on (nearly) pristine or restored grazing land)?	230	211	1345	0
Fuel wood provision (Total annual yield t)	Is local fuel wood demand met?	2400	2300	2200	2300
Water supply (Total annual yield, Mill m³)	Does water supply increase?	45.9	45.3	48.8	44.0
Erosion prevention (Annual sediment retained, Mill t)	Does erosion decrease?	362	363	362	363
Carbon sequestration (Storage, 1 000 t)	Does carbon increase (i.e. addition carbon is sequestered)?	1246	1616	1784	1784
Ecotourism (Mean ecotouristic suitability, %)	Does ecotourism increase?	46	46	49	66
Biodiversity (Mean MSA, 0-1)	Does biodiversity increase?	0.84	0.86	0.83	0.90

The DoF scenario provides fuel wood and three other non-provisioning ESs sufficiently. In general, forage production appears to be the most sensitive to the land management changes, followed by carbon sequestration and ecotourism. The rest of the ESs show minimal relative change. The relative change in fuel wood provision, water supply and especially in sediment retention is small under all scenarios regardless of meeting the targets.

4.4 Discussion

4.4.1 Limitations and opportunities of modelling and validation of results

This study demonstrates how ESs can be assessed spatially in a data-scarce area, by using relatively simple methods for mapping and modelling. These methods build upon the available studies from other parts of South Africa, and of widely used generic models such as the InVEST and the GLOBIO3. The resulting maps and models could be used to evaluate different scenarios for the area and to verify choices of stakeholders.

The methodology used implies various sources of uncertainty, as the data from the area were scarce. We were however able to verify some outcomes by comparing them with some overall estimates. In the following, we describe these verifications for each ES.

In contrast to other grazing capacity models applied in South Africa (e.g. Moore and Odendaal (1987)), the model we used captures the effect of historical grazing. Therefore, it gives an estimate closer to the long-term forage production (or grazing) capacity than the methods based on actual biomass production (Schmidt et al., 1995). Vegetation degradation-induced decrease in forage production has been demonstrated also by others (Reyers et al., 2009). Our mean forage production estimate (0.03 LSU/ha) is close to, but is slightly lower than what the coarse provincial grazing capacity map shows (0.05 LSU/ha, (GIS Unit Department of Agriculture, 2004)). Differences may be due to our translation of vegetation degradation levels into prescribed veld condition scores or the spatial resolution of the analysis.

Using biomass growth makes the comparison between fuel wood production and consumption possible (Van Jaarsveld et al., 2005). Our estimated fuel wood provision range (0-64 kg/ha), based on the measured data of Pote et al. (2006), falls within the range (0-80 kg/ha) reported for Southern Africa (Van Jaarsveld et al., 2005). Results showed that fuel wood provision is below or hardly exceeds local demand under any of the management options. In practice people collect wood only at areas nearby the roads and villages, thus not all fuel wood provided is collected and used. These make the Baviaanskloof vulnerable to fuel wood shortages, similarly to the majority of South Africa (Van Jaarsveld et al., 2005).

Mapping carbon sequestration as the change in vegetation carbon storage is a widely-used method (Kareiva et al., 2011). Thicket carbon sequestration shows a variation through the literature. The annual sequestration rate for thicket restoration areas is 1.9 t/ha calculating with the 30 years scenario timeline and Powel's (2009) carbon stock data used in present study. Mills and Cowling (2006) calculated 4.2 and 2.4 t/ha/year carbon sequestration rate for nearby areas. These differences are due to the variation in planting density, environmental condition and restoration timeline, among others.

Forage production, carbon sequestration and fuel wood provision can be derived alternatively from net primary productivity. In our case, net primary productivity estimates derived from satellite images (MODIS) yielded coarse (1x1km) carbon sequestration results without differences along vegetation degradation and grazing gradients.

Water yield and erosion prevention were both estimated previously by the hydrological model ACRU[1] as part of an initial Payment for Water Services feasibility study (Mander et al., 2010a). In the ACRU model application land management was restricted to thicket restoration and ESs were not mapped, only quantified. As opposed to traditional hydrological models, the InVEST model is suitable for spatial analysis of multiple ESs, although it represents hydrologic process in a simplified way (Vigerstol and Aukema, 2011). Water yield and erosion prevention results of the InVEST model are more reliable at catchment level rather than at pixel level (Tallis et al., 2011). Our annual water yield estimate for the whole catchment (45.9 Million m^3) is very close to previous hydrological modelling results (45.7 million m^3 (Jansen, 2008) and 47.1 million m^3 (Mander et al., 2010a)). Because we did not consider the water supply infrastructure and the timing of water flows, the estimated water yield may differ from the realized water supply. The decrease in water yield and increase in sediment retention as a result of increased vegetation cover and thicket restoration is supported by the prior hydrological model results (Mander et al., 2010a) and other studies (Le Maitre et al., 2007; Reyers et al., 2009; van Luijk et al., 2013). Van Luijk et al. (2013) measured a slightly lower increase in water yield (two times), than the present study (two and half times), as a result of thicket degradation in the Baviaanskloof Catchment. The same authors measured much higher change in sediment retention, than the present study. The difference can be caused by the indicator use, our model parameterization of degradation and grazing effects, and the differences in methods (field sampling vs. modelling).

Combining indicators is a frequently used way to map the potential for ecotourism, as empirical and quantitative data on the spatial dynamics of tourists are normally absent

[1] http://dbnweb2.ukzn.ac.za/unp/beeh/acru/, Accessed last November 20th 2013

(c.f. Reyers et al. (2009), O'Farrell et al. (2010) and Petz and Van Oudenhoven (2012)). The Baviaanskloof Catchment is heavily visited area (Powell and Mander, 2009), but to our best knowledge no maps of the tourists visits exist. Although historically overgrazed lands have high touristic potential because of the infrastructure provided, the tourism potential of a pristine or restored landscape is greater than tourism potential of degraded landscape (Powell et al., 2009; Reyers et al., 2009). This is supported by our scenario results, as ecotourism has the greatest potential under the RfN scenario.

Prior to this study, GLOBIO3 was applied in a multiple ESs context at global scale and at regional scale only in Europe (Maes et al., 2012). In contrast to other biodiversity indexes that are based on actual observations, such as the biodiversity intactness index (Scholes and Biggs, 2005), the MSA index can be used to estimate relative land management impacts without requiring local empirical data on species numbers. Scholes and Biggs (2005) and Biggs et al. (2006) used biodiversity intactness index to map biodiversity in South Africa. Our mean biodiversity value (0.84) falls in the same range as these prior larger-scale studies indicate (biodiversity intactness index 0.77-0.88 (Scholes and Biggs, 2005) and 0.73-0.83 (Biggs et al., 2006) for the vegetation types also present in the Baviaanskloof). Our result about the decrease in MSA on degraded lands compared to pristine lands (38%) is close to field a measure conducted on a similar site in the Eastern Cape (35% decrease in species richness and 30% decrease in species diversity (Lechmere-Oertel et al., 2005b)). Our results also underline that Baviaanskloof Nature Reserve, which is a core are for nature conservation and biodiversity maintenance, has the highest biodiversity in the whole catchment.

The goal of mapping and modelling was to understand the spatial distribution of ESs and compare land management options. We emphasize that ESs should be viewed in combination, rather than as single and separate services. We believe that our ESs results are valid, because they were comparable to available estimates from other sources and methods; either applied locally or in similar regions.

4.4.2 Evaluation of land management choice

Thicket restoration decreases water supply on short term, but it enhances water regulation important for the long-term provision of water and other ESs (Reyers et al., 2009; van Luijk et al., 2013). Restoration is, therefore, important considering the growing water shortage in Southern Africa (Van Jaarsveld et al., 2005) and in the Baviaanskloof (Mander et al., 2010b). The full restoration of thicket ecosystem, its ecological processes and restoration of converted areas, however, may take much longer than the 30 years scenario timeline (Le Maitre et al., 2007; Mills and Cowling, 2006).

The state of ES depends not only on its provision, but also on human needs for this service (Paetzold et al., 2010). For example, the catchment supplies water for downstream communities (van Eck et al., 2010) hence the spatial dynamics of water provision and use can be very roughly quantified (Mander et al., 2010a). The comparison of land management scenarios and their evaluation of ESs against targets and local demand are important steps towards inclusion of ESs into local decision-making and planning. Meeting the targets alone, however, does not necessarily imply benefits for local inhabitants. For example, sustainable agriculture is a target of catchment management, but not necessarily of all the individual farmers, only if paired with incentives. Furthermore, the sufficient fuel wood provision is important for the local communities, but fuel wood collection is not allowed on conservation areas and is not supported by the catchment management plans. There is a general tension between the government and the local communities regarding the conservation and the management of the Baviaanskloof Catchment (Hough and Prozesky, 2010), also indicated by the exclusion of the local communities from the scenario development and choice.

The nature conservation and restoration-oriented RfN and the compromise (between agriculture, conservation and restoration) LwN scenarios meet slightly more management targets than the agriculture-oriented DoF scenario. The chosen LwN scenario falls short on fuel wood, erosion prevention and biodiversity. The applied mapping and modelling methods and the quantitative results presented in Table 4.5 can help to improve the choice of land management options and may help to define another, more optimal, scenario. For example, the partial transformation of game farms to private conservation with a voluntary agreement between the farmers and the Eastern Cape Parks and Tourism Agency could enhance biodiversity. Undertaking light grazing on the remaining game farms could reduce erosion. In addition, the interests of local communities and the ESs they depend could also be considered in order to prevent potential fuel wood shortages, illegal extractions and achieve successful conservation.

4.5 Conclusion

This study aimed at the evaluation of alternative land management options through quantifying and mapping multiple ESs in the South African Baviaanskloof Catchment. Although, simplifications and assumptions were required to bridge data gaps, solid ESs estimates could be obtained, using available data and knowledge of land management-ES provision relationships. Our study shows that combining various mapping and modelling methods with scenario analysis is an efficient way to enhance understanding of ESs and, subsequently, to guide land use and management decisions in data-scarce areas. Lack of high resolution biophysical data is often a problem in ES modelling (Leh et al., 2013), as was the case for this study. We emphasize that the strength of a simple model is that it yields quantitative estimates when empirical data are limited.

Results show that the provision of ESs depends on land management as pristine ecosystems provide a substantially different mix of ESs than ecosystems recently restored or managed as grazing lands. While livestock and game farming in combination with thicket restoration provide a wide set of ESs, formal nature conservation and crop cultivation provide a small number of different ESs. We show that a combination of light grazing, low input agriculture and nature conservation and restoration is the most promising for the sufficient provision of multiple ESs in the Baviaanskloof Catchment. The findings can help to strengthen and further optimize the local stakeholders' choice regarding the future management of the area. Farmers start to diversify towards a more sustainable agriculture, restore thicket vegetation and plans are developed for carbon and water trading (Mander et al., 2010a). Land management for tourism and nature conservation private lands can be combined, especially if partnered with incentives (Reyers et al., 2009). This also fits into the larger scale vision of creating a Baviaanskloof conservation mega-reserve (van Eck et al., 2010) and conservation and migration corridors across the whole subtropical thicket biome (Rouget et al., 2006).

Acknowledgements

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Grazing cattle on natural rangelands

Chapter 5

Mapping and modelling trade-offs and synergies between grazing intensity and ecosystem services in rangelands using global-scale datasets and models

Vast areas of rangelands across the world are grazed with increasing intensity, but little attention has been paid to various interactions between livestock production, biodiversity and other ecosystem services. This study explicitly determines trade-offs and synergies between ecosystem services and livestock grazing intensity on natural rangelands. Grazing intensity and its effects on forage utilization by livestock, carbon sequestration, erosion prevention and biodiversity are quantified and mapped, using global datasets and models. Results show that on average 4.2% of the biomass produced is consumed by livestock annually. On average, erosion prevention is 10% lower in areas with a high grazing intensity compared to areas with a low grazing intensity, whereas carbon emission is more than four times higher under high grazing intensity compared to low grazing intensity. Rangelands with the highest grazing intensity are located in the Sahel, Pakistan, West India, Middle East, North Africa and parts of Brazil. These high grazing intensities result in carbon emission, low biodiversity values, low capacity for erosion prevention and unsustainable forage utilization. Although the applied models simplify the processes of ecosystem service supply, our results provide geographically explicit and policy-relevant information to protect biodiversity and manage ecosystem services on natural rangelands. This is important, as natural rangelands will likely be put under more pressures with the increasing future demand for livestock products.

Keywords: biodiversity, net primary production, carbon sequestration, erosion prevention, natural rangeland, livestock production

K. Petz, R. Alkemade, M. Bakkenes, C.J.E. Schulp, M. van der Velde, R. Leemans (submitted) Mapping and modelling trade-offs and synergies between grazing intensity and ecosystem services in rangelands using global-scale datasets and models

5.1 Introduction

Rangelands are primarily natural grasslands, scrublands, woodlands, wetlands and (semi-)deserts and they cover between a quarter to half of the world’s land area (Alkemade et al., 2012; WRI, 1986). Vast areas of the rangelands are used and managed for pastoral livestock grazing with increasing intensity (MA, 2005a; Steinfeld et al., 2006). Livestock production, which is the principle land use in the world, creates livelihoods for one billion poor people through their pastoralist livestock husbandry (Steinfeld et al., 2006). The largest extent of pastoral livestock grazing systems is found in savannas, grasslands, shrublands, and (semi-)deserts (Asner et al., 2004). In these areas, people rely directly on the ecosystem services (ESs), such as raw materials, food and water. Additionally, rangelands store a vast amount of carbon (Herrero et al., 2009; MA, 2005a). The supply of ESs highly depends on the natural productivity and management of rangelands. Management determines the grazing intensity, a ratio between biomass grazed and biomass produced (Bouwman et al., 2005). Increasing livestock numbers and poor management causes widespread overgrazing and degradation of rangelands (Asner et al., 2004; Khan and Hanjra, 2009) and their ESs (Gisladottir and Stocking, 2005; MA, 2005a). Over the past decades, biodiversity decline and ESs degradation raised international concerns. This resulted in the inclusion of ESs in the Convention on Biological Diversity’s 2020 Aichi targets (Larigauderie et al., 2012; Mace et al., 2010).

Understanding the spatial pattern of livestock grazing intensity and its effect on ESs supply is important to manage rangelands sustainably. However, the spatial pattern of grazing intensity is poorly known and comprehensive global data on grazing systems are scarce (Kuemmerle et al., 2013). The few studies on the impact of grazing on ecological and hydrological processes or on ESs are either qualitative (e.g. Asner at al. (2004)) or are limited to a smaller geographic area (e.g. Ford et al. (2012)).

A continuous increase of livestock grazing intensity, partly driven by global demand for livestock products, may (further) impair biodiversity, enhance climate change (i.e. through additional carbon emission or lowered sequestration capacity), accelerate soil erosion and decrease water quality on rangelands (Herrero et al., 2009; Steinfeld et al., 2006). Sustainable rangeland management could minimize these management-related trade-offs and may even stimulate synergies between multiple ESs (i.e. simultaneous enhancement of food production, biodiversity and other ESs). Linking a robust quantification of ESs (Crossman et al., 2013a) to grazing intensity would be a first step in identifying and quantifying the various trade-offs and possible synergies.

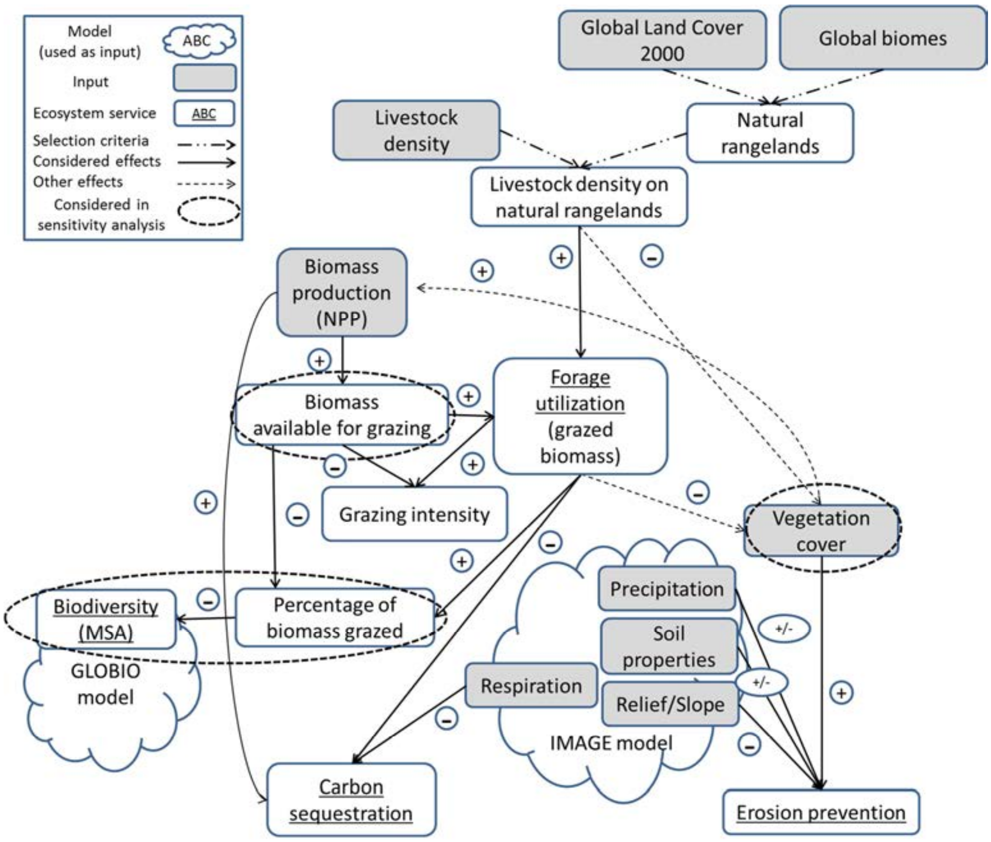

Figure 5.1: Conceptual framework of grazing intensity, forage utilization, carbon sequestration, erosion prevention and biodiversity quantification and modelling. (+) indicates positive effects, (-) negative indicates effects and (+/-) indicates complex effects that can be positive or negative.

This study aims to quantify trade-offs and synergies between forage utilization for livestock production, carbon sequestration, erosion prevention and biodiversity over a gradient of grazing intensity. Depending on the grazing intensity, different livestock production systems are possible: 1) grazing-based, 2) crop feed-based or 3) mixed system, combining the two (Bouwman et al., 2006; Herrero et al., 2009). We studied only natural rangelands relevant for livestock production. The productivity of these areas relates directly to the natural production capacity (Easdale and Aguiar, 2012; MA, 2005a). These rangelands stretch from tropical to temperate regions and are characterized by grazing-based and mixed livestock production systems. We delineate areas where grazing and livestock production are currently unsustainable, meaning that ESs are impaired by livestock grazing. This is achieved by analysing livestock grazing intensity and its consequences for ESs using global datasets and integrated models, such as the Integrated Model to Assess the Global Environment (IMAGE, PBL, 2006) and the Global Biodiversity Model framework (GLOBIO3, Alkemade et al., 2009).

5.2 Methods

Figure 5.1 shows the conceptual framework of the processes related to ES provision in rangelands. Based on a review of literature, data and models, we summarized the relations among the input data, livestock density and ESs. We first collected all available the data sources (Appendix 3), and selected and delineated natural rangelands using global maps (Section 5.2.1). Second, we quantified grazing intensity (Section 5.2.2). Third, we established relationships with ESs (Section 5.2.3). Fourth, we analysed the trade-offs and synergies between ESs under varying grazing intensity and finally, we performed a sensitivity analysis to quantify the effects of model inputs on ESs outputs (Section 5.2.4).

5.2.1 Data sources and delineation of natural rangelands

Global biophysical and socio-economic data were identified to derive ES estimates for rangeland ecosystems (the complete list of biophysical and socio-economic datasets is available in Appendix 3). For the present study, we selected spatial data on biophysical properties and livestock density (Table 5.1). The selected datasets have a high resolution (higher than 30 arc minutes, the common resolution of global assessment models) and are consistent with the data needed for the IMAGE model (PBL, 2006). These datasets were used in combination with intermediate outputs from the IMAGE model (Table 5.1).

Table 5.1: Spatial datasets used as input in this study. For gridded datasets resolution is given.

Data description	Unit	Resolution	Source and year
Net primary production	tC ha year^{-1}	30 arc second	MODIS (Zhao et al., 2005)
Livestock density (cattle, buffalo, sheep and goat)	TLU km^{-2}	3 arc minutes	The Gridded Livestock of the World (FAO, 2007)
Vegetation cover and fraction	Classes	30 arc second	GLC 2000 (JRC 2003)
Rangeland selection	Classes	polygons	WWF biome (Olson et al., 2001)
Respiration	tC km^{-2}year^{-1}	30 arc minutes	IMAGE model
Precipitation (monthly sum)	mm	30 arc minutes	IMAGE model
Soil texture (clay/silt percentage in 0-30 cm soil)	%	30 arc second	Harmonized soil database (FAO et al., 2012)
Bulk density (0-30 cm topsoil)	Kg m^{-3}	30 arc second	Harmonized soil database (FAO et al., 2012)
Soil depth	cm	30 arc minutes	Unpublished report (Schulp, 2012)
Relief	m	30 arc second	GTOPO30DEM (GLOBE Task Team, 1999)

Following Alkemade et al.'s (2012) approach, we used the Global Land Cover (GLC) 2000 (JRC, 2003) database to select vegetation types characteristic for natural rangelands (selected classes are 1) shrub cover, closed-open, evergreen, 2) shrub cover, closed-open, deciduous, 3) herbaceous cover, close-open, 4) sparse herbaceous or sparse shrub cover and 5) regularly flooded shrub and/or herbaceous cover). We used the biome classification of Olson et al. (2001) to select areas that fall under one of the rangeland biomes (selected biome classes are 1) desert, 2) tropical and subtropical grassland, savannas and shrubland, 3) temperate grassland, savannas and shrubland, 4) montane grassland, savannas and shrubland, and 5) mediterranean forests, woodlands and scrubs).

5.2.2 Quantifying grazing intensity

Grazing intensity (0-1) was calculated as the ratio between biomass grazed and biomass available for grazing. The grazed biomass is the plant biomass consumed by livestock and depends on livestock density. The available biomass depends on the net primary production (NPP) and on the edibility of the vegetation. Livestock density data were obtained from FAO (2007). We aggregated different livestock types into Tropical Livestock Units (TLU) as a common unit. Cattle, buffalo, sheep and goat densities were converted into TLUs and were summed (1 TLU equals 250 kg body weight (de Leeuw and Tothill, 1990) and conversion factors are: goat and sheep 0.1, cattle 0.6, buffalo 0.5 (FAO, 2012b)) (Figure 5.2). NPP was estimated from satellite imagery (Zhao et al., 2005) (Figure 5.2).

Table 5.2: Grazing intensities and corresponding livestock production systems, based on Alkemade et al. (2012; 2009), Herrero et al. (2009) and de Groot et al (2010b).

Livestock grazing intensity	**Description of grazing intensity and corresponding livestock production system**
Low intensity	Nearly pristine natural rangeland with marginal grazing-based livestock production system and minimal human intervention. Natural plant species grazed by domestic animals at rates similar to those of free-roaming wildlife. Livestock production is below the natural production capacity.
Moderate intensity	Natural rangeland with grazing-based production system. Human intervention is restricted to low external input (e.g. manure). These rangelands have moderate stocking rates. Grazing follows seasonal patterns. The vegetation structure differs from pristine natural rangelands, but the original ecosystem structure and species composition remains. Livestock production equals the natural production capacity.
High intensity	Intensively used and (partly) modified natural rangelands with mixed production system. Management heavily depends on external inputs and high resource extractions from the original ecosystems. Stocking rate is high. Livestock production exceeds the natural production capacity and grazing is supplemented with feed application.

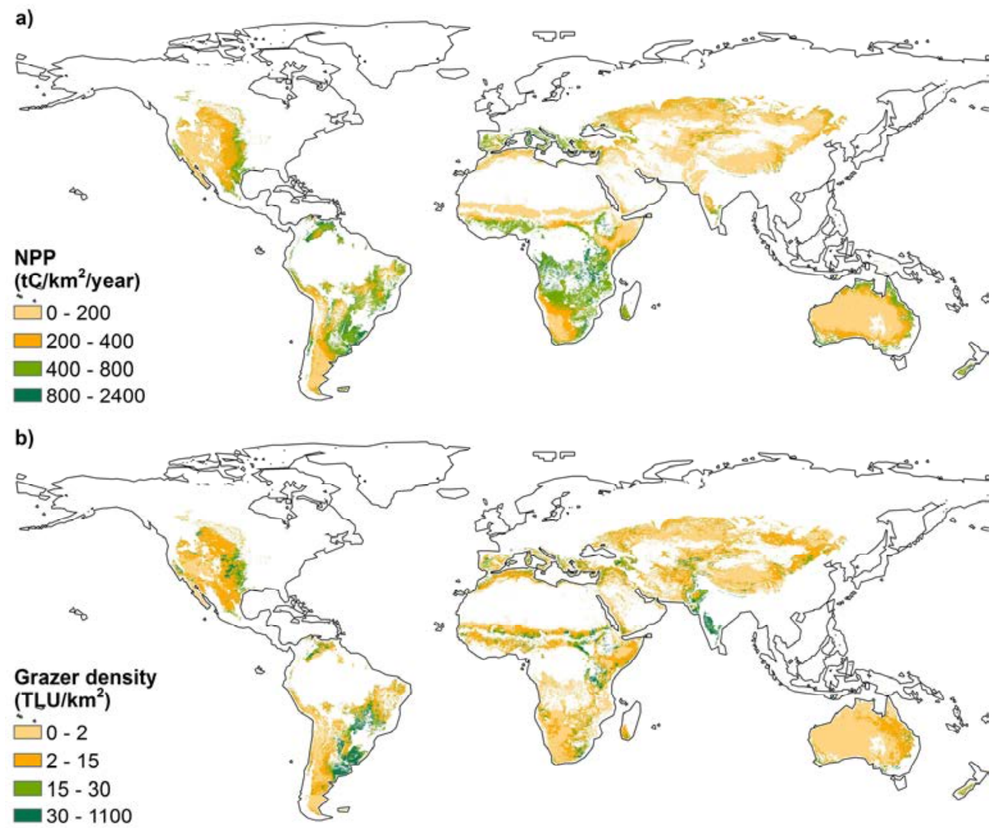

Figure 5.2: Annual biomass production (a) and grazer density (b) on natural rangelands.

We identified three livestock grazing intensity categories: low, moderate and high intensity grazing (Table 5.2). When livestock's forage needs are lower than the available biomass (i.e. low intensity) or equal the available biomass (i.e. moderate intensity), grazing-based production was assumed. Otherwise, the difference is supplemented with feed (i.e. high intensity).

5.2.3 Modelling ecosystem services

Considering the impact of livestock grazing and the availability of global quantitative data, we assessed forage utilization, carbon sequestration, erosion prevention, and biodiversity. A separate model for each ES and biodiversity was chosen and further developed to estimate the interactions between livestock grazing, ecosystem and other environmental properties (e.g. climate), and the production of ESs. In our models, forage utilization, carbon sequestration and biodiversity were linked to livestock density, whereas erosion prevention was linked to the fraction of vegetation cover (Figure 5.1). The modelling and spatial analysis was carried out in the ArcGIS 10 environment (ESRI, 2011).

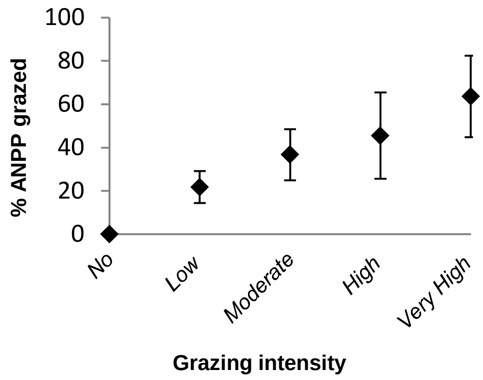

Figure 5.3: Average aboveground NPP grazed (%) and corresponding standard errors for qualitative grazing intensities based on the data provided by Milchunas and Lauenroth (1993). ANPP= aboveground net primary production

Forage utilization for livestock production

Forage utilization (tC km^{-2} year^{-1}) is defined as the biomass grazed annually by livestock. Forage utilization is the product of the forage requirement of an individual animal and livestock density. Daily dry matter intake of an individual animal is usually taken as 2.5% of the body weight (Bekure et al., 1991; de Leeuw and Tothill, 1990; Desalew et al., 2010). Including also the biomass loss caused by trampling, the annual forage requirement of 1 TLU was taken as 1.8 t C, calculating with 10 kg TLU^{-1} daily dry matter intake (Bekure et al., 1991).

Forage utilization is related to the productivity of rangeland and cannot exceed the biomass available for grazing. The amount of biomass edible for livestock varies among ecosystems and plant species. The maximum biomass available for grazing is defined as the average percentage of aboveground NPP grazed under the highest grazing intensity according to Milchunas and Lauenroth (1993), and is taken as 64% (Figure 5.3). The share of the aboveground NNP varies between and within different rangelands ecosystems (House and Hall, 2001; Ruimy et al., 1994). We assume the aboveground NPP to be 60% of NPP, a percentage House et al. ((2001) pp.374) reported for tropical savannas and grasslands based on various sources.

Carbon sequestration

Carbon sequestration (tC km^{-2} year^{-1}) is the annual surplus of carbon remaining in an ecosystem and is calculated by subtracting the biomass removed by gazing (i.e. forage utilization) and the heterotrophic respiration, from the NPP. This is the net ecosystem productivity (Koffi et al., 2012; Randerson et al., 2002; Schulp et al., 2012). We used the heterotrophic respiration output simulated by the IMAGE carbon model (resolution 30 arc minutes) (van Minnen et al., 2006). IMAGE calculates the heterotrophic respiration from carbon stock in different soil compartments, turnover rates, soil water and temperature (Goldewijk et al., 1994). Because of lack of consistency in studies on the effect of grazing (Schuman et al., 2002; Smith et al., 2008), the effect of grazing on respiration was not included.

Erosion prevention

In general, long-term livestock grazing negatively affects vegetation cover (Amezaga et al., 2004; Jones, 2000; Schuman et al., 1999) and therefore can increase erosion risk (Asner et al., 2004; Reynolds and Stafford Smith, 2002). Annual average erosion risk was estimated by the IMAGE-USLE model (Hootsmans et al., 2001). The IMAGE-USLE model (resolution 30 arc minutes) is a simplified version of the Universal Soil Loss Equation (USLE) developed for global-scale erosion analysis. A dimensionless erosion risk index (0-1, with 1 indicating the highest risk) is calculated as the product of erodibility, rainfall erosivity and land use/cover's susceptibility to erosion. Erodibility reflects a soil's sensitivity to erosion and depends on soil properties (i.e. texture, bulk density and soil depth) and relief. Rainfall erosivity depends on rainfall parameters (for input see Table 5.2). The land use/cover index reflects the level of erosion protection by the land cover (0-1, with 1 indicating the lowest protection). We used the erodibility and rainfall erosivity maps calculated by the IMAGE-USLE model and refined the land use/cover index based on the GLC2000 global land cover map (JRC, 2003). We derived the vegetation cover fraction (0-1) for grass, shrub or woodland vegetation types (closed to open shrub & herbaceous = 0.6, sparse shrub & herbaceous = 0.1, regularly flooded shrub & herbaceous = 1.0, forest & woodland = 0.6). The cover fraction was multiplied with the percentage of the corresponding vegetation type. The specific cell values were summed for all natural rangelands to obtain the total vegetation fraction. This fraction was subtracted from 1 to indicate the negative linear relationship between vegetation cover fraction and susceptibility to erosion (Roose, 1996; Wischmeier and Smith, 1978). The resulting value is the land use/cover index. Erosion risk was calculated from the land use/cover index, erodibility and rainfall erosivity maps. Annual erosion risk was subtracted from 1 to express erosion prevention.

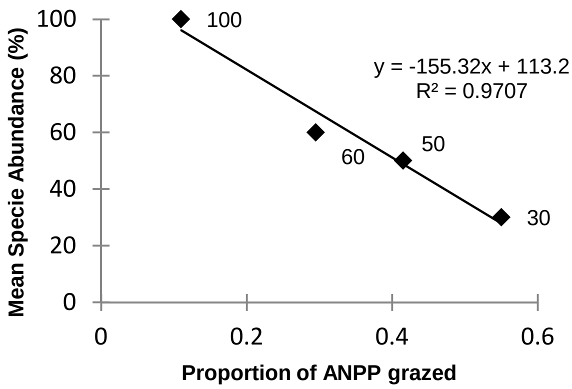

Figure 5.4: Proportion of aboveground NPP grazed related to MSA values. The linear function was used to calculate MSA from the proportion of aboveground NPP grazed. ANPP= aboveground net primary production

Biodiversity

The GLOBIO3 global biodiversity modelling framework (Alkemade et al., 2009) uses the Mean Species Abundance index (MSA, 0-1) to describe biodiversity intactness. We used the relationship from GLOBIO3 between MSA and livestock grazing intensity for biodiversity estimates. This relationship is based on an extensive systematic literature review (Alkemade et al., 2012; Alkemade et al., 2009). Heavy livestock grazing negatively affects biodiversity by changing plant composition and soil compaction, homogenization of landscape and competition with wildlife (Herrero et al., 2009). We estimated the effect of livestock grazing on biodiversity from the amount of biomass grazed. This was based on a linear regression between the percentage of aboveground biomass grazed, provided by Milchunas and Lauenroth (1993), and the grazing intensities, and therefore the MSA values provided by Alkemade et al. (2012) (Figure 5.4).

5.2.4 Analysis of ecosystem services under changing grazing intensity

First, we analysed and visually compared the spatial distribution of grazing intensity, forage utilization, carbon sequestration, erosion prevention and biodiversity. Next, minimum, maximum and average values of forage utilization, carbon sequestration, erosion prevention and biodiversity were calculated for the three grazing intensity categories and trade-offs and synergies were quantified. After this trade-offs and synergies were mapped by overlaying the grazing intensity and ESs maps. A trade-off was defined as high grazing intensity (or forage utilization) combined with low ESs and biodiversity supply, or vice versa. A synergy was defined as high grazing intensity (or forage utilization) combined with high ESs and biodiversity supply. Meaningful thresholds for high and low values were identified:

- Grazing intensity: <0.4 is low, and >0.6 is high (derived from the definitions in Table 5.2)
- Carbon sequestration: <0 is emission, and >0 is sequestration
- Erosion prevention: <0.7 low/moderate, and >0.7 is high (1-erosion sensitivity following the classification of Hootsmans et al. (2001))
- Biodiversity: <0.7 low, and >0.7 high (based on Alkemade et al. (2009))

Finally, we analysed how sensitive each ES and biodiversity is to changes in model input by selecting and testing three inputs. The 'maximum biomass available for grazing' influences forage utilization, carbon sequestration and biodiversity. The '% of aboveground NPP grazed – MSA' relation influences biodiversity and the 'vegetation cover fraction' input influences erosion prevention output (Figure 5.1). The following changes in the three inputs were made and their effects on ESs were analysed:

- Reducing the 'maximum biomass available for grazing' by half (from 64% to 32% of the aboveground NPP);
- Modifying the '% of aboveground NPP grazed – MSA' relation by taking first the minimum ($y = -248.37x + 114.31$) and then the maximum aboveground NPP grazed value ($y = -113.78x + 112.44$) for corresponding management intensities; and
- Changing the 'vegetation cover fraction' input maps:
    1) GLC2000 global land cover (herbaceous, shrub, forest/woodland) (JRC, 2003);
    2) MODIS tree cover map (Hansen et al., 2003); and
    3) MODIS grass and shrub cover maps[2]

[2] Binary MODIS MOD12C1 0.25 Degree Land Cover Climate Modeler Grid. Available at http://duckwater.bu.edu/lc/ from Department of Geography, Boston University, Boston, Massachusetts, USA. Accessed last November 10th 2013

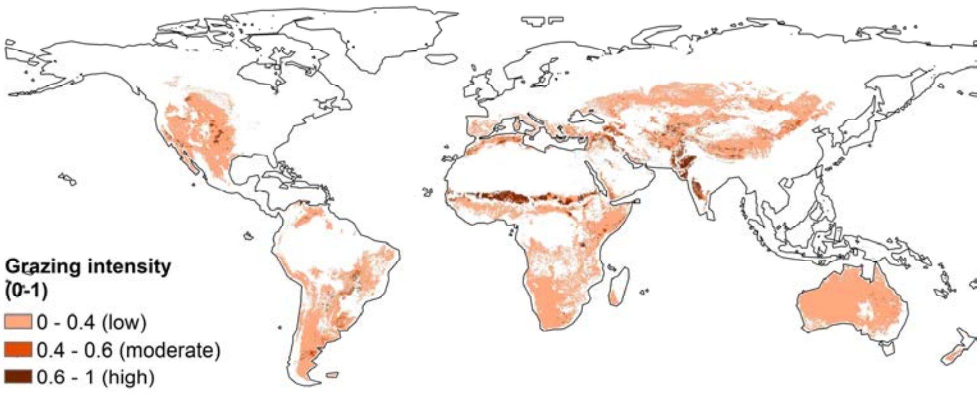

Figure 5.5: The spatial distribution of management intensity and corresponding management categories on worldwide natural rangeland.

5.3 Results

5.3.1 Level and spatial distribution of grazing intensity

Of the c. 10.86 PgC biomass produced on natural rangelands about 4.2% is grazed by livestock (0.46 PgC). A visual comparison between the NPP and livestock density maps indicates that most areas with higher biomass production support higher livestock densities. Most of the natural rangelands have low grazing intensity (Figure 5.5). Moderate and high grazing intensity occur in the Sahel; in west India, Pakistan and Afghanistan; and in the Middle East. Some intensively grazed spots are located also in Brazil, southern Argentina and in the Midwest USA. Grazing intensities close to one mean that grazing intensity is close to the maximum production capacity. It is highly probable that in these regions livestock is supplemented with feed.

5.3.2 Spatial distribution of ecosystem services

The distribution of forage utilization, carbon sequestration, erosion prevention, and biodiversity varies across rangelands (Figure 5.6). Annual forage utilization is the highest (above 30 tC km^{-2}) in Central Argentina, West India, Midwest USA and parts of North and East Africa (Figure 5.6a). Annual forage utilization is the lowest (below 2 tC km^{-2}) in Western Australia, parts of Southern Africa, the Andes Mountains (West Argentina and Chile), Kazakhstan and the Western USA (Figure 5.6a).

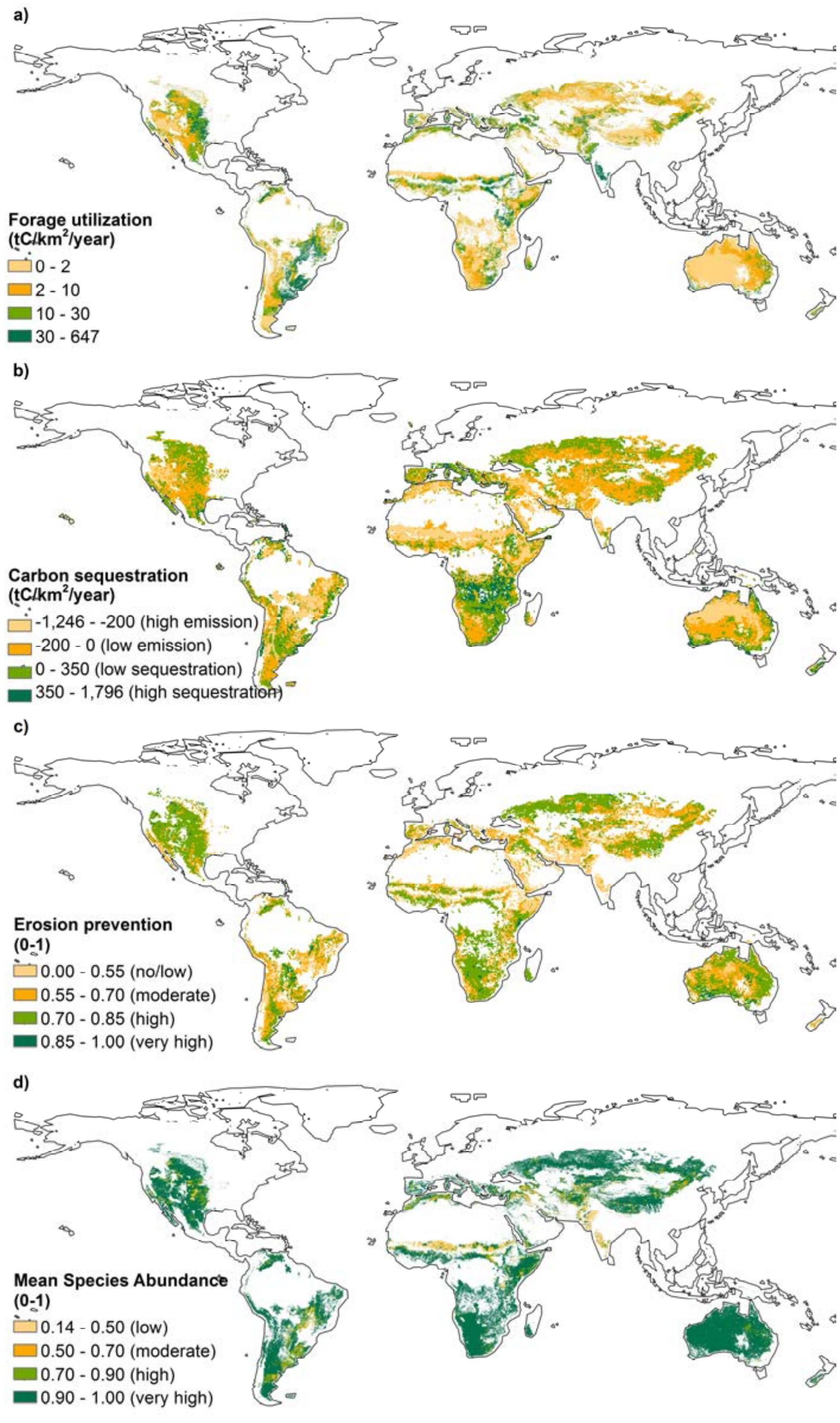

Figure 5.6: The spatial distribution of forage utilization (a), carbon sequestration (b), erosion prevention and (c) biodiversity (d) on global natural rangeland. The first two maps have 3 arc minutes resolution, whereas the second two 30 arc minutes.

Annual carbon emission is highest (above 200 tC km^{-2}) in Sahel, North West Australia, North Africa, Western USA and in Brazil south of the Amazon Basin (Figure 5.6b). Annual carbon sequestration is highest (above 350 tC km^{-2}) in parts of Southern Africa, the southern edge of Australia and Southern Europe (Figure 5.6b). Most of the rangelands are net carbon emitters. High carbon emission result from low biomass production and high livestock density.

Erosion prevention is low (below 0.55, following the classification of Hootsmans et al. (2001)) in West India and Pakistan, the Horn of Africa, Morocco, Chile and Central Western Australia (Figure 5.6c). Low erosion prevention coincides with high grazing intensity only in West India and Pakistan. Low erosion prevention is a result of steep slopes with sensitive soil (e.g. west coast of South America and North Coast of Africa), scarce vegetation cover (e.g. north coast of Africa, parts of South America, Central Australia), heavy rain events (e.g. West India, Pakistan, the Horn of Africa, Chile, North West Australia), or the combination of these. Erosion prevention is high (above 0.70) in Kazakhstan, parts of China, Southern Africa, the Midwest USA and some parts of Australia (Figure 5.6c).

Biodiversity on most of the natural rangelands is hardly impaired by livestock grazing (Figure 5.6d). This high biodiversity is the result of low or moderate grazing intensity of large areas. Low biodiversity (MSA below 0.5) occurs in the Sahel, Middle East, West India and Pakistan.

5.3.3 Synergies and trade-offs between grazing intensity and ecosystem services

Management intensity shows a synergy with forage utilization and a trade-off with carbon sequestration, erosion prevention and biodiversity (Table 5.3). Hence, areas with high grazing intensity emit more carbon, hold lower biodiversity, demonstrate lower erosion prevention and utilize more forage, in comparison with areas with low grazing intensity.

Table 5.3: Average (minimum-maximum) values of forage utilization, carbon sequestration, erosion prevention, and biodiversity for all natural rangelands and for changing grazing intensities.

Grazing intensity	**Forage utilization (tC km^{-2} $year^{-1}$)**	**Carbon sequestr. (t km^{-2})**	**Erosion prevention (0-1)**	**Biodiversity (0-1)**
Low (0.0-0.4)	9 (0-241)	-56 (-1082 -1456)	0.69 (0.07-1.00)	0.98 (0.73-1.00)
Moderate (0.4-0.6)	44 (0-302)	-147 (-888-968)	0.64 (0.16-0.98)	0.65 (0.53-0.73)
High (0.6-1.0)	43 (0-647)	-230 (-992-857)	0.62 (0.11-0.96)	0.24 (0.14-0.53)
All natural rangelands	12 (0-647)	-69 (-1082-1456)	0.68 (0.07-1.00)	0.92 (0.14-1.00)

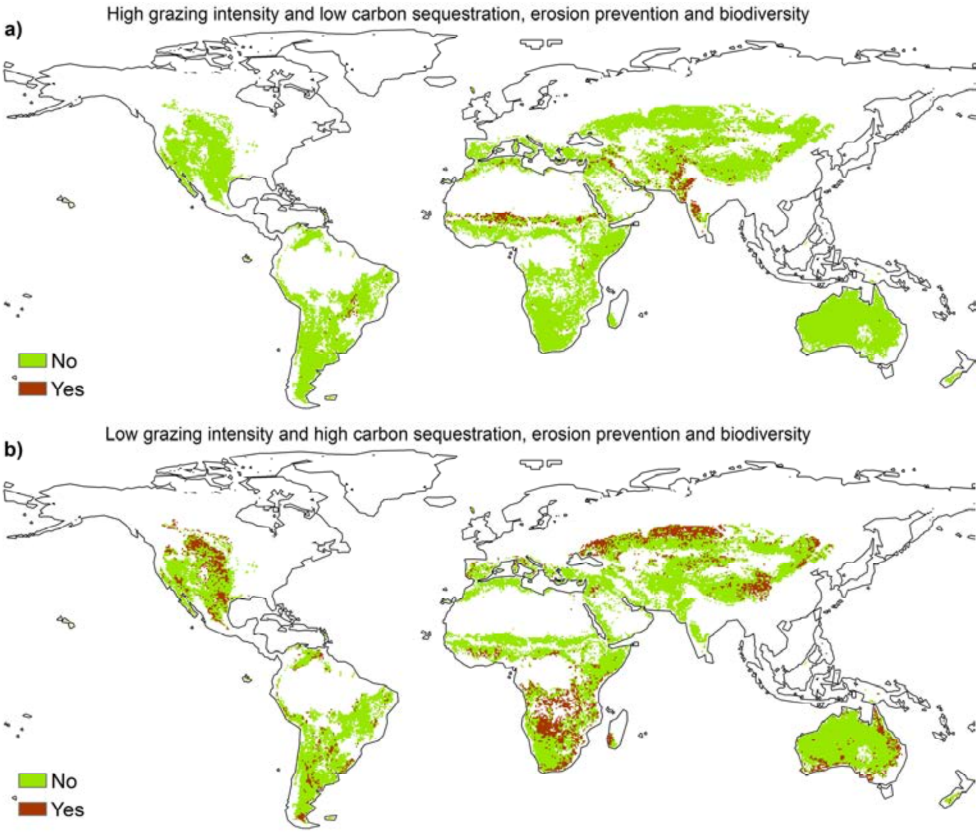

Figure 5.7: Trade-offs between grazing and carbon sequestration, erosion prevention and biodiversity.

On average, erosion prevention is 10% lower in areas with a high grazing intensity compared to areas with a low grazing intensity. Carbon sequestration shows the largest range over the intensity categories, with more than four times higher carbon emission in areas with a high grazing intensity compared to areas with a low grazing intensity (Table 5.3).

High grazing intensity and low carbon sequestration, erosion prevention and biodiversity are found in the Sahel, West India and Pakistan, Middle East, and parts of Brazil and Northern Africa (Figure 5.7a). On these areas carbon sequestration, erosion prevention and biodiversity are the most impaired by livestock grazing and forage utilization is unsustainable. Low grazing intensity and high carbon sequestration, erosion prevention and biodiversity are found in Southern Africa, Midwest USA, Kazakhstan and parts of China (Figure 5.7b). On these areas carbon sequestration, erosion prevention and biodiversity are the least impaired by livestock grazing and forage utilization is sustainable. No areas with a synergy between grazing intensity and carbon sequestration, erosion prevention and biodiversity could be identified.

5.3.4 Sensitivity analysis

Reducing the 'maximum biomass available for grazing' by half had a local and negligible effect on forage utilization, carbon sequestration and biodiversity because of the low biomass consumption on most of the natural rangelands. Thus only small areas with high biomass consumption disappeared. The average biomass consumption and the total proportion of grazed biomass showed only decimal changes. The carbon emission/sequestration range did not change and the mean carbon emission changed negligibly. Areas with low biodiversity slightly expanded in Sahel, West India and Pakistan.

The modification of the '% aboveground NPP grazing – MSA' relation had a small effect on biodiversity. Using the minimum values of the aboveground NPP grazed decreased the average MSA from 0.92 to 0.87, and using the maximum values of the aboveground NPP grazed increased the average MSA to 0.95. The spatial distribution of biodiversity did not change.

The map of the vegetation cover fraction had a bigger effect on the spatial pattern of erosion prevention than on the average value of erosion prevention. The value range of erosion prevention slightly decreased with the MODIS tree cover map (0.13-1.00) and decreased even further with the MODIS grass and shrub cover maps (0.29-1.00) compared to the GLC2000 map (0.07-1.00). Opposed to the GLC2000 map, none of the MODIS maps covers natural rangelands completely. The MODIS tree cover map has the smallest coverage and the corresponding erosion map shows low erosion prevention in the Horn of Africa, Chile and North Australia. This is similar to the erosion map based on the GLC2000 map. The erosion map derived from the MODIS grass and shrub maps indicates low erosion prevention in the Horn of Africa, Midwest USA and Kazakhstan-Mongolia-Inner-China. This is rather different from the erosion map based on the GLC2000 map. We attribute these relatively big differences in the erosion maps to the difference in coverage and vegetation types and classifications of the inputs.

5.4 Discussion

Because of the lack of empirical information, spatially explicit management impacts on ESs are, generally, quantified using spatial models. Our quantitative results indicate that high grazing intensity has an adverse effect on biodiversity, carbon sequestration and erosion prevention. We believe that the presented approach and results are credible as we used widely accepted global data sets, and robust model parameters and assumptions. All our calculations and results are also crosschecked with literature and other model results.

Our methodology's limitation and strengths, the result's validity and management relevance are discussed below.

5.4.1 Advances of the applied modelling method

At global level mapping and modelling of ESs have been limited to a few, mainly provisioning and regulating services including food provision, water availability and carbon storage and sequestration (Naidoo et al., 2008; Schulp et al., 2012). We investigated these most-studied ESs, together with biodiversity and placed them in the context of rangeland management and grazing intensity. Existing land use statistics on grazing land are limited to livestock densities, and data about the extent of grazing land and the spatial pattern and amount of biomass grazed are missing (Erb et al., 2009; Kuemmerle et al., 2013). This study goes beyond the current knowledge by quantifying grazed biomass and studying the impacts of livestock production and its sustainability in the context of natural production capacity using ESs.

The current study combined spatial datasets and data from meta-analysis with model relations. Model relations bridge data gaps and generate extensive spatial information, when quantitative or empirical data are missing or are available only from a small suite of cases. Applying empirical relationships to ecosystem properties is a common ESs modelling approach (e.g. Schulp and Alkemade (2011) and Maes et al. (2012)).

The approach presented in this article has several advances. Biodiversity intactness was estimated by GLOBIO3 that uses general relationships for effects of land use, infrastructure, climate change and fragmentation (PBL, 2006). This prior study reclassified GLC2000 land cover/use classes to biodiversity impacts. As an improvement to this, we established a continuous relationship between grazed biomass and its effect on biodiversity to obtain a more refined picture on grazing effects. The 3 arc minutes resolution at which we mapped forage utilization and biodiversity, is higher than the resolution at which most global models operate. The refined land use/cover index we used to quantify and map erosion prevention is also an improvement compared to the rough land cover and crop classes that Hootsmans et al. (2001) used. Future research may focus on the inclusion of additional ESs and socio-economic data (Appendix 3).

5.4.2 Uncertainties of global datasets

Combining different datasets in a model may induce uncertainties in the final results. Global datasets are often estimated from base data sources with modelling techniques and involve a certain error. One should be aware of the uncertainty involved in global-scale datasets and data products. The uncertainties of inputs and modelling relations propagates into model outputs and can have large effects on the results, especially when it comes to global datasets (Schulp and Alkemade, 2011).

High-resolution and high-accuracy spatial data are key for global ES modelling. Although the spatial resolution of global models is often limited to 30 arc minutes (Verburg et al., 2012), this study showed that higher resolution (3 arc minutes) modelling is also possible. Higher resolution, however, does not necessarily mean higher accuracy. The lower resolution GLC2000 global land cover map, for example, has a higher accuracy than the finer GlobCover land cover map (Fritz et al., 2011) and was therefore preferred for modelling spatial patterns of ESs.

Although proper validation of global data sets is extremely difficult due to differences in temporal and spatial consistencies, classification systems and scaling (Kuemmerle et al., 2013; Verburg et al., 2011b), these datasets remain the sole information sources when it comes to global environmental modelling. As no alternative dataset for livestock density is available (Appendix 3), the FAO livestock density map is the best possible dataset to analyse feed requirements and quantify and distribute environmental impacts of livestock production (FAO, 2007).

The effects of uncertainties of the input data can be addressed adequately with a sensitivity analysis. The sensitivity analysis revealed that maps of vegetation cover fraction strongly alter the results, while the impact of uncertainties in the other inputs was relatively limited. The results are considered therefore robust.

5.4.3 Grazed biomass close to prior estimates

To assess the credibility of the results, model results were compared with independent datasets and model outputs, where possible. We are unaware of any appropriate global spatial dataset about grazed biomass. The few global-scale studies that estimated annually grazed biomass show big variations depending on the definition of grazing land and applied methods (Haberl et al., 2007). Although our estimate (0.46 PgC) is on the lower side, it is of the same order of magnitude as previous estimates calculated from feed balances (1.2 PgC and 0.4 PgC (Haberl et al., 2007), 2.4 PgC (Imhoff et al., 2004)). Bondeau et al. (2007) estimated the annual grazed biomass at ~3.8PgC on all grazed land for the year 2000 with the global LPJ model. This estimate is very high because high livestock intensity was simulated. We studied only natural rangelands, while other studies included a larger extent of grazing land, explaining our low estimate. Additionally, we used a livestock density map derived from actual statistics, thus closely matching actual livestock numbers.

The areas contributing most to the global livestock production are outside natural rangelands (i.e. feed-based systems in Europe, India, USA etc.). In Latin America, the

Middle East, the Indian subcontinent, and East Asia, grazing is dominated by domesticated animals in extensive (i.e. low intensity) production systems (Bouwman et al., 1997). This is well represented in the geographic coverage and in the results of our study.

About 3-10% of the consumable NPP is eaten by wild animals (Bouwman et al., 1997). This is comparable to the average biomass consumption we calculated for livestock (c. 4.2%). It would be worth to investigate the effect of livestock grazing on wildlife and the effect of wildlife grazing on biomass. This is particularly relevant for areas where wild grazers are abundant and overgrazing is a problem, such as Africa and Australia.

5.4.4 Interpretation of ecosystem services results and comparison with other studies

To support our findings, we visually compared ESs value ranges and patterns with several other studies. We did not find any independent spatial datasets about forage utilization and the amount of biomass grazed. The pattern of our forage utilization map agrees with the global grassland (meat) production map of Naidoo et al. (2008), as both indicate high values for Central Argentina, West India, Midwest USA and parts of East Africa. This is not surprising, as Naidoo et al. (2008) based the calculations also on the FAO livestock density map (2007).

The carbon sequestration we estimated is seemingly low, as most natural rangelands were found to emit carbon. Savannas and grasslands sequester less carbon or even emit carbon as a consequence of dry weather (Potter et al., 2012) grazing and degradation (Grace et al., 2006; Steinfeld et al., 2006). Our carbon sequestration estimates agree with the high carbon emission in Sahel, parts of Brazil, Central Australia and Pakistan-India indicated by Naidoo et al. (2008). At the same time, our results show net sequestration for Southern Africa whereas Naidoo et al. (2008) showed net emission. The high NPP and low intensity grazing explains our carbon sequestration result for Southern Africa. Similarly to Naidoo et al. (2008), we also found a negative relationship between grassland (meat) production and carbon sequestration.

Our estimated effect of livestock grazing on biodiversity seems low. Much of the natural rangelands fall under biodiversity hotspots and conservation priority areas (Myers et al., 2000), including the areas impaired by livestock grazing in North Africa and India (Figure 5.6d). Our results only show the effect of grazing itself. However, grazing is related to additional indirect pressures, such as land use change and fragmentation. Generally the conversion of original ecosystems to agriculture is the main cause of habitat loss and biodiversity loss (Pereira et al., 2012), while the impact of grazing itself is only a small fraction of the impact on biodiversity. This is also supported by prior GLOBIO3 model results (PBL, 2006).

Higher erosion prevention was projected for areas with higher tree cover and lower erosion prevention was projected for areas with higher bare cover. This is a much more refined picture of erosion prevention compared to the original results by Hootsmans et al. (2001). The pattern of erosion prevention agrees with the global erosion modelling results of Yang et al. (2003), who estimated similarly low erosion prevention for Pakistan-India, parts of South America, North Africa and the Horn of Africa. At the same time, they estimated higher erosion prevention for Australia and lower erosion prevention for the Western USA and parts of South Africa. As we demonstrated that the erosion model used in this study is sensitive to the input vegetation cover map, this small difference may come from the differences in input datasets. Also, Yang et al. (2003) used another erosion risk model (RUSLE vs. IMAGE-USLE).

The on-going transformation and degradation of natural rangelands and intensified livestock management leads to loss of biodiversity (Pereira et al., 2012) and to net carbon loss to the atmosphere (Grace et al., 2006). Our study emphasizes the big pressures certain regions face, especially the Sahel, Middle East, and parts of India and North Africa. Campbell et al. (2000) identified semi-arid and subtropical grassland (e.g. Sahel, Mongolia and China) as regions most exposed to environmental change, because of their sensitive vegetation and the increasing pressure due to food production requirements. Sahel is among the regions that experience the biggest increase also in crop production areas (Phalan et al., 2013).

5.5 Conclusion

Rangeland decision makers, such as land managers and national or international policy makers, need information on the natural productivity of rangelands and livestock grazing effects to develop region-specific policies and management strategies (Campbell and Stafford Smith, 2000). Policies and management strategies affect multiple ecosystems and ESs, while data are often available only from one or a few ecosystems (Campbell and Stafford Smith, 2000). Therefore, spatially explicit global-scale studies that quantify trade-offs and synergies among ESs have important implications for land managers and policy makers. Spatial data and GIS has been used to diagnose conservation problems and develop solutions for them, as well as to analyse the impact of management decisions on biodiversity and ESs at coarse scales (Swetnam and Reyers, 2011).

Our study is among one of the first studies to quantify the spatial patterns of ESs and link them to the management of rangelands. The presented spatially-explicit information about the effect of grazing intensity on ESs is important because, in contrast to trade-offs between cultivated agriculture and biodiversity (e.g. Tscharntke et al. (2012) and Phalan et al. (2013)), little attention has been paid to trade-offs between livestock production and biodiversity. The characterization of rangeland management is generally difficult compared to croplands, due to the complexity of rangeland ecosystems (e.g. wide diversity in plant communities, soils and landscapes), diverse grazing practices and inconsistent responses to grazing intensity (Schuman et al., 2002; Smith et al., 2008). By considering multiple ESs the trade-off analysis between agriculture and biodiversity becomes even more complete (Grau et al., 2013).

Our study emphasises that, although models are simplifications of the real world and the underlying data and assumptions contain many uncertainties, model applications provide policy relevant information to protect biodiversity and manage ESs. We emphasize that combining available spatial data sets with quantitative information from meta-analysis studies and models is an efficient way to quantify the spatial distribution of ESs, when quantitative empirical information is scarce. The present study demonstrated what previous studies suggested, namely that combining data from multiple scales and different sources can optimizes data use and improves global environmental modelling (Rounsevell et al., 2012; Verburg et al., 2011b).

Our study quantified trade-offs and synergies by quantifying and mapping the consequences of grazing management for ESs and biodiversity on natural rangelands. Results revealed a synergy between grazing and forage utilization and trade-offs between grazing and carbon sequestration, erosion prevention and biodiversity. Supported by the comparison with other studies we believe that our livestock density, NPP and vegetation cover fraction-based results are acceptable as a first-order estimate of the effect of grazing intensity on ESs. We conclude that increased livestock grazing triggers higher carbon emission, and lower erosion prevention and biodiversity. Areas with high carbon sequestration, erosion prevention and biodiversity (e.g. Southern Africa and parts of Central Asia) are currently sustainably grazed and are valuable for conservation. Opposed to this, restoration less intensive grazing and feed supplement are applicable on areas where one or more of these ESs or biodiversity are impaired. Regions with severe forage shortage, low erosion prevention, biodiversity degradation and high carbon emissions, are located in the Sahel, Middle East, Northern Africa, Pakistan and West India. On these areas livestock grazing reaches or exceeds the natural production capacity and is therefore unsustainable.

Livestock and environmental trade-offs are expected to further increase significantly in the future as a result of increasing demand for livestock products (Herrero et al., 2009). This increasing demand will likely put natural rangelands under an even bigger pressure. This study integrates knowledge about livestock grazing and multiple ESs provision, and is therefore important to facilitate sustainable rangeland management and biodiversity conservation.

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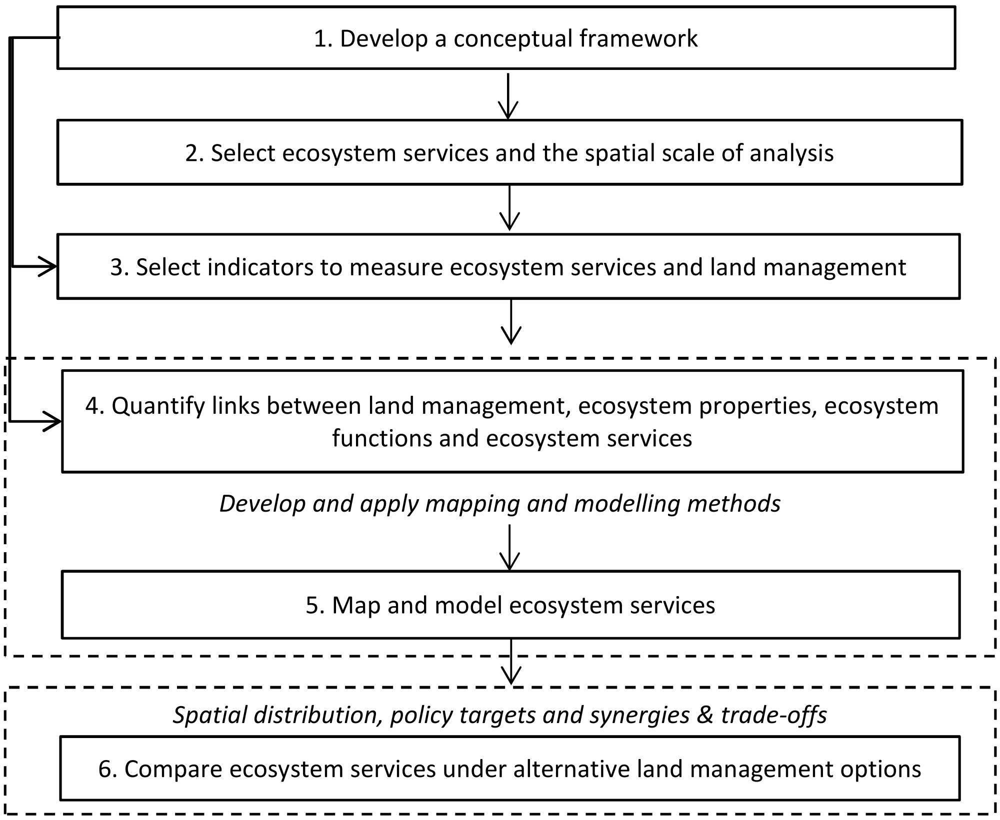
Chapter 6

Discussion and conclusions

The objective of this thesis was to develop a methodology to quantify the effect of land management on the spatial distribution of ecosystem services in order to determine ecosystem service trade-offs caused by land management, on a scale ranging from local ecosystems and landscapes to global biomes. The resulting methodology can be used to evaluate the effects of different land management options on ecosystem service delivery at different geographical scales.

The methodological steps followed in this study are summarized in Figure 6.1. First, a conceptual framework was developed (Step 1) including the study's boundaries and the choice of ecosystem services (Step 2). This framework was then applied to select the indicators (Step 3) and quantify (Step 4) and mapand model ecosystem services (Step 5). The framework development and its application to the selection of indicators were described in Chapter 2. All the steps that follow the framework development were undertaken in different case studies (Chapters 3, 4 and 5). The development and application of mapping and modelling methods and the comparison between alternative land management options were the main components of this research.



6.1 Methodological steps for spatially explicit quantification of land management effects on ecosystem services. Indicator selection and the quantification of ecosystem services are based directly on the framework (arrows on the left). The dashed boxes indicate the main contributions of the thesis work.

This chapter discusses the translation of the conceptual framework to actual mapping and modelling (Section 6.1). Next, the opportunities and limitations of the GIS-based mapping and modelling methods (Section 6.2), the heterogeneity and complexity of land management (Section 6.3), and dealing with complexity and data availability (Section 6.4) are discussed. After this, the methodological contributions of this thesis are synthetized (Section 6.5) and the main research findings are given (Section 6.6). Finally, implications for land management and policy (Section 6.6) and recommendations for future research are presented (Section 6.8), and conclusions are drawn (Section 6.9).

6.1 From a conceptual framework to actual mapping and modelling

The understanding and quantification of the relationships between land management and the provision of ecosystem services, which is the basis of the mapping and modelling work, required a considerable interdisciplinary effort, a comprehensive framework and the use of indicators. Since the publication of the framework presented in this thesis (van Oudenhoven et al., 2012), a few new ecosystem service frameworks have been developed, but none of them considered land management comprehensively (e.g. Bastian et al. (2013), Villamagna et al. (2013) and Kandziora et al. (2013)).

Table 6.1: Summary of ecosystem services studied in the three case studies in this thesis. The classification follows the TEEB study (TEEB, 2008, 2010).

Ecosystem service	**Groene Woud**		**Baviaanskloof**	**Natural rangelands**
	Indicator selection	Mapping and modelling		
Food provisioning (crop and meat)	✓	✓	✓	✓
Water supply			✓	
Provision of raw materials (fodder and fuel wood)		✓	✓	
Air quality regulation	✓	✓		
Climate regulation	✓	✓	✓	✓
Regulation of water flows	✓			
Erosion prevention			✓	✓
Pollination		✓		
Biological control	✓	✓		
Lifecycle maintenance (habitat for biodiversity and abundance of species)	✓	✓	✓	✓
Aesthetic information	✓			
Opportunities for recreation	✓	✓	✓	
Information for cognitive development	✓			

The conceptual framework facilitated the indicator selection. Ecosystem service indicators were chosen based on the scale of analysis, data availability and model applicability and requirements. Therefore, ecosystem services studied in multiple cases were not always measured with the same indicators. For example, lifecycle maintenance/biodiversity was assessed with a habitat suitability index in the Groene Woud (Chapter 3) and with the GLOBIO3 model (Alkemade et al., 2009), which uses the Mean Species Abundance index in the other two case studies (Chapters 4 and 5). Identifying an ecosystem service indicator did not necessarily mean that a service could be quantified, mapped or modelled. In the Groene Woud, indicators were selected for nine different ecosystem services. Regulation of water flows (water retention), aesthetic information and cognitive development could not be quantified and mapped in relation to land management (Table 6.1). In general, only ecosystem services for which information was available and which could be linked to land management and assessed with GIS-based methods were studied. In the Baviaanskloof Catchment, for example, there was insufficient quantitative spatial data on the generation of important ecosystem services, such as pollination, provision of medicinal resources or aesthetic information. More ecosystem services can be mapped and modelled at a lower level of the spatial scale than at a higher level. Ecosystem services related to landscape structures (i.e. air quality regulation, pollination and pest control) were studied only at landscape scale. The availability of spatial datasets and model relations made it possible to study the provision of food/raw materials, climate regulation (carbon sequestration) and lifecycle maintenance/biodiversity at all levels (Table 6.1).

The concept of ecosystem services has been used in an inconsistent way throughout the literature, sometimes in combination with the related concepts of ecosystem functions and landscape functions (Bastian et al., 2013; Villamagna et al., 2013). This thesis followed the TEEB (TEEB, 2008, 2010) definitions, according to which an ecosystem function is the ecosystem's capacity to provide an ecosystem services, whereas ecosystem services are the contributions to human well-being (i.e.human use or benefit from them (in)directly) (De Groot et al., 2010a). Distinguishing between function and service was impossible in some cases. Quantification and modelling were difficult, especially for habitat, and for some regulating and cultural services. The reasons for this are manifold. First, ecosystem functions and services are often measured in an inconsistent manner. For example, carbon sequestration or storage is used to measure both the ecosystem function and service (Chan et al., 2006; Raudsepp-Hearne et al., 2010). Second, the relationships between the function and the service are unknown, unquantifiable or uncertain. For instance, the quantitative contribution of carbon sequestration to global climate regulation is uncertain and it is difficult to translate the landscape's suitability for tourism to visitation rates (Petz and van Oudenhoven, 2012). Finally, spatial and quantitative data are lacking. This increases with spatial scale. Therefore, in the Groene Woud study, some ecosystem function and service indicators were modified to make a step from indicator selection to quantification, mapping and modelling. For example, lifecycle maintenance was first measured as the 'spatial distribution of migratory birds' (Chapter 2), and was modelled then as the 'occurrence of butterflies' (Chapter 3). In the South African and natural rangeland studies the difference between functions and services were not emphasized for practical reasons (i.e. scarcity of quantitative data and reliance on existing models).

A major contribution of this research is that it provides quantitative relationships between land use/management and ecosystem services, and quantifies ecosystem services under alternative land management systems and corresponding land use intensities. Some land management effects on particular ecosystem services could not be quantified, such as the effect of nutrient application on milk and maize production in the Groene Woud. Furthermore, in some cases arbitrary decisions needed to be made to translate the positive or negative effect of certain management practices into quantitative terms, such as the positive effect of landscape and wildlife diversity on recreation and ecotourism in the Groene Woud and Baviaanskloof Catchment. Nevertheless, these methodological shortcomings and assumptions did not compromise the results of ecosystem service bundles and the relative comparison of land management options.

6.2 Limitations and opportunities of GIS-based mapping and modelling methods

The mapping and modelling methods used in this thesis are similar to the GIS-based InVEST model (Kareiva et al., 2011). GIS has been used to diagnose conservation problems and develop related solutions, as well as to analyse the impact of management decisions on ecosystem services (Swetnam and Reyers, 2011). GIS is strong in spatial representation of land cover/use and ecosystem services, but it can hardly capture the dynamic character of land management activities. The InVEST model generally does not address land management effects beyond land cover/use change. The Baviaanskloof study showed that the InVEST water supply and erosion models (Kareiva et al., 2011) can be parameterized for grazing effects only if sufficient quantitative information is available from either the literature or local measurements.

Process-based dynamic models (Portela and Rademacher, 2001), also in combination with GIS (McKinney and Cai, 2002; Merwade et al., 2008; Nedkov and Burkhard, 2012), can simulate the effect of land management dynamics on ecosystem services. These approaches, however, are only suitable for a limited number of regulating services (e.g. regulation of water flow) (Villamagna et al., 2013) and may be less flexible when it comes to applications at various spatial scales.

On one hand, GIS-based mapping and modelling methods cannot describe all the relationships between ecosystems and social systems to define cultural services (Daniel et al., 2012). On the other hand, the qualitative approaches used for cultural services, such as aesthetic information and cognitive development, can hardly capture the link with land management and provide quantitative results.

All in all, GIS remains an environment suitable for the quantification and mapping of a relatively wide range of ecosystem services, as it is strong in the spatial visualization and analysis of environmental processes.

6.3 The heterogeneity and complexity of land management

There are several ways of measuring and analysing land management and the intensity of land use (Kuemmerle et al., 2013). In this thesis, land management was defined as the human activities that affect vegetation (land cover) directly or indirectly and aim to provide specific services. Land management activities, such as technical inputs, irrigation and animal choice, define the type and intensity of land use. This thesis analysed ecosystem services in relation to land cover, land use and the intensity of land use. Land cover/use has been studied widely before but not the changes that may occur within a land cover/use type. In this thesis, different aspects of land management and the corresponding land use intensification were studied at landscape, catchment and global level, depending on the objective, environmental context, spatial scale and availability of data. The case of natural rangelands analysed land use intensity within one land use type (i.e. grazing land) (Chapter 5), whereas in the other two cases multiple land cover/use types were studied (Chapters 3 and 4).

The different land use intensities are the actual consequence of varying land management practices. The effect of land management activities on ecosystem properties, functions and services was at the centre of the research, rather than the land management activities themselves. Therefore, this thesis mainly mapped the land cover, land use and ecosystem properties that are shaped by certain management activities and reflect the effect of land management and the intensity of land use. This complexity resulted in inconsistent terminology in some parts of the thesis (e.g. the terms ‘land management components’, ‘land management-related variables’ and ‘(ecosystem) properties’ used as synonyms in Chapter 3).

6.4 Dealing with complexity and data availability

This thesis emphasized the complexity of quantifying the impact of land management activities on ecosystem processes that produce or affect ecosystem services bundles. This complexity was dealt with by applying simplified tools for the analysis of the environmental system. These tools include causal frameworks, indicators, maps, models and scenarios. They made it possible to study land management effects, to quantify ecosystem services and to generate ecosystem service maps from ecosystem properties (or functions). Existing data were combined with model relationships derived for example from statistical relationships obtained from literature and process-based models, to overcome data scarcity. For example, as the InVEST pollination and biodiversity models (Kareiva et al., 2011) are very data-intensive, statistic-based generic distance-relationships were used to quantify and map pollination and biological control.

Uncertainty may arise from the methodological choices (e.g. indicator selection), uncertainties in model parameters (e.g. quantitative effect of grazing on vegetation parameters), data choice, and scenario assumptions (Finnveden, 2000; Janssen et al., 2005; Niemeijer and de Groot, 2008). The sources of uncertainty considered most relevant for the robustness of the results are discussed in each chapter.

6.5 Methodological contribution of this thesis: mapping and modelling methods

This thesis provides suggestions for choosing methods, which are most suitable for the analysis of each of the ecosystem service studied. The choice of methods depends largely on the nature of the ecosystem services and on data availability. The detailed description and analysis of ecosystem service mapping and modelling methods and examples of their prior application are presented in Chapter 1. The present section summarizes the findings about which mapping and modelling method suits which ecosystem services best. According to the methodological grouping of Chapter 1, proxy-based or look-up tables, causal relationships, and biophysical models were the main methods applied in this thesis. In addition, quantitative distance relations derived from statistical models were also applied (Figure 6.2).

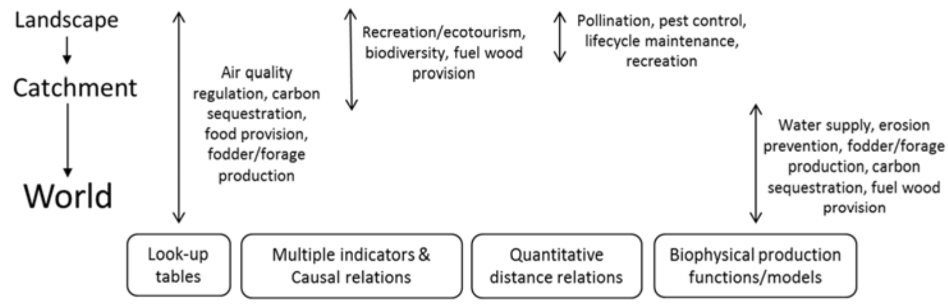


Figure 6.2: Ecosystem service mapping and modelling methods and the levels of spatial scale at which the methods were applied in this thesis. Examples of ecosystem services for each method are given.

(1) *Land use-based proxies* and *look-up tables* were used at all levels, for ecosystem services that can be derived directly from land use types or agricultural statistics (Chapter 3, 4 and 5). Ecosystem services that are measured directly or documented in statistics are mainly provisioning services (Villamagna et al., 2013). Additionally, carbon storage/sequestration is also commonly quantified and mapped with look-up tables (e.g. Kareiva et al.(2011)). In this thesis, look-up tables were used for food provision (livestock and milk), air quality regulation and climate regulation (carbon sequestration) services. Carbon sequestration and provision of raw materials (fodder and fuel wood) can be derived directly from biomass production (Chapters 4 and 5). Carbon sequestration is often derived from satellite image-based information about biomass on the local (Raudsepp-Hearne et al., 2010) and global levels (Kindermann et al., 2006; Schulp et al., 2012). In this thesis, this approach was used at the global level (Chapter 4). Land use-based proxies and look-up tables are simple methods, but are commonly used to obtain a first-order ecosystem service estimate at all level, especially if more detailed information is missing (Egoh et al., 2012; IEEP et al., 2009; Martínez-Harms and Balvanera, 2012).

(2) *Quantitative distance relations*, which are derived from statistical models using distance as the major explaining factor, are suitable to quantify and map landscape pattern-dependent services, such as pollination, pest control and lifecycle maintenance (Chapter 3). These relationships have a low data requirement in comparison to other data-intensive alternatives, such as the InVEST pollination model (Kareiva et al., 2011). Landscape pattern-dependent services are most commonly studied at landscape level, but pollination has also been quantified and mapped at the continental (Schulp et al., 2014) and global levels (Schulp and Alkemade, 2011). At larger scales, the 'distance to nature (or green landscape elements)' relationship can be replaced by the 'density of nature (or green landscape elements)' relationship. Quantitative distance relationships can also be used to map accessibility, which is an important factor for recreation/ecotourism (Chan et al., 2006; Petz and van Oudenhoven, 2012).

(3) *Causal relationships* based on multiple indicators were most suitable for recreation/ecotourism (Chapters 3 and 5). Cultural services largely depend on anthropogenic factors and on the (local) perception of the service (Daniel et al., 2012; Villamagna et al., 2013). The Groene Woud and the Baviaanklsoof Catchment studies underlined this. Indicators that describe recreation/ecotourism, aesthetic information and cognitive development were influenced by the location of the study area and its social context. Accessibility and built infrastructure are important factors for recreation/ecotourism in general (Daniel et al., 2012) and are used as indicators to map the service from the local (Chan et al., 2006) to the global levels (Schulp et al., 2012). The GLOBIO3 model is built upon pre-established relationships (Alkemade et al., 2009) and is

preferred to data-intensive alternatives, such as the InVEST biodiversity model (Kareiva et al., 2011), to map biodiversity when data are scarce. Chapter 4 showed that GLOBIO3 could also give an adequate biodiversity estimate at catchment level.

(4) Quantitative *biophysical models* describing the carbon and water cycle were applied for carbon sequestration at the global level, for water supply at the catchment level and for erosion control at both levels (Chapters 4 and 5). The Universal Soil Loss Equation was applied to quantify and map erosion prevention (Chapters 4 and 5) and a simplified hydrological model was used to quantify and map water supply (Chapter 4) (Kareiva et al., 2011). These models are data-intensive (Vigerstol and Aukema, 2011) and their parameterization for land management, particularly for grazing effects, required much information and many assumptions. Regulating services generally require an extensive knowledge of ecological and hydrological processes (Villamagna et al., 2013). Therefore, biophysical models are the most suitable for quantifying and mapping regulating services (Chapters 3, 4 and 5). At the same time, they are less suitable for studying cultural services (Kremen and Ostfeld, 2005).

6.6 Research findings: land management effects on ecosystem services

Achieving efficient and productive agriculture, while conserving biodiversity and a wide range of ecosystem services is a global challenge (Tscharntke et al., 2012). Many studies have shown that land use causes trade-offs between provisioning and other ecosystem services (Chan et al., 2006; Maes et al., 2012; Raudsepp-Hearne et al., 2010; Schulp et al., 2012). This thesis provides comprehensive quantitative information about ecosystem services for different land use types and levels of land use intensity. A general finding is that agriculture enhances food production but hinders regulating, cultural and habitat services, whereas nature conservation and restoration, and the presence of green landscape elements enhance regulating, cultural and habitat services. This was observed at all scales from landscape to global level (Figure 6.3). Therefore, combining agricultural intensification with nature conservation and restoration would help to provide ecosystem services of all types. This suggests that a multi-functional land use can optimize ecosystem service provision.

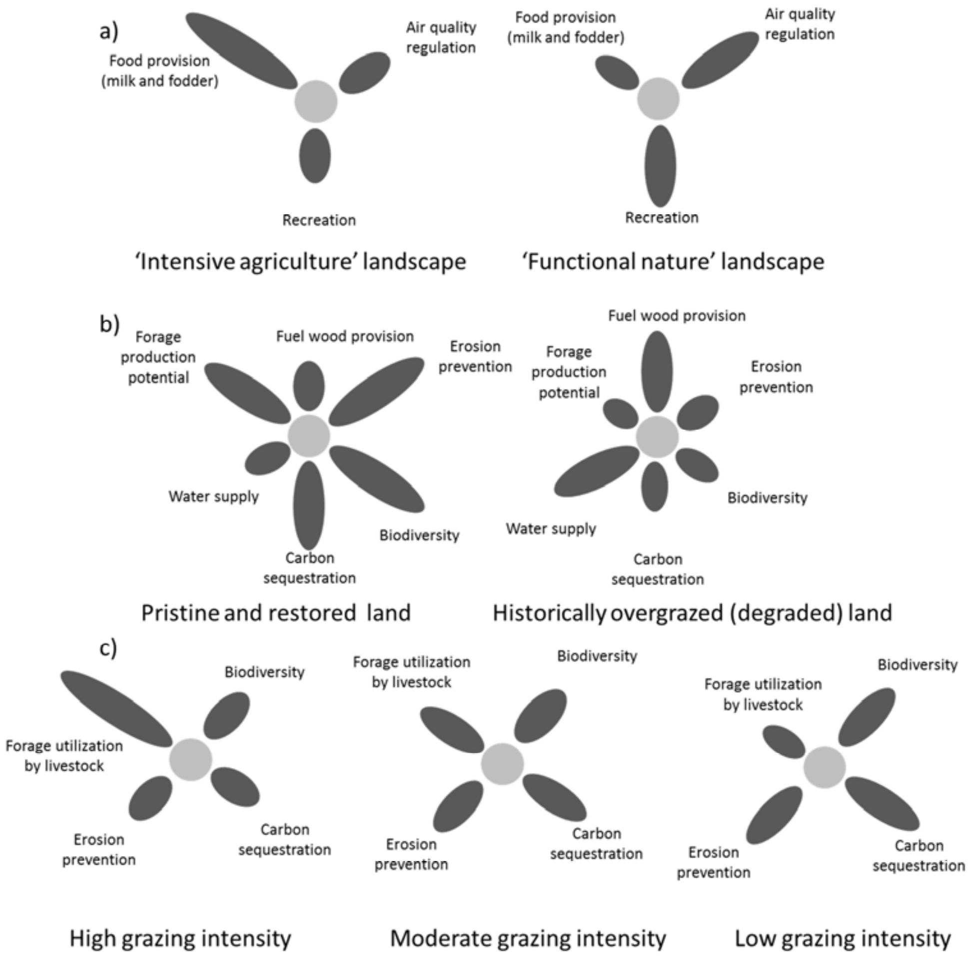


Figure 6.3: Ecosystem service trade-offs for different land use intensities or ecological states: the Groene Woud landscape (a), the Baviaanskloof Catchment (b) and global natural rangelands (c). The trade-offs are based on the quantitative data provided in Chapters 3, 4 and 5.

6.7 Implications for land management and policy

The concept of ecosystem services has great potential to influence environmental planning and decisions, because it links ecosystem processes, components and functions to human interests and needs (Villamagna et al., 2013). This thesis placed the mapping and modelling of ecosystem services in a management and policy context:

1) Improving the connectivity of landscape elements and the landscape heterogeneity and multi-functionality of the Groene Woud Dutch National Landscape (Chapters 2 and 3);
2) Creating a conservation mega-reserve and restoring and implementing sustainable management of the Baviaanskloof Catchment (Chapter 4); and
3) Halting the decline of biodiversity and ecosystem services at global level (Chapter 5).

Involving stakeholders, their interests and views in setting the scope and implementing the findings makes research more relevant and useful for management (Cash et al., 2003; Cowling et al., 2008; Smith et al., 2009). This can be operationalized by establishing a social learning network, such as the Thicket Forum (Smith et al., 2009) and the PRESENCE network (van Eck et al., 2010) in South Africa's Eastern Cape Province. Although the research presented in this thesis did not embed the analysis of ecosystem services in policy and social processes in practice, it provided analysis methods and quantitative information that is relevant for decision-making about land management in the case study areas and beyond. As this research was conducted in collaboration with the PRESENCE network, which facilitates the restoration and sustainable management of the Baviaanskloof Catchment, the research outcomes described in Chapter 4 are useful for the future management of the area. The use of scenarios developed by local stakeholders and the evaluation of ecosystem services against management targets placed the ecosystem service provision in the actual land management context. The study showed that involving multiple stakeholders in the decision-making process results in a land management compromise that emphasizes multi-functionality.

The results of the Groene Woud study confirmed that the green landscape elements play an important role in the provision of multiple ecosystem services. The maintenance and expansion of green landscape elements enhance the provision of biodiversity and a wide range of ecosystem services, a target of the European Union 2020 Biodiversity Strategy (Maes et al., 2012), and landscape multi-functionality, a target of the regional management strategy (Blom-Zandstra et al., 2010). Such landscape- and catchment-level analysis could support local decision-making and spatial planning directly. Nevertheless, both in the Groene Woud and in the Baviaanskloof Catchment, a close and long-term cooperation with local stakeholders would be necessary to implement research findings in practice effectively. This is, however, beyond the scope of this thesis.

Contrary to landscape- and catchment-level analysis, a global analysis provides information about global trends and patterns and may contribute to international science-policy assessments or support international policymaking. The global environmental models IMAGE (PBL, 2006) and GLOBIO3 (Alkemade et al., 2009) were used to provide information about the current state and possible future trends in environmental conditions and biodiversity (UNEP, 2007). These models were applied and further developed in Chapter 5, to locate across the world the natural rangelands where biodiversity and ecosystem services are most impaired by livestock grazing. The case study results can help to prioritize international effort aiming at sustainable rangeland management and biodiversity conservation and the refined model relationships could be applied for future assessments conducted by the Netherlands Environmental Assessment Agency (PBL).

6.8 Recommendations for future research

The research findings and discussion lead to the formulation of recommendations for future research. These recommendations fall into three categories: 1) collection of more extensive spatial data on ecosystem services and standardized information about land management; 2) better inclusion of land management in existing ecosystem service models and 3) better validation of ecosystem service maps and model results.

Ecosystem services are often derived from the interaction of a set of the ecosystem properties, as empirical data on ecosystem services are scarce. Appendix 3 provides a comprehensive overview of the current state of global biophysical and socio-economic data. This overview can be used as a basis for identifying data gaps and prioritize future data collection efforts. Alternatively, data about ecosystem services and land management can be compiled from case studies. An example of this approach is the recent initiative of the PBL and Wageningen UR to develop an ecosystem service database that links ecosystem services to land management by synthetizing information from case studies across the world.

Synthesising the available information into standardized quantitative relationships would enable the robust incorporation of land management effects into ecosystem service mapping and modelling. The effects of grazing on vegetation and hydrological processes, for example, could be included in model relations of the InVEST water supply model (Kareiva et al., 2011). Furthermore, improving the coupling of GIS and system dynamics models could enable better incorporation of land management dynamics, such as the timing of fertilizer application or wood extraction, in the provision of ecosystem services. Although not (yet) commonly used in ecosystem service modelling, the interface between GIS and system dynamics models is technically possible (Costanza and Maxwell, 1991; MA, 2003; Mazzoleni et al., 2003; Mazzoleni et al., 2006).

A final recommendation is to address the methodologies for the validation of ecosystem service maps and model results, and the data used to produce them. Proper validation of global data sets and data products is difficult due to temporal and spatial inconsistencies and differences in classification systems and scaling (Kuemmerle et al., 2013; Verburg et al., 2011b). There are statistical methods to study spatial autocorrelation between different spatial datasets (Hagen-Zanker, 2009; Monserud and Leemans, 1992), but as empirical data are scarce the validation of ecosystem services maps has not received much attention.

6.9 Conclusions

The objective of this thesis was to develop a methodology to quantify the effect of land management on the spatial distribution of ecosystem services in order to determine ecosystem service trade-offs caused by land management across the spatial scale from local ecosystems and landscapes to global biomes. Based on the research presented in this thesis's chapters, the following conclusions can be drawn for each research question:

How can land management and its effects on bundles of ecosystem services be characterized? A systematic framework enables the characterization and measurement of land management and its effect on bundles of ecosystem services. The main contribution of such a framework to ecosystem service mapping and modelling is the delineation of the interactions between the different processes and components, because not all elements of the framework can currently be filled with empirical data.

How can the effect of land management change on ecosystem services be quantified and modelled across the spatial scale when data are limited? Combining multiple mapping and modelling methods and datasets (depending on the nature of the specific ecosystem services and data availability) is an efficient way of quantifying the effect of land management on ecosystem services if sufficient information about land management, land use and the latter's intensity is available.

What is the effect of land management on the spatial distribution of bundles of ecosystems services? In the Groene Woud Dutch landscape, agricultural land provides food (crop and livestock), whereas natural areas, green landscape elements and their vicinity provide regulating, cultural and habitat services.

In the South African Baviaanskloof Catchment, historically overgrazed lands, which are located near the roads and settlements, provide less forage, carbon sequestration, protection against erosion and biodiversity, but more fuel wood and water. They also have a higher potential for ecotourism than pristine or restored lands located further away from the roads and settlements.

Global natural rangelands with a high grazing intensity have higher forage utilization (livestock production), but lower carbon sequestration, capacity for erosion prevention and biodiversity values compared to areas with a low grazing intensity.

Which land management option provides most ecosystem services and meets most policy targets? In the Groene Woud Dutch landscape, the combination of agriculture, nature conservation and maintenance of green landscapes provides most ecosystem services. In the South African Baviaanskloof Catchment, the combination of low input agriculture, nature conservation and restoration provides most ecosystem services and meets most policy targets.

What are the land management-related synergies and trade-offs between ecosystem services? Livestock grazing and forage utilization show a synergy, whereas livestock grazing and carbon sequestration, erosion prevention and biodiversity show trade-offs on the natural rangelands across the world.

What is the effect of changes in land management on bundles of ecosystem services from landscapes to worldwide ecosystems? Quantitative results demonstrate that agricultural land use provides food (i.e. crops and livestock), whereas natural areas, nature conservation and restoration provide regulating, habitat and cultural services. Therefore, as expected, agricultural intensification enhances food production but hinders regulating, cultural and habitat services.

This thesis showed how different mapping and modelling methods assist in the quantification of land management effects on ecosystem services and ultimately in the comparison of alternative land management options. My research also demonstrated how maps and models assist in guiding land management decisions when quantitative and empirical information is limited. I do not suggest a uniform mapping and modelling method but show that methods suitable for the analysis of combined ecosystem services can be developed. A main contribution of the work is that it provides comprehensive quantitative information about land management effects on ecosystem services and quantifies land management-induced ecosystem service trade-offs.

I showed that within one area, several management intensities are possible, resulting in different ecosystem service combinations. Therefore, studying the combination of land use and land management helps to develop a system that provides as many ecosystem services as possible. Based on these findings, I conclude that agricultural intensification leads to enhanced provisioning services, while concurrently decreasing regulating, habitat and cultural services. Therefore, if intensification does occur, combining it with nature conservation and restoration helps to continue providing all types of ecosystem services.

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Glossary

Benefit: the ecological, socio-cultural or economical welfare gain provided through the ecosystem service. Examples are health, employment and income (De Groot et al., 2010a).

Biodiversity: the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems (Convention on Biological Diversity, 1993).

Biomass: the mass of living tissues in either an individual or cumulatively across organisms in a population or ecosystem (MA, 2003).

Biome: a large geographic region, characterized by life forms that develop in response to relatively uniform climatic conditions. Examples are tropical rain forest, savannah, desert, tundra (TEEB, 2010).

Catchment: an area that forms a comprehensive water drainage system, and includes multiple land uses or landscapes (Allan, 2004).

Cultural services: the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, recreation and aesthetic experience (MA, 2003). They include aesthetic, spiritual and psychological benefits (TEEB, 2010).

Demand: the actually demanded or needed use of an ecosystem service by humans (Bastian et al., 2012). It refers to the currently consumed or used ecosystem service (Burkhard et al., 2012) and does not consider where ecosystem services actually are provided. See also ‘Supply’.

Driver: any natural or human-induced factor that directly or indirectly causes a change in an ecosystem (MA, 2003). A direct driver unequivocally influences ecosystem processes and can therefore be identified and measured to differing degrees of accuracy. Examples for direct drivers are land cover and land use changes, introduction of species, climate change and external human inputs (e.g. fertilizer, pesticide). An indirect driver operates by altering the level or rate of change of one or more direct drivers. Examples for indirect drivers are demographic, economic (e.g. globalization) and social (e.g. institutional) changes (MA, 2003).

Ecosystem: a dynamic complex of plant, animal, and microorganism communities and their non-living environment interacting as a functional unit (Convention on

Biological Diversity 1993). Ecosystems are present across the spatial scale, from small patches of, for example, local grasslands to global grassland biomes. The ecosystem concept includes natural systems (e.g. forest) as well as systems strongly modified by humans (e.g. agriculture or urban)(MA, 2005a).

Ecosystem capacity: see 'Ecosystem function'

Ecosystem function: the ecosystem's capacity to provide an ecosystem service (De Groot et al., 2010a). An ecosystem function, or potential (Bastian et al., 2012), is the subset of ecosystem properties, which indicates to what extent an ecosystem service can be provided.

Ecosystem management: an approach to maintain or restore the composition, structure, function and delivery of ecosystem services of natural and modified ecosystems for the goal of achieving sustainability. It is based on an adaptive, collaboratively developed vision of desired future conditions that integrates ecological, socioeconomic, and institutional perspectives, applied within a geographic framework, and defined primarily by natural ecological boundaries (MA, 2005a).

Ecosystem process: an intrinsic ecosystem characteristic whereby an ecosystem maintains its integrity. Ecosystem processes include decomposition, production, nutrient cycling, and fluxes of nutrients and energy (MA, 2005a).

Ecosystem properties: the size, biodiversity, stability, degree of organization, internal exchanges of materials and energy among different pools, and other properties that characterize an ecosystem (MA, 2003). The structures and processes of ecosystems and landscapes in its spatial and temporal variability, e.g. soil properties, biotic material production, nutrient cycles, bio-logical diversity (Bastian et al., 2012). They underpin the capacity of the ecosystem to provide ecosystem services.

Ecosystem services: the direct and indirect contributions of ecosystems to human well-being (TEEB, 2010).

Ecosystem service bundle: sets of ecosystem services that repeatedly appear together across space or time (Raudsepp-Hearne et al., 2010).

Feedback: see 'Negative feedback' and 'Positive feedback'

Geographic information system (GIS): a computerized system organizing data sets through a geographical referencing of all data included in its collections. A GIS allows the spatial display and analysis of information (MA, 2003).

Habitat: an area occupied by and supporting living organisms. Also used to mean the environmental attributes required by a particular species or its ecological niche (MA, 2003).

Habitat services: ecosystems' contribution to provide living spaces for plants or animals and to maintain a diversity of different breeds of plants and animals (De Groot et al., 2010a). They underpin other ecosystem services. See also 'Supporting services'.

Human well-being: see 'Well-being'

Indicator: information used to represent a particular attribute, characteristic, or property of a system (MA, 2003). Environmental indicators are measures of environmental trends and provide a signal of a complex message in a simplified and useful manner (Niemeijer and de Groot, 2008). They allow ecosystem services to be measured and assessed (Petz et al., 2012).

Institutions: the rules that guide how people within societies live, work, and interact with each other. Formal institutions are written or codified rules. Examples of formal institutions would be the constitution, the judiciary laws, the organized market, and property rights. Informal institutions are rules governed by social and behavioural norms of the society, family, or community (MA, 2003).

Land cover: the physical coverage of land, usually expressed in terms of vegetation cover or lack of it (MA, 2003). Land cover addresses the layer of soil and biomass, including natural vegetation, crops and human structures that cover the land surface. Land cover is thus directly observable, both in the field as well as from remote sensing images (Verburg et al., 2009).

Land management: human activities that affect land cover directly or indirectly and aim to provide specific services (this thesis).

Landscape: as a heterogeneous land area composed of a cluster of interacting ecosystems (woods, meadows, marshes, villages etc.) at kilometres wide "human scale" of perception and modification (Forman and Godron, 1986). An area of land that contains a mosaic of ecosystems, including human-dominated ecosystems (MA, 2003).

Land use: the human utilization of a piece of land for a certain purpose, such as irrigated agriculture or recreation (MA, 2003). Land use is determined by the interaction in space and time of biophysical factors (constraints) such as soil, climate, topography etc. and human factors, such as population, technology, economic conditions etc. (Veldkamp and Fresco, 1996). Land use refers to the purposes for which humans exploit the land cover (e.g. grazing or hay production on grasslands) and includes the land management practices (Verburg et al., 2009).

Level: the discrete levels of social organization, such as individuals, households, communities, and nations (MA, 2003). In this thesis, the concept is also applied for ecological organizations.

Management state: the level of land use intensity, which can be expressed by the degree of human input/extraction and naturalness. It ranges from light, extensive (i.e. low intensity) to (very) intensive (i.e. high intensity) management (after Foley (2005), Alkemade et al. (2012; 2009) and de Groot et al. (2010b)). It should be not mistaken with the 'ecological state', which is related to the health and degradation level of the ecosystem.

Mapping: the process of collection and visualization of geospatial data. A map visually represents certain features characteristic for an area (this thesis).

Modelling: the simulation and visualization of biophysical or socio-economic systemic processes by combining certain system elements and parameterizing their behaviour and interactions. How and which elements are combined depends on the purpose of the simulation and visualisation. A model is an abstract and simplified representation of reality used to understand a certain aspect of that reality (this thesis).

Natural capital: any stock of natural resources or environmental assets (such as soil, water, atmosphere, ecosystems) which provide a flow of ecosystem services, now and in the future (De Groot et al., 2003). It is also an economic metaphor for the limited stocks of physical and biological resources found on earth, and of the limited capacity of ecosystems to provide ecosystem services (TEEB, 2010).

Negative feedback: feedback that has a net effect of dampening perturbation (MA, 2005a).

Net primary production: see 'Primary production'

Policy-maker: a person with power to influence or determine policies and practices at an international, national, regional, or local level (MA, 2005a).

Positive feedback: a feedback that has a net effect of amplifying perturbation (MA, 2005a).

Primary production: assimilation (gross) or accumulation (net) of energy and nutrients by green plants and by organisms that use inorganic compounds as food (MA, 2003).

Provisioning services: ecosystem services that describe the material outputs from ecosystems. They include food, water and other resources (TEEB, 2010). The products obtained from ecosystems, including, for example, genetic resources, food and fiber, and fresh water (MA, 2003).

Rangeland: primarily natural grasslands, scrublands, woodlands, wetlands and deserts (Alkemade et al., 2012). It is also an area where the main land use is related to the support of grazing or browsing mammals, such as cattle, sheep, goats, camels, or antelope (MA, 2003).

Regulating services: ecosystem services that ecosystems provide by acting as regulators, such as regulating the quality of air and soil or by providing flood and disease control (TEEB, 2010). The benefits obtained from the regulation of ecosystem processes, including, for example, the regulation of climate, water, and some human diseases (MA, 2003).

Resolution (of observation or modelling): the spatial or temporal separation between observations (MA, 2003).

Responses: the human actions, including policies, strategies, and interventions, to address specific issues, needs, opportunities, or problems. In the context of ecosystem management, responses may be of legal, technical, institutional, economic, and behavioural nature and may operate at local, regional, national, or international level and at various time scales (MA, 2003).

Scale: the physical dimensions, in either space or time, of phenomena or observations (MA, 2003). It is both the limit of resolution where a phenomenon is discernible and the extent that the phenomena is characterised over space and time (White and Running, 1994).

Scenario: a plausible and often simplified description of how the future may develop based on a coherent and internally consistent set of assumptions about key driving forces (e.g. rate of technology change) and relationships. Scenarios are widely used to investigate the effects of socio-economic and environmental changes, and the effects of different policies (MA, 2003).

Spatial resolution: see ‘Resolution’

Supply: the generation of (the actually used) ecosystem services (Burkhard et al., 2012). A certain minimum level or quantity of ecosystem structure and process (including diversity, populations, interactions etc.) is required to maintain a well-functioning ecosystem capable of supplying services (Fisher et al., 2008). The supply does not consider where ecosystem services actually are used or consumed. See also ‘Demand’.

Supporting services: ecosystem services that are necessary for the production of all other ecosystem services. Some examples include biomass production, production of atmospheric oxygen, soil formation and retention, nutrient cycling, water cycling, and provisioning of habitat (MA, 2003). This category is used the

Millennium Ecosystem Assessment (2003) and was (partly) replaced by 'Habitat services' afterwards. See also 'Habitat services'.

Sustainability: a characteristic or state whereby the needs of the present and local population can be met without compromising the ability of future generations or populations in other locations to meet their needs (MA, 2003).

Synergy: when the combined effect of several forces operating is greater than the sum of the separate effects of the forces (MA, 2005a). In this thesis, it is used for a choice that involves gaining more qualities or services (of an ecosystem) simultaneously, as opposed to trade-off. See also 'Trade-off'.

Trade-off: management choice that intentionally or otherwise changes the type, magnitude, and relative mix of services provided by ecosystems (MA, 2005a). It is a choice that involves losing one quality or service (of an ecosystem), in return for gaining another quality or service. Many decisions affecting ecosystems involve trade-offs, sometimes mainly in the long term (TEEB, 2010). See also 'Synergy'.

Value: the contribution of an action or object to user-specified goals, objectives, or conditions (MA, 2003).

Valuation: the process of expressing a value for a particular good or service in a certain context (e.g. of decision-making) usually in terms of something that can be counted, often money, but also through methods and measures from other disciplines, such as sociology, ecology (MA, 2003).

Well-being: a context- and situation-dependent state, comprising basic material for a good life, freedom and choice, health, good social relations, and security (MA, 2003).

Appendix 1

Additional information for chapter 3 – Overview of indicators and relationships used for the spatial modelling of eight ecosystem services.

Milk production

Ecosystem properties. Land use, area need per cow, percentage of milk cows and milk productivity of cows.

Ecosystem function. Number of milk cows = F (land use, area need per cows, rate of milk cows) = grassland area × area need per cow × rate of milk cows.

Ecosystem service. Milk produced (L) = F (number of milk cows, milk productivity of cows) = number of milk cows × milk productivity of cows.

Fodder production

Ecosystem properties. Land use and maize productivity.

Ecosystem function. Area of maize production (ha).

Ecosystem service. Maize produced (kg) = F (area of maize production, maize productivity) = area of maize cultivation × maize yield.

Air quality regulation

Ecosystem properties. Land use, green elements, fine dust capture capacity of vegetation, emission, background concentration, vegetation-atmospheric fine dust concentration relationship and percentage of vegetation cover.

Ecosystem function. Fine dust captured by vegetation (kg/ha/year) = F (land use, green elements, fine dust capture capacity of vegetation, background concentration) = land use/green elements × fine dust capture capacity at given average concentration.

Ecosystem service. Change in atmospheric fine dust concentration (%) = F (% of vegetation cover, vegetation-atmospheric fine dust concentration relationship, emission) = vegetation cover (%) as measure of change in fine dust concentration.

Climate regulation

Ecosystem properties. Land use, green elements, the carbon emission factor and carbon equivalent.

Ecosystem function. Carbon flux (t/ha/year) = F (land use, green elements, carbon emission factor of land use) = land use or green elements × carbon emission factor.

Ecosystem service. Change in atmospheric CO_2 concentration = F (carbon flux, carbon equivalent) = carbon flux × carbon equivalent.

Pollination

Ecosystem properties. Land use, green elements, distance to nature, fruit set distance to nature curve, effective distance and pollinator-dependent crops.

Ecosystem function. Abundance of pollinators (measured by fruit set) (%) = F (land use, green elements, effective distance, fruit set distance to nature curve) = 60.0–0.98× sqrt (distance to forest, heath, natural grass or green landscape elements).

Ecosystem service. Changes in crop yields (%) = F (abundance of pollinators, pollinator-dependent crops) = fruit set at crop areas.

Biological control

Ecosystem properties. Land use, green elements, effective distance and location of pest-influenced crops.

Ecosystem function. Abundance of natural predators (measured by tree density) (%) = F (land use, green elements, effective distance) = area sum of forest and green landscape elements within 1000 m.

Ecosystem service. Changes in crop pest predation (%) = F (abundance of natural predators, pest-influenced crops) = tree density at crop areas.

Lifecycle maintenance

Ecosystem properties. Land use, green elements, species dispersal capacity, habitat fragmentation and nature protection areas.

Ecosystem function. Habitat suitability (%) = F (land use, green elements, species dispersal capacity, fragmentation, nature protection areas) = area sum of forest and green landscape elements within 1750 m × exponential decrease within 1000 meters of road/railway (Exp (-(3.5/1000) × distance from road/railway)) × 20–30% increase due to nature protection.

Ecosystem service. Species occurrence = F (habitat suitability) = habitat suitability > 50%.

Opportunities for recreation

Ecosystem properties. Land use, preference of land use, noise level, proximity to green landscape elements, number of residents and percentage of residents that walk .

Ecosystem function. Walking suitability (%) = F (land use preference, noise level, proximity to green landscape elements) = land use preference for walking × 60–80 % decrease due to noise × 10–30% increase due to green landscape elements.

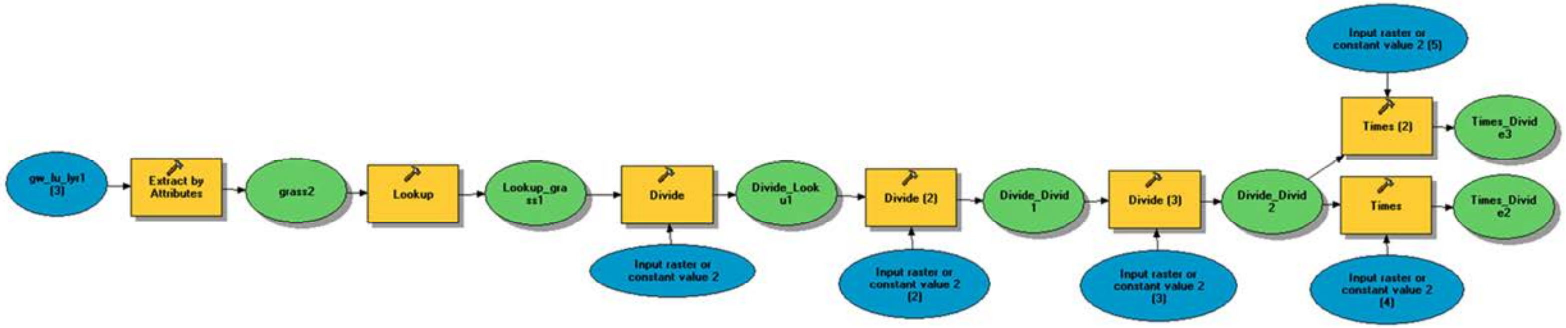
Ecosystem service. Number of walkers (people/ha) = F (walking suitability, % of residents that walk, number of residents) = % of residents that walk × number of residents / walking suitability > 60%.

Appendix 2

Additional information for chapter 3 – Ecosystem function and service models built in ArcGIS 9.3.

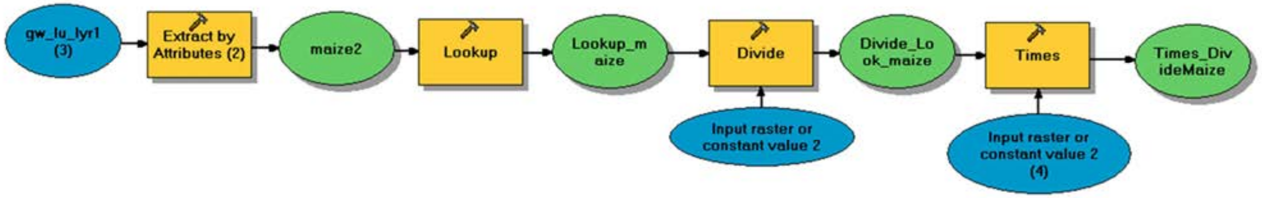
Food production (milk).

Extraction (extract by attributes), reclassification (lookup) and math (divide and times) ArcGIS tools were used.



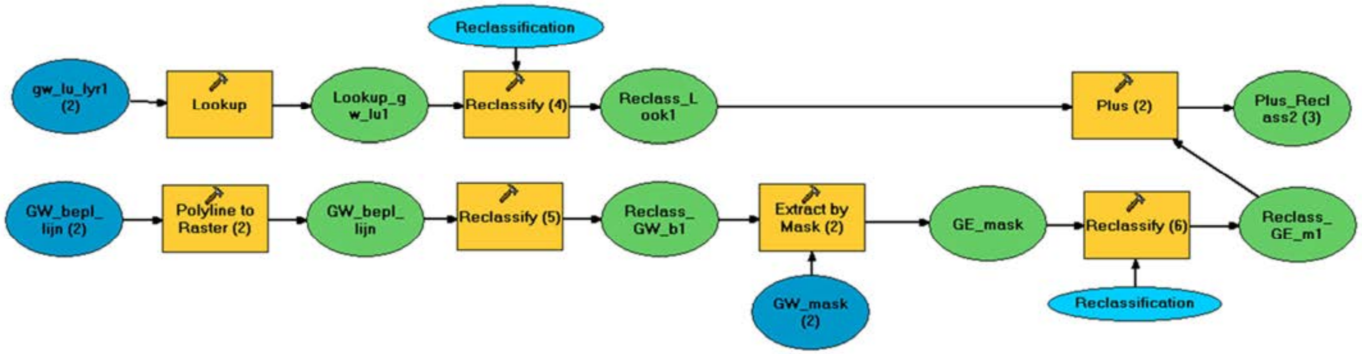
Production of raw materials (fodder).

Extraction (extract by attributes), reclassification (lookup) and math (divide and times) ArcGIS tools were used.



Air quality regulation.

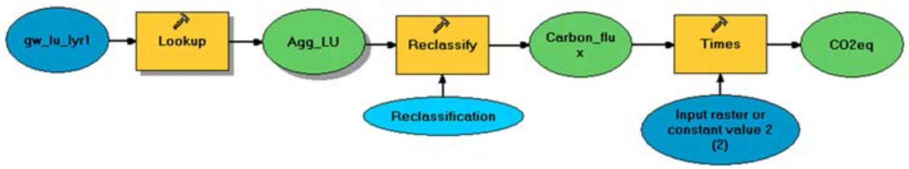
Conversion (polyline to raster), reclassification (lookup and reclassify), extraction (extract by mask) and math (plus) ArcGIS tools were used.



178

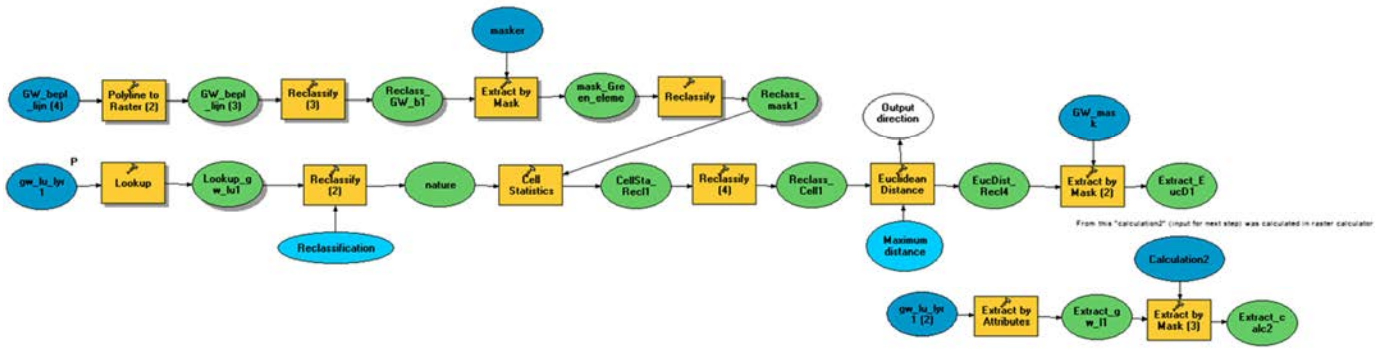
Climate regulation.

Reclassification (lookup and reclassify) and math (times) ArcGIS tools were used.



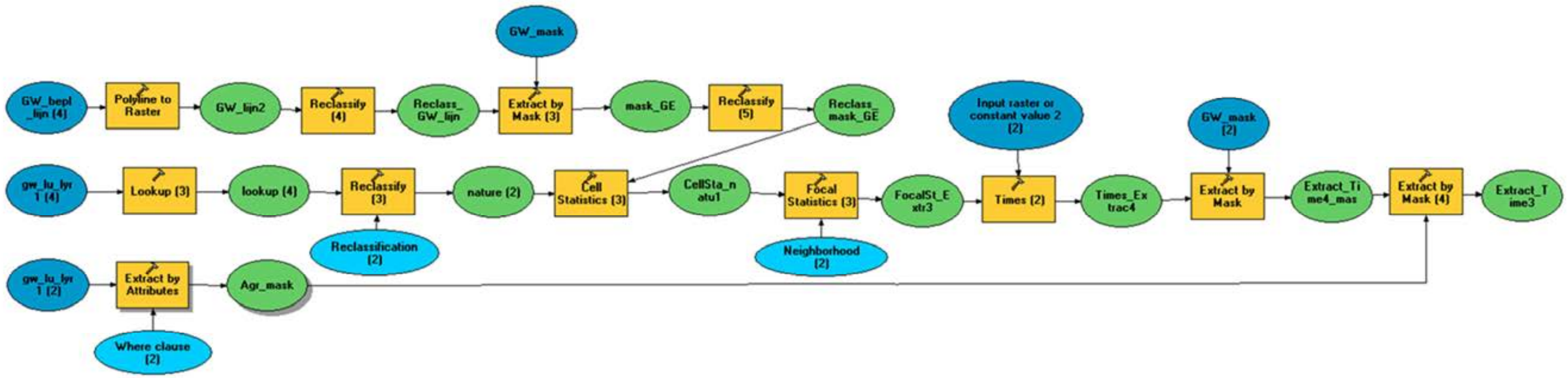
Pollination.

Conversion (polyline to raster), reclassification (lookup and reclassify), extraction (extract by attributes/mask), local statistics (cell statistics), distance (Eucledian distance) and map algebra (raster calculator) ArcGIS tools were used.



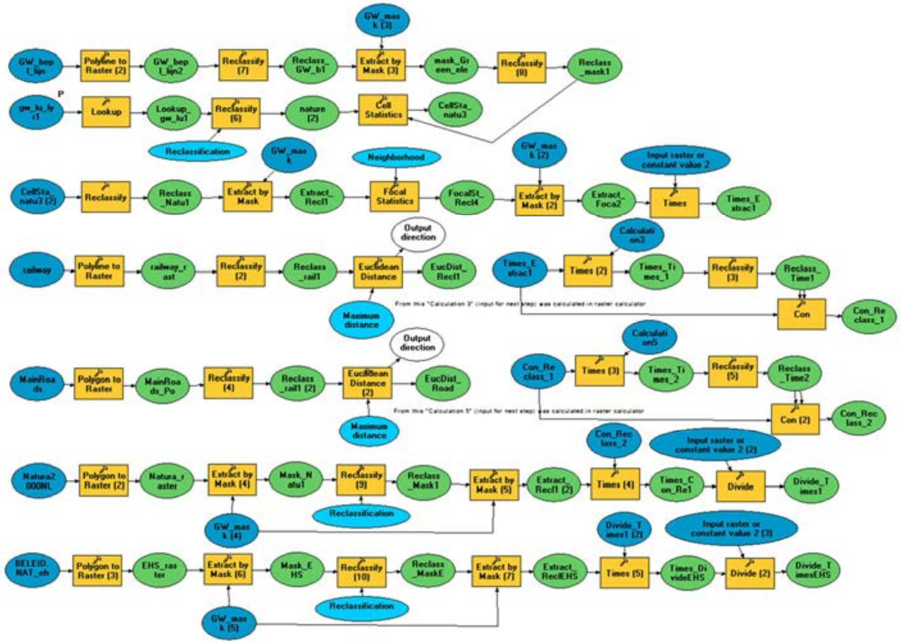
Biological control.

Conversion (polyline to raster), reclassification (lookup and reclassify), extraction (extract by attributes/mask), local statistics (cell statistics), neighbourhood statistics (focal statistic) and math (times) ArcGIS tools were used.



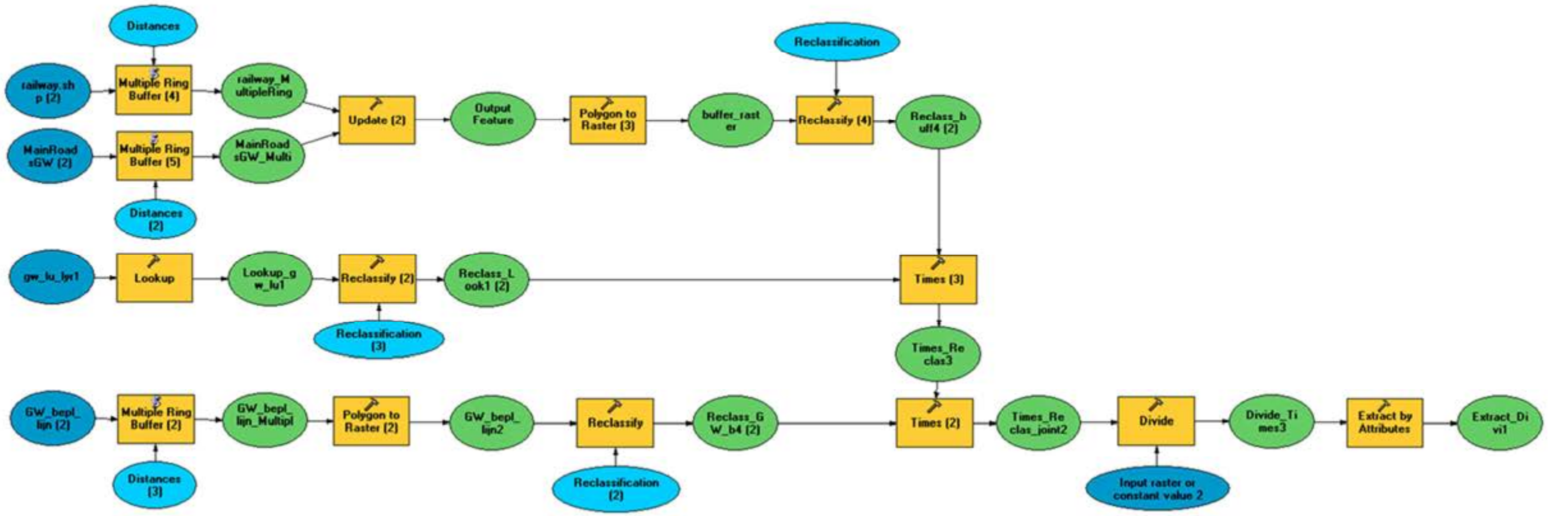
Lifecycle maintenance.

Conversion (polyline/polygon to raster), reclassification (lookup and reclassify), extraction (extract by mask), local statistics (cell statistics), neighbourhood statistics (focal statistic), distance (Eucledian distance), map algebra (raster calculator), math (divide and times) and conditional (con) ArcGIS tools were used.



Opportunities for recreation.

Conversion (polygon to raster), reclassification (lookup and reclassify), extraction (extract by attributes), proximity (multiple ring buffer), overlay (update) and math (times) ArcGIS tools were used.



Notes: The round boxes indicate input or output and the squares indicate processes or tools used in ArcGIS 9.3. We used the ‘analysis’, ‘conversion’ and ‘spatial analyst’ ArcGIS tools. The flowcharts are read from left to right, hence the inputs are on the most left and the final outputs are on most right.

Appendix 3

Additional information for chapter 5 – World coverage spatial data

Methods: Compiling an inventory of global spatial datasets

An inventory of global spatial observational datasets and data products (i.e. based on indicators or modelling) was compiled, in order to assess biophysical and socio-economic data availability and heterogeneity for land management and ecosystems service mapping and modelling. The search program Google Scholar was used to scan scientific literature and to search for websites of international databases relevant for global-level biophysical and socio-economic spatial data. The inventory was compiled based on 1) the state-of-the-art scientific literature that describes and applies global datasets (e.g. Verburg, Neumann et al. (2011b), Schulp, Alkemade et al. (2012), Foley, Ramankutty et al. (2011)), 2) database websites and data download sites of international institutes (e.g. FAO Data Network, UNEP Environmental Data Explorer, JRC, Global Environment Monitoring Unit, NASA Land Processes Distributed Active Archive Center) and 3) the digital database of the International Institute for Applied Systems Analysis (including the GEO-BENE database, Skalský et al., 2008). The inventory is structured according to biophysical, socio-economic and land management features (Figure A3.1) and summarizes the nature, resolution and source (reference and data download route) of spatial datasets and data products.

Characterizing global spatial datasets

In total 76 datasets were collected, almost half of which is biophysical (excluding land management) and the rest is land management-related and socio-economic (excluding land management) (Table A3.1). The datasets vary in resolution, most commonly from 30 arc seconds to 30 arc minutes. Biophysical data (land cover, topography and vegetation characteristics) as well as socio-economic data (population, constructed surfaces, market influence) are often available up to 30 arc seconds resolution.

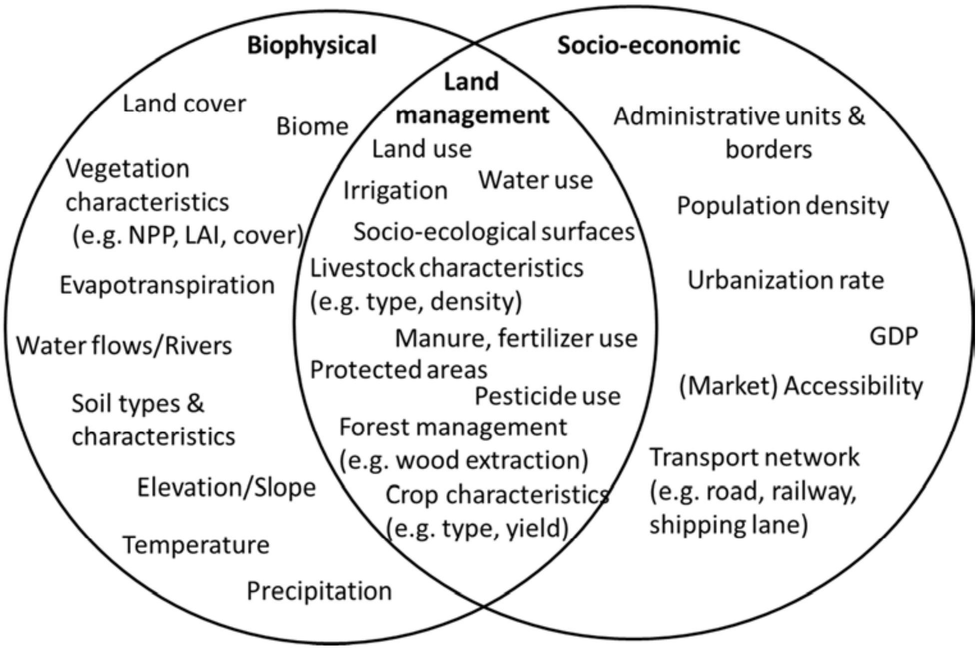


Figure A3.1: Overview of global biophysical and socio-economic spatial data.

A big variety of high-resolution (i.e. higher than 30 arc minutes) global land cover and land use datasets was found, each with a different classification system. High-resolution information of climatic and topographical parameters is provided based on observation and vegetation parameters derived from satellite imagery. Pixel-based data are more abundant to country-based data or shape files, especially for biophysical characteristics (Table A3.1). Information about aspects of land management, such as fertilizer application (5 arc minutes) and livestock production (3 arc minutes), is available at high resolution. Land management-related data are relevant mainly for crop, livestock and wood production. Most of these data are provided at country level or at grid level and are derived from the country-based statics, such as the FAO Gridded Livestock of the World dataset (FAO, 2007). Most of the socio-economic data are provided at country level and as a shape file; pixel-level information is available only of population density, modelled market influence and constructed surfaces (Table A3.2). Depending on the data availability and the nature of the studied feature, different methods were used to create the datasets. These methods include statistics-based (e.g. country-level: water use and wood production; model-based downscaling: land use systems and gridded livestock), satellite-derived (e.g. digital elevation models and land cover maps), model-based and observation-based (e.g. climate data) methods.

Table A3.1: Summary of basic information of the different data characteristics (and the number of datasets under each category).

Data characteristics	Main categories in the inventory
Nature of data	Biophysical (35)
	Land management (27)
	Socio-economic (14)
Type of data	Pixel-based (percentages per grid cell or one class per pixel) (49)
	Country-based (14)
	Shape (11)
Resolution	30 arc-seconds (18)
(for pixel-based data)	5 arc minutes (14)
	30 arc minutes (7)
	15 arc-seconds (2)
	15 arc minutes, 3 arc minutes, 2.5 arc minutes, 10 arc seconds, 8 arc seconds, 3 arc seconds and 1 arc seconds (each 1)

Implication of global spatial datasets

A potential application of the inventory is land management and ecosystem service mapping and modelling. A key input for ecosystem service mapping and modelling is land cover, often derived from satellite imagery (Schulp and Alkemade, 2011; Verburg et al., 2011b). Satellite imagery rarely provides information about the spatial distribution of other land management activities (Verburg et al., 2009). Therefore, this information is often derived from statistics. The inventory provides only limited information about the production and management of crops, livestock and forestry/wood. Nevertheless, these data can be used to derive first-order estimates of the production and use of some ecosystem services. The livestock statistics and biophysical data can be used for the mapping and modelling of livestock production impacts on the environment (FAO, 2007). Furthermore, the protected are maps can be used to identify areas important for eco-tourism (Schulp, 2012). Finally, the hydrological, climatic and topographic data are relevant for mapping and modelling flood regulation and the fertilizer, irrigation and water use data for studying the effect of agricultural production on the environment, among others.

Table A3.2: Inventory of world coverage biophysical and socio-economic spatial datasets and data products. For gridded datasets resolution is given. Shape files are polygons that do not have resolution.

Resolutions:

- 30 arc-minutes (~ 50 x 50 km at the equator, 0.5° x 0.5°)
- 15 arc-minutes (~ 28 km x 28 km at the equator, 0.25° x 0.25°)
- 5 arc-minutes (~ 10 x 10 km at the equator, 0.083° x 0.083°)
- 3 arc-min (~ 5 x 5 km at the equator, 0.05° x 0.05°)
- 2.5 arc-minutes (~ 5 x 5 km at the equator)
- 30 arc-seconds (~ 1 x 1 km at the equator, 0.0083° x 0.0083°)
- 15 arc-seconds (~ 500 x 500 m)
- 10 arc-seconds (~ 300 x 300 m at the equator, 0.0027° x 0.0027°)
- 8 arc-seconds (~ 250 x 250 m at the equator, 0.0022° x 0.0022°)
- 3 arc-second (~ 90 x 90 m at the equator, 0.00083° x 0.00083°)
- 1 arc-second (~ 30 x 30 m at the equator, 0.00027° x 0.00027°)

Category	Specific description	Resolution	Source
Biophysical (Including climatic)			
Land cover	GlobCover 2009 (Global Land Cover Map)	10 arc-sec	(European Space Agency and UCLouvain, 2011) http://due.esrin.esa.int/globcover/
	Global Land Cover (GLC) 2000	30 arc-sec	(JRC, 2003) http://bioval.jrc.ec.europa.eu/products/glc2000/glc2000.php

Category	Specific description	Resolution	Source
	Global cover satellite images (e.g. MODIS and Landsat)	Multiple (MODIS: 15 arc- sec; Landsat:1 arc-sec)	https://lpdaac.usgs.gov/products/modis_products_table (MODIS) http://landsat.gsfc.nasa.gov/ (Landsat)
	Global Lakes and Wetlands Database (GLWD)	30 arc-sec / Shapefile	(Lehner and Doll, 2004) http://gcmd.nasa.gov/records/GCMD_GLWD.html
	World Forest Map 2000	30 arc-sec	GeoNetwork Food and Agriculture Organization of the United Nations, Rome. http://www.fao.org/geonetwork
	Tree cover fraction (MODIS)	15 arc-sec	(Hansen et al., 2003) http://glcf.umd.edu/data/vcf/
	Land cover fraction (MODIS) (barren, closed shrubland, cropland/natural vegetation mosaic, cropland, deciduous broadleaf forest, deciduous needleleaf forest, evergreen broadleaf forest, evergreen needleleaf forest, grassland)	15 arc-min	(Friedl et al., ongoing) http://webmap.ornl.gov/wcsdown/dataset.jsp?ds_id=10011
Temperature	Annual mean temperature, monthly minimum, maximum, mean temperature, temperature seasonality (1950-2000)	30 arc-sec	World Clim – Global Climate Data (Hijmans et al., 2005) http://www.worldclim.org/download
	Daily mean, minimum and maximum temperature, cloud cover (1901-2002)	30 arc-min	Tyndall CRU CL 2.1 data-set (New et al., 2002) http://www.cru.uea.ac.uk/cru/data/hrg/timm/grid/CRU_TS_2_1.html
	Diurnal temperature range, mean temperature, sunshine and wind-speed (1961-1990)	10 arc-min	Tyndall CRU CL 2.0 data-set (New et al., 2002) http://www.cru.uea.ac.uk/~timm/grid/CRU_CL_2_0.html
	MODIS Land surface temperature	30 arc-sec	https://lpdaac.usgs.gov/products/modis_products_table
Precipitation	Annual precipitation, monthly total precipitation, precipitation seasonality (1950-2000)	30 arc-sec	World Clim – Global Climate Data (Hijmans et al., 2005) http://www.worldclim.org/download

Category	Specific description	Resolution	Source
	Number of wet days, frost days, precipitation (1901-2002)	30 arc-min	Tyndall CRU CL 2.1 data-set (Schröter et al., 2005) http://www.cru.uea.ac.uk/cru/data/hrg/timm/grid/CRU_TS_2_1.html
	Wet-day frequency, frost-day frequency, relative humidity (1961-1990)	10 arc-min	Tyndall CRU CL 2.0 data-set (New et al., 2002) http://www.cru.uea.ac.uk/~timm/grid/CRU_CL_2_0.html
Evapotranspiration	Reference evapotranspiration	10 arc-min	(FAO, 2004) http://www.fao.org/geonetwork/srv/en/metadata.show?id=7416
	Land evapotranspiration	30 arc-sec	MODIS http://modis.gsfc.nasa.gov/data/dataprod/dataproducts.php?MOD_NUMBER=16
Biomes	Terrestrial eco-regions	Shape file	(Olson et al., 2001) http://www.worldwildlife.org/science/data/item1875.html
	Terrestrial, freshwater and marine eco-regions	Shape file	Terrestrial: (Olson and Dinerstein, 2002) Freshwater: (Abell et al., 2008) Marine: (Spalding et al., 2007) http://maps.tnc.org/gis_data.html
Soil types and characteristics	World Soil Database WISE	5 arc-min	(Batjes, 2006) http://www.isric.org/data/data-download
	Digital Soil Map of the World and Derived Soil Properties	Shape file	(FAO/UNESCO, 2003) http://www.fao.org/icatalog/search/dett.asp?aries_id=103540
	Harmonized World Soil Database (organic Carbon, pH, water storage capacity, soil depth, cation exchange capacity of the soil and the clay fraction, total exchangeable nutrients, lime and gypsum contents, sodium exchange percentage, salinity, textural class and granulometry)	30 arc-sec	(FAO et al., 2012) http://www.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML/index.html http://www.fao.org/geonetwork/srv/en/metadata.show?id=37140
Elevation/Slope	Global digital elevation model (Gtopo30)	30 arc-sec	USGS Earth Resources Observation and Science (EROS) Center http://eros.usgs.gov/#/Find_Data/Products_and_Data_Available/gtopo30_info

Category	Specific description	Resolution	Source
	Digital Elevation Model (STRM2)	3 arc-sec	Shuttle Radar Topography Mission (SRTM) NASA http://www2.jpl.nasa.gov/srtm/
	Global Terrain Slope and Aspect Data	30 arc-sec	(Fischer et al., 2007) http://www.iiasa.ac.at/Research/LUC/Products-Datasets/global-terrain-slope.html
	Altitude	30 arc-sec	World Clim – Global Climate Data (Hijmans, Cameron et al. 2005) http://www.worldclim.org/download
Vegetation characteristics	Global Forest Biomass: forest growing stock, above/belowground biomass, dead wood, total forest biomass; above-ground/below-ground/dead wood/litter/soil carbon	30 arc-min	(Kindermann et al., 2008) http://www.iiasa.ac.at/Research/FOR/biomass.html
	Gross (GPP) and Net Primary Productivity (NPP) (GPP-autotrophic respiration)	30 arc-sec	MODIS https://lpdaac.usgs.gov/products/modis_products_table
	Leaf Area Index	30 arc-sec	MODIS https://lpdaac.usgs.gov/products/modis_products_table
	Vegetation indices (normalized difference vegetation index (NDVI), enhanced vegetation index (EVI))	~8 arc-sec arc-30 sec	MODIS https://lpdaac.usgs.gov/products/modis_products_table
	Modelled potential Net Primary Productivity	30 arc-min	(Cramer et al., 1999)
	Human appropriation of net primary production (HANNP) 2000: potential/actual/after harvest NPP, land use-induced reduction in NPP,HANPP, HANPP as a percentage of NPP	5 arc-min	(Haberl et al., 2007) http://www.uni-klu.ac.at/socec/inhalt/1191.htm
Water flows/Rivers	Global River Network	Shape file	(ESRI, 1993)
	Hydrologically correct DEM, derived flow directions, flow accumulations, slope, aspect, and a compound topographic (wetness) index	30 arc-sec	HYDRO1K Drainage Basins dataset, USGS Earth Resources Observation and Science (EROS) Center http://eros.usgs.gov/#/Find_Data/Products_and_Data_Available/gtopo30/README

Category	Specific description	Resolution	Source
	Major river basins and rivers of the World (GRDC, 2007)	Shape file	(Global Runoff Data Centre, 2007) http://www.bafg.de/nn_267044/GRDC/EN/02__Services/02__DataProducts/MajorRiverBasins/riverbasins__node.html?__nnn=true
Threatened species richness	Threatened amphibian, mammal, coral, reptile, bird, fish, mangrove, seagrass richness	Shape file	(International Union for Conservation of Nature, 2012) http://www.iucnredlist.org/
Land management			
Socio-ecological surfaces	Land systems	5 arc-min	(van Asselen and Verburg, 2012)
	Land use systems	5 arc- min	(Letourneau et al., 2012)
	Anthropogenic Biomes: Anthromes (v2) maps in GIS formats for AD 1700 to 2000	5 arc-min	(Ellis et al., 2010) http://ecotope.org/anthromes/v1/guide/
	Combined biome, biodiversity and anthrome maps	5 arc-min	(Ellis et al., 2012) http://ecotope.org/anthromes/v1/guide/
Land use	Land use areas	Country	(FAO) http://faostat.fao.org/site/377/default.aspx#ancor
	Global land use 2000: fraction of infrstursture/cropland/grazingland/forest/non-productive areas/grazing suitability	5 arc-min	(Erb et al., 2007) http://www.uni-klu.ac.at/socec/inhalt/1189.htm
	Global distribution of croplands and pastures (2000)	5 arc-min	(Ramankutty et al., 2008)
	Global distribution of crop areas, types and net primary productions (2000)	5 arc- min	(Monfreda et al., 2008)
	Cropland and grasslands (1990-2000)	5 arc-min	(Goldewijk et al., 2007)
	Cropland, built-up land, grazing land, wetlands, irrigated land, inundated land (1990-2000)	5 arc-min	(Sterling and Ducharne, 2008)
	Land use and land cover	5 arc-min	(Fischer et al., 2008) http://www.iiasa.ac.at/Research/LUC/External-World-soil-

Category	Specific description	Resolution	Source
	(rain-fed and irrigated cultivated land, forest, rass/scrub/woodland, residential and infrastructure built-up, barren land, water bodies)		database/HTML/LandUseShares.html?sb=9n
Irrigation	Digital Global Map of Irrigated Areas (GMIA) version 4.0 (2000)	30 arc-min	(Siebert et al., 2005) http://www.geo.uni-frankfurt.de/ipg/ag/dl/forschung/Global_Irrigation_Map/index.html
	Global data set of monthly irrigated and rainfed crop areas around the year 2000 (1998-2002) (MIRCA2000)	5 arc-min	(Portmann et al., 2010)http://www.geo.uni-frankfurt.de/ipg/ag/dl/forschung/MIRCA/
	Irrigated crop are and drainage	Country	(FAO, 2012a) http://www.fao.org/nr/water/aquastat/main/index.stm
Water use	Water resources and withdrawal	Country	(FAO, 2012a) http://www.fao.org/nr/water/aquastat/main/index.stm
Manure, fertilizer and pesticide use	Fertilizer (N, K, P) production, trade and consumption; and pesticide use and trade	Country	(FAO) http://faostat.fao.org/site/575/default.aspx#ancor
	Fertilizer use by crop (2002-2006)	Country	(FAO, 2006)
	Fertilizer (N, K, P) production, trade and consumption	Country	International Fertilizer Industry Association, http://www.fertilizer.org/ifa/HomePage/STATISTICS
	Global N and P fertilizer and manure application rates	30 arc-min	(Potter et al., 2010) http://sedac.ciesin.columbia.edu/data/collection/ferman-v1
	P balances (input, output) for cropland for year 2000	30 arc-min	(MacDonald et al., 2011)
	N balances (input, output) for cropland for year 2000	5 arc-min	(Liu et al., 2010)
Livestock characteristics	Animal livestock types, numbers and processed	Country	(FAO) http://faostat.fao.org/site/569/default.aspx#ancor
	The Gridded Livestock of the World (GLW): Livestock densities, Livestock production systems and Supply and Demand	3 arc-min	(FAO, 2007) http://www.fao.org/AG/againfo/resources/en/glw/GLW_dens.html

Category	Specific description	Resolution	Source
Crop characteristics	Crop types, harvested areas, yields and processed	Country	(FAO) http://faostat.fao.org/site/567/default.aspx#ancor
	Agro-MAPS : Global Spatial Database of Agricultural Land-use Statistics	Sub-national and country	http://www.fao.org/landandwater/agll/agromaps/interactive/page.jspx
Forest management	Production and imports and exports of woods and paper	Country	(FAO) http://faostat.fao.org/site/630/default.aspx
Protected areas	World Database on Protected Areas (WDPA)	Sub-national shape file	United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC) and the International Union for Conservation of Nature, World Commission on Protected Areas (IUCN - WCPA) http://www.wdpa.org/ http://protectedplanet.net/#9_48_14.25_0 http://geodata.grid.unep.ch/
Socio-economic			
Administrative units	Global Administrative Unit Layers (GAUL)	Shape file	(FAO, 2009) http://www.fao.org/geonetwork/srv/en/metadata.show?id=12691
Borders	Land outlines and political boundaries	Shape file	CIA World DataBank II (Gorny and Carter, 1987) http://www.evl.uic.edu/pape/data/WDB/
Population and urbanization	Percentage of urban/rural population and population of major agglomerations	Country	UNEP major urban agglomeration and population database http://geodata.grid.unep.ch/ ((United Nations, 2012) http://esa.un.org/unpd/wup/Maps/maps_urban_2011.htm)
	Population: urban/rural, male/female	Country	(FAO) http://faostat.fao.org/site/550/default.aspx#ancor
	Global population distribution data	30 arc-sec	(LandScan) http://www.ornl.gov/sci/landscan/landscan_documentation.shtml
	Gridded Population of the World (People/sq km)	2.5 min (national and sub-	(Center for International Earth Science Information Network (CIESIN) Columbia University; and Centro Internacional de Agricultura Tropical (CIAT), 2005)

Category	Specific description	Resolution	Source
		national input units)	http://sedac.ciesin.columbia.edu/gpw/
Transport network	Roads location, type of roads and road density	Multiple scale (sub-national-global)	Global Road Inventory Project (GRIP), PBLhttp://geoservice.pbl.nl/website/flexviewer/index.html?config=cfg/PBL_GRIP.xml&center=5.2,52.1333&scale=5000000
	Global Distribution and Density of Constructed Impervious Surfaces	30 arc-sec	(Elvidge et al., 2007) http://www.ngdc.noaa.gov/dmsp/download_global_isa.html
	Major road, rail networks, hydrologic drainage systems, utility networks (cross-country pipelines and communication lines), major airports, elevation contours, coastlines, international boundaries and populated places	Shape file	http://www.mapability.com/index1.html?http&&&www.mapability.com/info/vmap0_download.html
	Navigable rivers	Shape file	CIA World DataBank II (Gorny and Carter, 1987) http://www.evl.uic.edu/pape/data/WDB/
	Shipping lanes	---	(Halpern et al., 2008) http://www.nceas.ucsb.edu/GlobalMarine/impacts
Gross Domestic Product (GDP)	GDP (US$) and GDP per capita (US$)	Country	The World Bank data.worldbank.org
	GDP (US$) and GDP per capita (US$)	Country	CIA World factbook https://www.cia.gov/library/publications/the-world-factbook/index.html
Market influence	Market accessibility index, market influence index and market influence density index	30 arc-sec	(Verburg et al., 2011a)

Hungarian grey cattle

Summary

Ecosystems provide numerous benefits to people. These benefits are called ecosystem services. Ecosystem services include food, fresh water, fertile soils, timber, medicines and recreation opportunities. In order to meet increasing human needs, ecosystems have been modified to deliver many ecosystem services. Unsustainable use depletes and degrades biodiversity and ecosystem services. In response to this, ecosystem restoration, and sustainable management of ecosystem services and biodiversity are being incorporated in national and international policies. Land management refers to human activities that affect land cover directly or indirectly and aim to provide specific ecosystem services. It defines land use and the intensity of use driven by human activities. Changes in land management alter the composition of ecosystem services, and maximize one or a limited set of services at the cost of others. This creates trade-offs between the supply of different ecosystem services. For example, intensive agriculture or forestry maximizes crop or timber production, whereas conservation management supports a wide range of ecosystem services, including water regulation, provision of habitat for wildlife and opportunities for recreation. To minimize land management-induced ecosystem service trade-offs and support decision-making with regard to land management, a better understanding of ecosystem service provision and the quantification of multiple ecosystem services are necessary.

The objective of this thesis is therefore to develop a methodology to quantify the effect of land management on the spatial distribution of ecosystem services in order to determine ecosystem service trade-offs caused by land management across the spatial scale from local ecosystems and landscapes to global biomes. A wide range of ecosystem services (also called ecosystem service bundles) is studied. The existing, but scattered information about the dependencies between land use, land management, ecosystem properties and ecosystem service provision is integrated using maps and models. Maps and models are useful tools to synthesize information, quantify and visualize ecosystem services and to communicate this information to decision makers.

Some existing mapping and modelling tools are used (Chapter 1) and some new quantitative relationships between land management and the provision of ecosystem services are developed and applied to map and model ecosystem services in a GIS environment. Because of their complex character, ecosystem services are assessed with the help of indicators. A comprehensive but generic framework is developed to support indicator selection, quantification, mapping and modelling (Chapter 2). Three cases,

ranging from landscape to global levels, are studied. First, ecosystem services are quantified and modelled for the Dutch landscape Groene Woud (Chapter 3). Secondly, land management options are evaluated for ecosystem services and related management targets in the South African Baviaanskloof Catchment (Chapter 4). Finally, land management-related ecosystem service synergies and trade-offs are identified for natural rangelands across the world (Chapter 5).

In the Dutch and South African case studies, the mapping and modelling tools are applied in combination with scenario analysis. Scenarios are used to compare different land management options because of their applicability for mapping and involvement of stakeholders' visions. In the Dutch case, a scenario analysis demonstrates the expected effect of different levels of land use intensity on ecosystem services. Results show that agriculture mainly provides provisioning services (i.e. food), whereas natural areas, green landscape elements and their vicinity provide regulating, habitat and cultural services (Chapter 3). In the South African case, results show that a compromise between (extensive) agriculture, restoration and conservation is the best for the provision of multiple ecosystem services. This type of land management also meets most of the management targets and is in line with stakeholders' visions for future land management (Chapter 4). The last case study shows that natural rangelands with the highest grazing intensity emit most carbon, have the lowest capacity for erosion prevention and hold the lowest biodiversity. These areas are found in the Sahel, West India, Pakistan, Middle East, Northern Africa and some parts of Brazil (Chapter 5).

To conclude, the methodological contributions, the main findings and their relevance for land management and policies are discussed (Chapter 6). The development of mapping and modelling methods are central to this thesis research. My research demonstrates that land management and its effect on bundles of ecosystem services can be characterized and measured with a systematic framework. The main contribution of such stepwise framework to ecosystem service mapping and modelling is the delineation of the logic and interactions between the different processes and components, especially when empirical data are scarce. I do not suggest a uniform mapping and modelling method but show that methods suitable for the analysis of combined ecosystem services can be developed. The choice of methods depends on the nature of the ecosystem service and data availability. In addition, I also demonstrate how these methods can be used to assess and evaluate land management effects on ecosystem services when data are scarce. Within one area, several management intensities are possible, resulting in different ecosystem service combinations. Therefore, studying the combination of land use and land management helps to develop a system that provides as many ecosystem services as possible. Based on these findings, I conclude that agricultural intensification leads to enhanced provisioning services, while concurrently decreasing regulating, habitat and cultural services. Therefore, if intensification does occur, combining this with nature conservation and restoration helps to continue providing all types of ecosystem services. My research therefore provides comprehensive quantitative information about land management effects on ecosystem services and quantifies land management-induced ecosystem service trade-offs.

Samenvatting

Ecosystemen leveren baten op voor de mens. Dit zijn de zogenaamde ecosysteemdiensten. Deze diensten omvatten o.a. voedsel, drinkwater, bodemvruchtbaarheid, hout, medicijnen en recreatiemogelijkheden. Om aan de vraag aan ecosysteemdiensten te voldoen zijn veel ecosystemen door de mens sterk beïnvloedt en veranderd. Onduurzaam gebruik zorgt voor uitputting en degradatie van biodiversiteit en ecosysteemdiensten. Herstel en duurzaam beheer van biodiversiteit en ecosysteemdiensten worden daarom steeds meer geïntegreerd in nationaal en internationaal beleid. Landmanagement verwijst naar die menselijke activiteiten die specifieke ecosysteemdiensten leveren en dus ecosystemen direct of indirect beïnvloeden. Landmanagement bepaalt het landgebruik en de intensiteit. Veranderingen in landmanagement beinvloeden de samenstelling van ecosysteemdiensten en maximaliseert meestal één dienst of een beperkte set van diensten. Deze trade-offs (uitruil) gaat vaak ten koste van enkele ecosysteemdiensten. Bijvoorbeeld, intensieve landbouw maximaliseert gewasopbrengst, terwijl een natuurlijk beheer een breed scala van ecosysteemdiensten voortbrengt, zoals waterregulering, biodiversiteit en recreatiemogelijkheden. Om deze trade-offs tussen ecosysteemdiensten te minimaliseren en landmanagement goed te ondersteunen, moet de kwantificering van ecosysteemdiensten verbeterd worden.

In dit proefschrift wordt een methodologie ontwikkeld voor de kwantificering van effecten van landmanagement op de ruimtelijke verspreiding van ecosysteemdiensten, zodat de door landmanagement veroorzaakte trade-offs tussen ecosysteemdiensten bepaald kunnen worden voor zowel lokale ecosystemen en landschappen als regionale en mondiale biomen. Een groot aantal ecosysteemdiensten zijn bestudeerd. De bestaande, maar gefragmenteerd beschikbare informatie over afhankelijkheid tussen landgebruik, landmanagement, ecosysteemeigenschappen en -functioneren, en ecosysteemdiensten is geïntegreerd met behulp van kaarten en modellen. Dit zijn nuttige hulpmiddelen om ecosysteemdiensten niet alleen te synthetiseren, visualiseren en kwantificeren, maar ook te communiceren aan managers, gebruikers en beleidsmakers.

Verschillende bestaande kartering- en modelleringsmethoden zijn toegepast (Hoofdstuk 1) en een aantal nieuwe kwantitatieve relaties tussen landmanagement en de levering van ecosysteemdiensten zijn ontwikkeld en toegepast voor karteren en modelleren in een GIS omgeving. Vanwege hun complexiteit zijn ecosysteemdiensten beschreven met behulp van indicatoren. Een uitgebreid, maar generiek raamwerk is ontwikkeld om het selecteren, kwantificeren, karteren en modelleren van indicatoren te ondersteunen (Hoofdstuk 2). Drie case studies, variërend van landschap tot biome, zijn bestudeerd. Eerst zijn ecosysteemdiensten gekwantificeerd en gemodelleerd voor het Nederlandse landschap 'Het Groene Woud' (Hoofdstuk 3). Daarna zijn landmanagement opties geëvalueerd voor ecosysteemdiensten en gerelateerde beheerdoelstellingen in het stroomgebied van Baviaanskloof in Zuid-Afrika (Hoofdstuk 4). Tenslotte zijn synergiën en trade-offs tussen verschillende ecosysteemdiensten geïdentificeerd voor natuurlijke weidegronden wereldwijd (Hoofdstuk 5).

De karterings- en modelleringsmethoden zijn toegepast en gecombineerd met scenario-analyse in de Nederlandse en Zuid-Afrikaanse studies. Scenario's zijn gebruikt om verschillende landmanagement opties te vergelijken. De visies van belanghebbenden kon op deze manier met behulp van scenariobeelden in de kaarten weergegeven worden. In de Nederlandse studie, toont de scenario-analyse het verwachte effect van de verschillende managementintensiteiten van landgebruik op ecosysteemdiensten aan. Resultaten laten zien dat landbouw voornamelijk productie-diensten (d.w.z. voedsel) levert, terwijl natuurgebieden, groene landschapselementen en hun omgeving regulerende, habitat en culturele diensten leveren (Hoofdstuk 3). In de Zuid-Afrikaanse studie, laten de resultaten zien dat een compromis tussen (extensieve) landbouw, natuurherstel en natuurbehoud de levering van meerdere ecosysteemdiensten stimuleert. Deze vorm van landmanagement ondersteunt ook de meeste beheerdoelstellingen en is in lijn met de toekomstige visies van belanghebbenden (Hoofdstuk 4). De mondiale studie laat zien dat natuurlijke weidegronden met de hoogste begrazingsintensiteit de meeste koolstof uitstoten, en de laagste capaciteit voor erosiepreventie en minder biodiversiteit bevatten. Deze gebieden liggen in de Sahel, Westelijk India, Pakistan, het Midden-Oosten, Noord-Afrika en in sommige delen van Brazilië (Hoofdstuk 5).

Tot slot zijn de methodologische bijdragen, de belangrijkste bevindingen en hun relevantie voor ruimtelijke ordening en beleid bediscussieerd (Hoofdstuk 6). De ontwikkeling van karterings- en modelleringsmethoden staat centraal in dit proefschrift. Mijn onderzoek toont aan dat landmanagement en het effect ervan op ecosysteemdiensten kan worden gekarakteriseerd en gemeten met een systematische benadering. Mijn belangrijkste bijdrage aan kartering en het modellering van ecosysteemdiensten is de karakterisering van de onderliggende logica en interacties tussen de verschillende processen en componenten, en dan met name wanneer empirische gegevens schaars zijn. Ik raad geen uniforme karterings- en modelleringsmethode aan, maar laat zien dat methoden die geschikt zijn voor de analyse van gecombineerde ecosysteemdiensten, goed gebruikt kunnen worden.

De keuze van de methode is echter afhankelijk van de aard van de ecosysteemdiensten en beschikbaarheid van gegevens. Ik laat zien hoe deze methoden effectief kunnen worden gebruikt om de effecten van landmanagement op ecosysteemdiensten te analyseren als gegevens schaars zijn. Binnen ééngebied zijn meerdere intensiteiten van landgebruik mogelijk, die tot verschillende combinaties van ecosysteemdiensten leiden. Het bestuderen van deze combinatie helpt dus om een strategie te ontwikkelen die zoveel mogelijk ecosysteemdiensten levert. Op basis van deze bevindingen, concludeer ik dat intensivering van de landbouw wel leidt tot de beoogde verhoogde productiviteit, maar tegelijkertijd de regulerende, habitat en culturele diensten vermindert. Het gelijktijdig plannen en implementeren van intensivering in combinatie met natuurbehoud en natuurherstel kan daarom bijdragen aan het leveren van vele verschillende ecosysteemdiensten. Mijn onderzoek geeft uitgebreide kwantitatieve informatie over de effecten van landmanagement op ecosysteemdiensten en kwantificeert de door de landmanagement geïnduceerde synergiën en trade-offs van ecosysteemdiensten.

The Lumen building at Wageningen Univesity

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Curriculum Vitae

Katalin Petz was born on the 15th of June 1983 in Budapest, Hungary. During her childhood, she lived several years abroad with her family, in Germany, Belgium, Italy and Japan. She developed an interest for nature and environment in these early years. After finishing the secondary school in 2001, Katalin went to study horticultural engineering at the Corvinus University of Budapest. She spent a half year at the Swedish University of Agricultural Sciences (SLU) in Uppsala, where she followed her first integrated environmental assessment course. After this, Katalin went to study MSc Environmental Sciences at the Wageningen University, The Netherlands. She specialized herself on environmental systems analysis. She studied ecosystem services in the Hungarian Tisza River Basin as part of her thesis and conducted an internship at the Potsdam Institute for Climate Impact Research (PIK) in Germany. During her internship, she implemented the Water Evaluation and Planning System (WEAP) model and the concept of ecosystem services for the Upper-Tana River Basin in Kenya. After completing her studies, Katalin worked as conservation volunteer in New Zealand and as environmental volunteer at the Jane Goodall Institute in Budapest. She also worked as project assistant at the Research Institute for Solid State Physics and Optics in Budapest for a short period. In September 2009 she started her PhD research at the Environmental Systems Analysis group of Wageningen University in collaboration with the Netherlands Environmental Assessment Agency. She finished her dissertation in November 2013. As environmental scientist, Katalin is interested in human-environment interactions and complex environmental issues. She is specifically interested in sustainable landscape and natural resources management and science-policy-practice interface from national to European and international levels.

Publications

K. Petz, R. Alkemade, M. Bakkenes, C.J.E. Schulp, M. van der Velde, R. Leemans (submitted) Mapping and modelling trade-offs and synergies between grazing intensity and ecosystem services on rangelands using global-scale datasets and models

K. Petz, J. Glenday, R. Alkemade (submitted) Land management implications for ecosystem services provision in a South African rangeland

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SENSE Education Certificate

Netherlands Research School for the
Socio-Economic and Natural Sciences of the Environment

C E R T I F I C A T E

The Netherlands Research School for the
Socio-Economic and Natural Sciences of the Environment
(SENSE), declares that

Katalin Petz

born on 15 June 1983 in Budapest, Hungary

has successfully fulfilled all requirements of the
Educational Programme of SENSE.

Wageningen, 12 March 2014

the Chairman of the SENSE board	the SENSE Director of Education
Prof. dr. Rik Leemans	Dr. Ad van Dommelen

The SENSE Research School has been accredited by the Royal Netherlands Academy of Arts and Sciences (KNAW)

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The SENSE Research School declares that **Ms. Katalin Petz** has successfully fulfilled all requirements of the Educational PhD Programme of SENSE with a work load of 62 ECTS, including the following activities:

SENSE PhD Courses

- Environmental Research in Context
- Research Context Activity: Co-organizing 4th Ecosystem Services Partnership (ESP) conference on "Ecosystem Service: Integrating Science and Practice" (Wageningen, 2011)
- The Art of Modelling

Other PhD and Advanced MSc Courses

- Spatial modelling and Statistics
- Techniques for Writing and Presenting Scientific Papers
- Mobilising your Scientific Network course
- Teaching and supervising MSc thesis students
- GISLERS Summer School
- Alter-NET Summer School
- Fuzzy-modelling course

Management and Didactic Skills Training

- Teaching and supervising three MSc thesis students
- Teaching, lecturing and organizing GIS practicals at the Regional Management Course
- Co-organisation of SENSE PhD day: Meet & Greet Research Cluster XIII – Land use, spatial analysis & modelling / Ecosystem and Landscape Services (28 May 2010, Wageningen)

Oral Presentations

- *Modelling the Effect of Land Management Decisions on Ecosystem Services.* 3rd International ESP Conference "Solutions for Sustaining Natural Capital and Ecosystem Services", 7-11 June 2010, Salzau, Germany
- *Modelling the impact of land use management on ecosystem services.* WIMEK SENSE symposium "Modelling and observing earth system compartments", 22 February 2011, Wageningen, Netherlands
- *Spatial modelling of land management, ecosystem functions and services at landscape scale: a case study of the Groene Woud.* The Netherlands, 4th International ESP Conference "Ecosystem Services: Integrating Science and Practice", 4-7 October 2011, Wageningen, Netherlands
- *Modelling the effect of land management on the provision of ecosystem services.* Frontiers of Ecosystem Service Science Colloquium for Young Scientists 2012, 7 February 2012, Stellenbosch, South Africa
- *Land management in ecosystem services modelling.* IIASA YSSP Late Summer Workshop, 24 August 2012, Laxenburg, Austria
- *Biomass production and land management.* SENSE Science Market: Towards a Biobased Economy, 25 October 2012, Hague, Netherlands

SENSE Coordinator PhD Education

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