QUANTIFYING THE EFFECTS OF MANAGEMENT ON ECOSYSTEM SERVICES

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Thesis committee

Promotor
Prof. Dr H.B.J. Leemans
Professor of Environmental Systems Analysis
Wageningen University

Co-promotors
Dr R.S. de Groot
Associate professor, Environmental Systems Analysis Group
Wageningen University

Dr J.R.M. Alkemade
Senior Researcher
PBL - Netherlands Environmental Assessment Agency, Bilthoven

Other members
Prof. Dr P.F.M. Opdam, Wageningen University
Prof. Dr P.H. Verburg, VU University Amsterdam
Dr J. Maes – European Commission - Joint Research Centre, Ispra, Italy
Dr B. Burkhard – University of Kiel, Germany

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With the publication of this PhD thesis, a journey comes to an end during which I asked a lot of my supervisors, colleagues, friends, family and myself. However, the journey could have ended quite differently. Just two week after submitting this thesis, I was hit by a car while cycling back home. For several hours I had no notion of time and no recollection of recent events. With the help of very friendly ambulance staff, I could slowly put together the pieces of the puzzle. I was utterly relieved to find I could still remember what my thesis was about; the rest followed soon afterwards. The event and the reactions that followed reminded me to not take memories for granted and to be forever grateful to the organisations and people that have contributed to these memories.

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My PhD journey took me to faraway places that quickly became a home to me, even though I could only spend limited time on location. When in Wageningen, my thoughts were always with the people working in the Eastern Cape (South Africa) and Java (Indonesia). Both projects enriched my research and personal life, and provided my journey with context, purpose and motivation.

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and learn from you. I could always count on Ita Sualia and Etwin Sabarini, who dealt with our questions and requests wonderfully well. Eka Damastuti's smiles, courage and determination were very important to me and the team. Thanks also to Mark Spalding and Anna McIvor for your feedback, both during the project's initial phase and whenever I had questions.

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Parts of this thesis have been published as peer-reviewed scientific articles. For this thesis, the text of the published articles or the submitted manuscript has been integrally adopted. Editorial changes were made for reasons of uniformity of presentation in this thesis. Reference should be made to the original article(s).
Note: this is a low resolution reading version of the thesis. White pages have been removed for the reader's convenience. High resolution images can be requested with the author.

Livestock graze on luscious green pastures. The 'Palmiet' vegetation (central in picture, from right to left) reminds of wetlands that once dominated this valley's landscape ('Klein Langkloof', South Africa). Although livestock grazing provides meat and dairy products, the original wetlands used to hold precious water, protect against floods and assimilate carbon- and nutrient-rich soils. Such trade-offs are central to this PhD thesis, which deals with the consequences of how, and for what, humans manage the Earth's land cover.

1 GENERAL INTRODUCTION

1.1 BACKGROUND

Humans have altered a large proportion of the Earth's ecosystems to meet growing demands for food, fresh water and other natural resources (Foley et al. 2005, MA 2005b). Over 75% of the world's ice-free surface shows evidence of altered environmental features and processes, such as water cycles, biodiversity and primary production (Ellis and Ramankutty 2008, Verburg et al. 2013b), and human activities have appropriated over 50% of the global ecosystem production (Imhoff et al. 2004). The most dominant transformations made to the Earth's surface relate to expanding cities and villages, and converting land for intensive agriculture (Foley et al. 2005).

Four terms are central to this thesis: 'land cover, 'land use', 'land management', and 'ecosystem services'. Land cover refers to all physical biotic and abiotic components that make up the landscape, including natural vegetation, soils, cropland, water and human structures (Young 1994, Verburg et al. 2009). Land use is the purpose for which humans change land cover to their own benefit (Fresco 1994, Verburg et al. 2011) and consists of a series of different activities. The purpose can be food or fibre production, nature conservation or water storage. Land management involves human activities that together determine land use and directly affect land cover. Examples of land management activities include applying pesticides or irrigation, constructing fences and water-buffering weirs, clearing invasive species. Together, these activities contribute to or support a certain land use and to providing ecosystem services. In this thesis I will use 'land management' interchangeably with 'management', when applicable (i.e. when the study concerns land). Ecosystem services are defined as the direct and indirect contributions of ecosystems to human wellbeing (The Economics of Ecosystems and Biodiversity (TEEB), 2010).

Ironically, human society has become increasingly dependent on ecosystem services, including natural resources, while management activities to use these resources and services have contributed to land degradation and loss of ecosystems and biological diversity (i.e. biodiversity) (Foley et al. 2005, MA 2005b). 'Ecosystem services' have become an increasingly popular concept to demonstrate how biodiversity loss and land degradation affect an ecosystem's capacity to provide critical services, such as fresh water and food (Norgaard 2010, Mace et al. 2012). The concept's origin can be traced back to the 1970s (see Gómez-Baggethun et al. (2010) for an overview). To increase public interest in biodiversity conservation, ecosystem services were framed as underpinning beneficial economic activities. The concept bridges natural and social sciences, and focuses on human-environment interactions. Most research topics related to ecosystem services go beyond or combine issues of individual traditional academic disciplines (Carpenter et al. 2009).

The Millennium Ecosystem Assessment (MA 2005b) provided an unprecedented overview of the state of the world's ecosystems, the services they provide and how human wellbeing is affected. The MA reported that 15 out of 24 identified ecosystem services were being degraded or managed unsustainably. Moreover, the MA projected that ecosystem services' degradation would likely worsen in the coming decades and especially regulating services, such as global climate regulation, air purification and water regulation were deemed particularly vulnerable (MA 2005b). This continued degradation has far-reaching consequences, because vital ecosystem services and proper management thereof are essential for poverty alleviation and human wellbeing (MA 2005b, Carpenter et al. 2009). Management alters ecological processes and efforts to increase one ecosystem service often result in the loss of several others (Foley et al. 2005, ICSU-UNESCO-UNU 2008). Moreover, management practices intended to improve ecosystem services are often based on untested assumptions or sparse information (Carpenter et al. 2009). The full impact of management practices on the total bundle of ecosystem services is still poorly understood and this limited understanding generally leads to under-appreciation of ecosystems and their services.

The worldwide degradation and transformation of ecosystems suggest that managers and decision makers have limited understanding of what is at stake in terms of economic and social costs, benefits and values (Barbier et al. 2008). Failing to consider important ecosystem services and their values in current policy and management decisions strongly contributes to continued ecosystem degradation (TEEB 2010b, Barbier et al. 2011). Considering the economic consequences in terms of ecosystem services gained or lost is critical because most ecosystems face the risk of conversion to another land use to support economic activities (Chan et al. 2011). Only when the 'true value' of ecosystems and their services are known, and realistic outcomes and targets can be approved, appropriate management plans can be developed on basis of these practical compromises (Barbier et al. 2008). TEEB (2010b) provided insight in the economic significance of ecosystems and helped to increase the importance of the 'ecosystem services' concept for policy making. The establishment of the International science-policy Platform on Biodiversity and Ecosystem Services (IPBES), the incorporation of ecosystem services in the 2020 targets set by the 10th Conference of Parties to the Convention on Biological Diversity and several national follow-up studies (Larigauderie and Mooney 2010, Kumar et al. 2013) illustrate this increase in importance. Informing policy- and decision makers is crucial, as decision making shapes human activities and behaviour and therefore determines important drivers of ecological degradation and change (Daily et al. 2009, Fürst et al. 2011). However, although many scientific studies inform policymakers, the science behind them is not yet well developed (Ghazoul 2007, Daily et al. 2009, Kienast et al. 2009).

Understanding the effects of management on providing ecosystem services is crucial in projecting the consequences of policies and decisions. Compiling and analysing empirical evidence to support land management is required, as most management tends to be grounded in poorly verified assumptions (ICSU-UNESCO-UNU 2008, Carpenter et al. 2009). The following challenges for analysing the effects of management on ecosystem services provision can be formulated: (a) identifying indicators to characterise and quantify ecosystem service provision (Villa et al. 2009, Layke et al. 2012); (b) characterising land management and its effect on ecosystem services (De Groot et al. 2010b, Eppink et al. 2012); and (c) accounting for changes in land management (Erb et al. 2013, Van Asselen and Verburg 2013);. These three research challenges are further explained in Sections 1.3 to 1.5, while Section 1.2 describes frameworks that have been used for analysing ecosystem service provision. Each section also describes what tools or research steps are needed to overcome the research challenges. Together, these tools and research steps will contribute to reaching the main objective of this thesis, which is "to quantify the effects of management on ecosystem service provision" (Section 1.6). Three research questions are formulated in line with the research needs, also in section 1.6. The final section in this introduction (1.7) provides the thesis' outline and explains which research questions are addressed in the different chapters.

1.2 DEFINITIONS, CLASSIFICATIONS AND A FRAMEWORK FOR ANALYSING ECOSYSTEM SERVICE PROVISION

The 'ecosystem services' concept is relatively young and developing continuously. Definitions and classifications have also been heavily debated in literature (Costanza 2008, Fisher et al. 2009). The most frequently used definition used to be "the benefits people obtain from ecosystems" (MA 2003, 2005b), but more recent studies claim that the definition has led to confusion between benefits and services. As a result, TEEB (2010b) defined ecosystem services as "the direct and indirect contributions of ecosystems to human wellbeing", to which Haines-Young and Potschin (2013) added ".. and arise from the interaction of biotic and abiotic processes" in their 'Common International Classification of Ecosystem Services' (CICES) report. A universally acceptable definition seems highly implausible and definitions depend strongly on the purpose and perspective of the assessment. Therefore, Costanza (2008) argues that definitions will remain "appropriately vague" but can also be fine-tuned according to the research context. The different definitions by the MA, TEEB and CICES are good examples; the MA aimed to communicate general findings, TEEB focused on economic valuation of ecosystem services and CICES aims to develop an ecosystem accounting approach. This difference in aims has led to subtly adapted definitions. Because of this thesis' scope, I use the definition by TEEB (2010b): the definition describes ecosystem services as outcomes of ecosystems and as contributions to human wellbeing.

Classifications of ecosystem services have been equally diverse and differences occur because of the specific biophysical and socio-cultural context in which they are defined or scientific discipline of the researcher (Gómez-Baggethun et al. 2010). As reviewed by De Groot et al. (2010a), most ecosystem service categories have constantly featured in mainstream classifications, such as those by Daily et al. (1997), De Groot et al. (2002), MA (2005b) and TEEB (2010b). *Provisioning services* refer to biotic resources that can be extracted (De Groot et al. 2002). *Regulating services* refer mostly to processes rather than actual harvestable resources and include water purification, carbon sequestration, maintenance of soil fertility and pollination. The MA (2005b) also included *supporting services* (e.g. nutrient cycling and soil formation) and described them as underpinning other ecosystem services. However, many authors have voiced their concern about risks of 'double-counting' because supporting services would be accounted for in the provision of other services (Fisher et al. 2009, Nahlik et al. 2012). Barbier et al. (2008) compare supporting services to the 'infrastructure' required to provide other services, indicating that supporting services are of crucial importance but this importance should be considered when assessing the services they support. Because of the ambiguity around supporting services, TEEB (2010b) re-introduced *habitat services* in their classification. Habitat services include lifecycle maintenance (e.g. nursery for migratory species) and gene pool protection (e.g. maintenance of genetic diversity) and are crucial for the world's biodiversity and, consequently, most ecosystem services. *Cultural services*, finally, are the most 'human-centred' category as they refer to ecosystems as vital sources of inspiration for art, culture and spirituality, and subjects of interest for education and science. This thesis follows the classification by TEEB (2010b) but note that its 22 categories (Table 1.1) include several sub-categories that should also be distinguished, depending on the research scope; services such as food, raw materials and coastal protection have several 'sub-services' that each depend on different ecosystem processes and are provided in different environmental and societal contexts.

Ecosystem services research can have different aims, but a framework that consistently and comprehensively characterises ecosystem service provision is always required. A framework provides structure to the research and enables better comparison and validation of its outcomes (Bockstaller and Girardin 2003). Seppelt et al. (2011) underline the need for consistent and generic characterisation of ecosystem services. Well-known frameworks by MA (2003) and Daily et al. (2009) position ecosystem services in the interface between ecosystems and society, and relate service provision to (in)direct drivers. However, the actual provision, or 'flow', of ecosystem services has become the topic of an on-going debate (e.g. Villamagna et al. 2013, Spangenberg et al. 2014). Central in this debate are the questions whether all services can be characterised in similar ways and how to differentiate between actual and potential service provision. The 'cascade-model' of Haines-Young and Potschin (2010, Figure 1.1) has inspired discussions about the ecosystem services 'flow'. The model was developed to explain ecosystem service provision to a multi-disciplinary audience (Haines-Young and Potschin 2010). Many scientists have since discussed and further refined the elements of the cascade-model (e.g. De Groot et al. 2010a, Bastian et al. 2012). The most important suggestions to refine the figure include adding the 'value' step (which lacked in the original figure), further clarifying differences between ecosystem 'properties' and 'function', and incorporating feedbacks into the unidirectional (left-to-right) figure. Interestingly, an analytical purpose has been added to the original communication purpose of the model. Various methods and approaches have been based on the model, which is a clear sign of a creative scientific progress. This on-going process, however, faces the risk of leading to misunderstanding or confusion among the larger community of end-users of the ecosystem services concept (Seppelt et al. 2011).

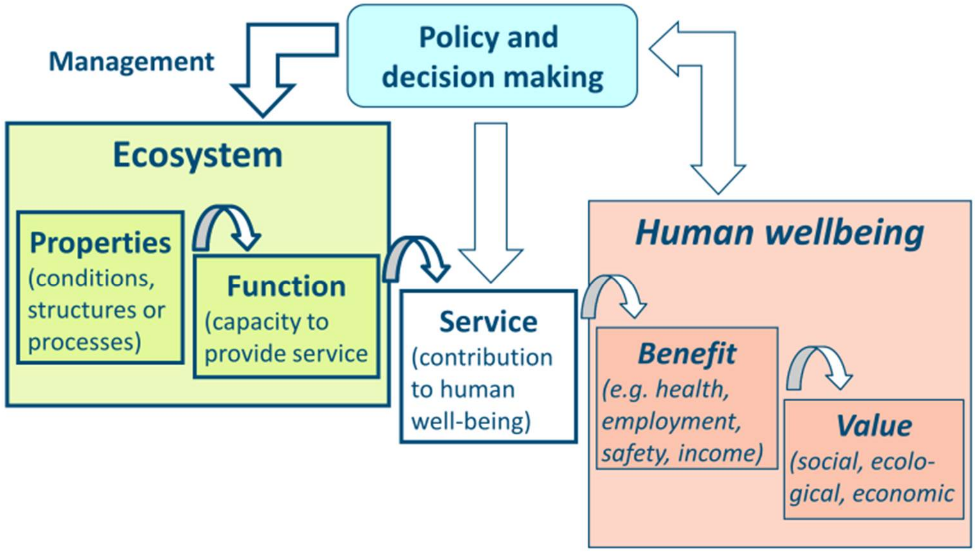

Figure 1.1: The cascade-model illustrates stepwise ecosystem services provision. The thickly outlined shapes indicate this thesis' focus. The figure is inspired by De Groot et al. (2010a) and Haines-Young and Potschin (2010).

The cascade-model (Figure 1.1) illustrates the stepwise provision of ecosystem services, from left to right. The cascade-model places the 'service' between 'ecosystem' and 'human wellbeing', which implies that no service is provided without ecosystems and no service is used without humans (Potschin and Haines-Young 2011, Spangenberg et al. 2014). The model reduces ecological complexity to 'ecosystem properties' that underpin 'ecosystem functions'. Ecosystem properties include the conditions, structures and process of ecosystems such as soil properties, nutrient cycles and biological diversity (Bastian et al. 2012). Ecosystem properties can be assessed without considering potential or actual use of ecosystem services; they are 'just there' (Bastian et al. 2012, Spangenberg et al. 2014). However, the properties form the existence of any kind of services that can be utilised by society. The conceptualisation of how ecosystem properties are converted to actual services has been the subject of a long-standing debate (Kienast et al. 2009, Spangenberg et al. 2014). 'Ecosystem function' is defined as the capacity to provide an ecosystem service (De Groot et al. 2010a), but this capacity has also been referred to as potential supply, ecosystem potential, ecosystem stocks and even ecosystem service per se (c.f. Villamagna et al. 2013). Most authors agree that the capacity to provide services differs from the actual services that are enjoyed by society, for instance due to low accessibility, absence of beneficiaries, or management choices (Schröter et al. 2014a). In other words: not all ecosystem properties constitute an ecosystem service (Schröter et al. 2014a, Spangenberg et al. 2014). There is less consensus, however, about how the service 'flows' out of the capacity (Villamagna et al. 2013). The flow is defined as the actual use of an ecosystem service (Schröter et al., 2012). Scientific consensus exists on how the flow for provisioning services can be assessed, but whether the flow for cultural, regulating and habitat ecosystem services can be determined is still debated (Fisher and Turner 2008, Ringold et al. 2013) Regardless of the differences in opinion on how to characterise them, distinguishing between potential and actual service provision is crucial because it enables the assessment of the biophysical capacity of an area to provide ecosystem services and the sustainability of ecosystem service use (UNEP-WCMC 2011, Villamagna et al. 2013, Schröter et al. 2014a). My research considers potential and actual provision for all relevant ecosystem service categories.

The scope of this thesis is indicated in Figure 1.1. My research will focus on the provision of ecosystem services in biophysical and ecological terms, i.e. 'properties', 'function' and 'service'. The use and appreciation thereof by society, i.e. 'benefit' and 'value', lies beyond this thesis' scope. The benefit is the socio-cultural or economical welfare gain provided through the ecosystem service, such as health, employment and income. Value is defined as the contribution of ecosystem services goals, objectives or conditions that are specified by a user (Costanza 2000, Farber et al. 2002). Policy and decision making (Figure 1.1) form preconditions, constraints and incentives for land management and other drivers (Daily et al. 2009, Fisher et al. 2009, see Section 1.4). I used the cascade-model to develop the framework for this thesis' research, because it assists in distinguishing between ecosystem properties, function and service. However, the cascade-model has rarely been used for a quantitative analysis of multiple ecosystem services. In my framework I furthermore added the notion of land management as influencing ecosystem properties and, consequently, ecosystem service provision. Chapter 2 describes the framework in detail and illustrates how it was used for indicator selection to assess the effects of land management on ecosystem services. The following sections deal with indicators for ecosystem service provision (1.3), characterising land management (1.4) and land management regimes (1.5).

1.3 INDICATORS FOR ECOSYSTEM SERVICE PROVISION

Indicators are crucial for quantifying ecosystem service provision. An indicator is a measure or metric based on verifiable data that conveys information about more than itself (BIP). A measure is a value that is quantified against a standard, whereas a metric is a set of data collected and used to underpin each indicator (BIP). Indicators can provide information to decision makers and land managers based upon which interventions can be identified, prioritized and executed (OECD 2001, Layke 2009a). A study by UNEP-WCMC (2011) reviewed which ecosystem services indicators were used for 11 sub-global Millennium Assessments (SGAs). The study provides the state of the art of ecosystem service indicators for multiple regions, spatial scales and used for communicating with decision makers. UNEP-WCMC (2011) found that considerably more indicators had been used for provisioning services (54), as compared to regulating (34), cultural (16) and supporting services (18, habitat services were not considered in the MA). Only a few indicators in SGAs were found to use underlying metrics, i.e. that actually measured service provision. Most ‘indicators’ relied on metrics related to the condition or extent of land covers, others relied on already processed outputs of ecosystem services or value thereof (UNEP-WCMC 2011). Many ecosystem service assessments lack consistent ecosystem service indicators and metrics that measure these indicators directly, due to limited data availability (Layke et al. 2012, Tallis et al. 2012).

The quantification of ecosystem services requires multiple indicators that correspond with the steps of the cascade-model (De Groot et al. 2010b, Villamagna et al. 2013). Measuring an ecosystem’s capacity to provide services is necessary but not sufficient to precisely determine the level of service provision (Tallis et al. 2012). De Groot et al. (2010b) propose two main types of indicators: (a) ‘State indicators’ describing the capacity of the ecosystem to provide the service, and (b) ‘Performance indicators’ describing how much of the service is actually used. Examples of both indicators are provided for all ecosystem services in Table 1.1. State indicators correspond to the ecosystem function (capacity) step of the cascade-model and performance indicators correspond to the actual service. Quantified information on both indicators can give information on the availability of an ecosystem service as well as the sustainability of ecosystem-service use (ratio performance / state) (Villamagna et al. 2013). However, Table 1.1 also shows that indicators for regulating, habitat and cultural services are less consistent than for provisioning services and do not always indicate actual and potential service provision. This reflects the findings of UNEP-WCMC (2011); indicators for all but provisioning services are difficult to find and quantify, which shows that data availability is a limiting factor for the successful quantification of ecosystem services. As the cascade-model shows, potential service provision is underpinned by ecosystem properties but these properties are rarely used, and their interaction with state and performance indicators is largely unknown (Bastian et al. 2012). Indicators for ecosystem properties (e.g. soil type, biological diversity, NPP and age of vegetation) could help to provide additional information in data-scarce environments, because they can act as proxies for potential and actual service provision (UNEP-WCMC 2011). Moreover, because ecosystem properties are directly affected by management activities they can provide information on how management (indirectly) affects service provision.

Table 1.1: Examples of 'state' and 'performance' indicators of ecosystem service provision. Based on this thesis and inspired by De Groot et al. (2010b) and UNEP-WCMC (2011). Ecosystem services classification by De Groot et al. (2010a). Services in shaded rows are not studied in this thesis. Units are given between parentheses, if relevant.

Ecosystem service	State indicator	Performance indicator
PROVISIONING SERVICES		
1. Food	Available stock (kg ha^{-1})	Actual productivity (kg ha^{-1} yr^{-1})
2. Water	Available water (m^3)	Extracted water (m^3 yr^{-1})
3. Raw materials	Available biomass (kg ha^{-1})	Harvested biomass (kg ha^{-1} yr^{-1})
4. Genetic resources	Availability of useful species	Harvested species
5. Medicinal resources	Available medicinal resources	Harvested medicinal resources
6. Ornamental resources	Available biomass (kg ha^{-1})	Harvested biomass ((kg ha^{-1} yr^{-1})
REGULATING SERVICES		
7. Air quality regulation	Potential air pollution removal (kg ha^{-1})	Captured air pollution (kg ha^{-1} yr^{-1}), improved air quality (%)
8. Climate regulation	Carbon storage (kg ha^{-1})	Difference between carbon storage of intact and impacted ecosystem (kg ha^{-1} yr^{-1})
9. Moderating extreme events	Vegetation with water-buffering capacity	Reduced flood risk, reduced damage
10. Water flow regulation	Water storage capacity (m^3 ha^{-1})	Increased water availability (m^3 ha^{-1} yr^{-1})
11. Waste treatment	Potential water purification (kg ha^{-1})	Amount of pollutant captured (kg ha yr^{-1})
12. Erosion prevention	Potential erosion prevention (kg ha^{-1})	Topsoil maintained (kg ha^{-1} yr^{-1})
13. Soil fertility maintenance	Soil organic matter content	Fertile topsoil (re)generated (kg ha^{-1} yr^{-1})
14. Pollination	Pollinator abundance, pollination rate	Harvest dependence on pollination (%)
15. Biological control	Natural predator abundance (%)	Harvest protected by biological control (%)
HABITAT SERVICES		
16. Nursery service	Number of maturing juvenile fish that depend on ecosystem	Dependence fisheries on nursery service, contribution to fish stock
17. Maintenance of genetic diversity	Abundance of keystone species	Biodiversity relative to intact ecosystem
CULTURAL SERVICES		
18. Aesthetic enjoyment	Area with stated preference	Recreation, house sales near that area
19. Nature-based recreation and tourism	Potential for recreation, potential number of visitors	Actual number of visitors
20. Inspiration for culture, art, and design	Natural features with cultural value	Rituals, art based on these features
21. Spiritual experience	Natural features with spiritual value	People engaging in spiritual activities
22. Information for cognitive development	Natural features with educational, scientific value	Number of studies, number of excursions

For the quantification of management effects on ecosystem service provision, information is needed on how management affects ecosystem properties and, consequently, how this can be related to indicators of ecosystem functions and services. The next section deals with how land management and its effect on ecosystem services can be characterised.

1.4 CHARACTERISING MANAGEMENT AND ITS EFFECT ON ECOSYSTEM SERVICES

Many scholars agree that ecosystem-services research should focus on how management interventions change ecosystem services, and argue that most management is not grounded on evidence-based assumptions (e.g. ICSU-UNESCO-UNU 2008, Carpenter et al. 2009, Mace et al. 2012). In addition, the 'ecosystem services' concept generally provides insight into the economics of conflicting land management goals (Perrings et al. 2010, Eppink et al. 2012). However, although many studies acknowledge the importance of management for ecosystem services provision and biodiversity, attempts to characterise and quantify management effects are variable and inconsistent. A quick search through ScienceDirect, Google Scholar and Web of Science for keywords 'ecosystem service*' OR 'ecosystem function*' in combination with '*(land) management*' OR 'ecosystem management' OR 'land use' returned over 300 results, but only one study (i.e. Eppink et al. 2012) defined 'management' in relation to land use or ecosystem services. In this section, I describe how management has been characterised in the literature and how scientists relate land management to ecosystem services, land use and decision making.

Contrary to my definition, Eppink et al. (2012 p. 55) define land management as "the organisation of the use and development of land and its natural resources", rather than as the human activities that together determine land use and directly affect land cover. Other studies do not define (land) management, but instead give context-specific examples by listing specific activities (e.g. mulching, moving and ploughing), land-use types, spatial plans or policy regulations. Studies that claim to analyse ecosystem services provision in relation to 'management' mostly focus on other drivers, ranging from direct, local drivers (e.g. agricultural and forestry techniques) to general and indirect drivers (e.g. policy measures, spatial planning, economic instruments, or land use types). Indirect drivers explain how land is managed and as such affect land cover indirectly, and at a large spatial scale. Management involves direct and local drivers, but insights into how management relates to indirect drivers are needed.

The research on so-called 'land systems' offers a timely and useful description of how management activities can be framed in relation to policy, land use and land cover, and how they can be captured in a more general context. Land systems represent the terrestrial component of the Earth system and encompass all processes and activities related to the human use of land (Verburg et al. 2013b). These processes and activities include investments, technological advancement, organisational arrangements, as well as the benefits gained from and unintended social and ecological outcomes of societal activities (Crossman et al. 2013, Erb et al. 2013, Verburg et al. 2013b). Just like ecosystem-service science, land-system science operates at the interface of the social and natural sciences, as it studies the interplay between human-environment systems that together determine land use and shape land cover. Land-system science has evolved from the study of land-use and land-cover change, and takes a systems perspective on the social and ecological aspects of land use rather than just monitoring consequences of land-cover change (Verburg et al. 2013b). Recent studies on 'management' and 'ecosystem services' have been mostly limited to relating land use and land cover to ecosystem services, rather than taking a systems perspective (e.g. Burkhard et al. 2009, Fürst et al. 2011, Burkhard et al. 2012, Poggio et al. 2013). Land use and land cover are important and easily up-scaled proxies of human activities, but fail to fully account for their direct effects on ecological processes and interactions. Therefore, in this thesis I explicitly define and distinguish between land use, land cover, land management and ecosystem services. Management adds an extra dimension to the broader land-use research.

Management activities are generally embedded in an organisation or institutional structure that coordinates land use and spatial planning (Eppink et al. 2012, Verburg et al. 2013b). The purpose for which humans undertake management activities (i.e. land use) is influenced by policy regulations, socio-economic developments, climate change, local challenges and traditions (Verburg et al. 2013c). Although other direct and mainly climate-related drivers (e.g. drought, fire and floods) also affect land cover, my research focuses solely on the effects of human activities. Note also that land management activities can affect land cover to support a specific purpose but, at the same time, can have unintended and undesired effects on other land uses and ecosystem services (ICSU-UNESCO-UNU 2008, Chan et al. 2011). Management includes, but is not limited to, ecosystem management (Brussard et al. 1998) and area-specific examples include coastal zone (Peña-Cortés et al. 2013), forest (FAO 1994) and grassland management (Jones and Hayes 1999). In the 'ecosystem services' literature, ecosystem management is inconsistently defined, and often incorrectly used as a synonym for (land) management. Note that ecosystem management refers to managing an area to conserve ecosystem services and biological resources, while sustaining human use (Brussard et al. 1998, MA 2005b). In other words, a balanced human-nature relationship is assumed and this automatically excludes many other situations in which intensive and mono-functional land management takes place. Furthermore, management activities also include nature conservation or restoration of important (characteristics of) ecosystems (e.g. biodiversity, water quality and aesthetic value). For example, managers or local communities that are responsible for national parks, protected areas and recreation areas focus on these activities (Turner et al. 1995, Chan et al. 2011). Restoration activities include replanting vegetation, clearing alien invasive species, and redirecting waterways. Conservation-related activities include constructing fences to limit access to resources or locations and promoting eco-tourism by constructing hiking tracks and accommodation.

In section 1.2, I introduced a stepwise way of characterising ecosystem services provision. Ecosystem properties can be converted into services (Haines-Young and Potschin 2010, Bastian et al. 2012). Management would logically mainly affect ecosystem properties and, consequently, the ecosystem's capacity to provide services. The updated cascade model by De Groot et al. (2010a; Figure 1.1) and other conceptual studies (e.g. Bastian et al. 2012) also refer to management as a possible direct driver of ecosystem change. Empirical studies into management effects on biodiversity and ecosystem services underline that most management is directed at altering ecological features, water balance, soil properties and plant functional traits (Kremen 2005, Balvanera et al. 2006, Başkent et al. 2011). However, as stated before, most studies focus on the intended management effects, for example in the context of agricultural production, wood production and crop pollination. Land management should take a wide perspective that also considers unintended and often undesired management effects on multiple services or 'bundles' rather than just focusing on specific land-use purposes.

Raudsepp-Hearne et al. (2010) define ecosystem service bundles as "sets of ecosystem services that repeatedly appear together across space or time". They consider the combined provision of multiple ecosystem services and claim that this enables management of both trade-offs and synergies, reduces their associated costs to society and enhances landscape multi-functionality. Raudsepp-Hearne et al. (2010) use an aggregated scale and sticking to administrative and land-use related boundaries. Moreover, the proxies used for ecosystem service provision are straightforward and relate to end-use of services rather than to ecosystem properties. Therefore, their study elucidates few direct management effects on ecosystem services. However, the authors correlate statistically different services, thus providing useful insights in trade-offs that occur on landscapes. Many other studies have focused on the impacts of land use on ecosystem provision, which can be assessed and aggregated for various spatial scales. Two landmark papers (Foley et al. 2005, Braat et al. 2008) relate land use and some management aspects with ecosystem services provision but these studies only discuss hypothetical cases.

Foley et al. (2005) describe land-use changes as a major driver for global change. They provide a much-referred-to case that compares natural ecosystems, intensive croplands and croplands with restored ecosystem services. The restored landscape represents a 'middle ground', in which a broad range of ecosystem services, including regulating services, would be provided. This is in sharp contrast to natural ecosystems that only provide regulating services, and intensive croplands that maximise food production at the expense of other services. The example by Foley et al. (2005) is referred to by many researchers because it offers an important conceptual framing that should be investigated empirically and generalised for different ecosystems and land-use types.

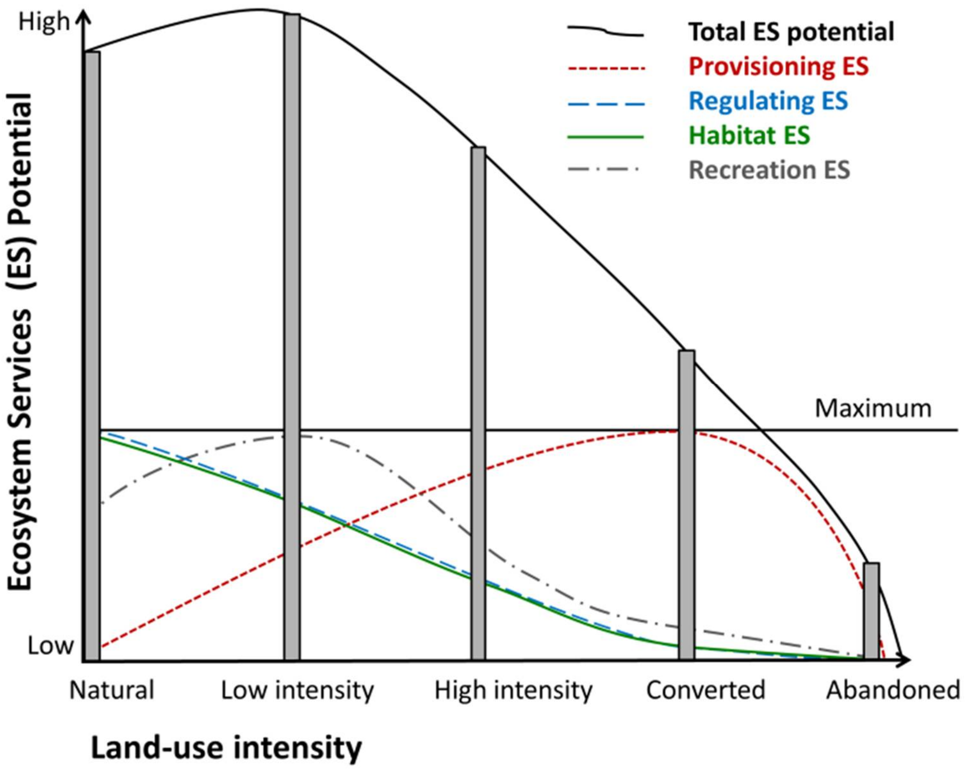

Figure 1.2: Hypothetical trend of potential ecosystem services (ES) provision with increasing land management intensity. Inspired by Braat and de Groot (2012).

Foley et al. (2005) argue that mapping changes in land cover, biodiversity and fertilizer use contributes to assessing global impacts of land-use intensity. Especially fertilizer use is seen as an important management indicator; the land-use purpose can be the same but different quantities and techniques of fertilizer application have different effects on ecosystems and their services.

The 'Cost of Policy Inaction' (COPI) study by Braat et al. (2008) also conceptualises how land-use intensity affects biodiversity and ecosystem service provision. They illustrate the management effect on providing ecosystem services by three hypothetical 'management systems'. Land use, management nor management systems are defined in the study, but the management systems are explained by to land use rather than to specific management activities. The first system is a natural ecosystem, in which all services are balanced. The second system is the original land cover that has been altered for extensive food production. This equally decreases the provision of all other services than food production. The third system is an intensive food-production system in which all other services are strongly reduced. The original vegetation has been cleared. The three contrasting systems in both studies (i.e. Foley et al. 2005, Braat et al. 2008) are similar. Foley et al. (2005), however, refer to the 'middle-ground' management system as 'restored ecosystem services', whereas Braat et al. (2008) describe such systems as 'extensive food production'. These are crucially different systems because they are characterised by a different purpose (restoration vs. large-scale food production) and, therefore, different management activities.

In the COPI-report, Braat et al. (2008) also graphically presented the level of ecosystem services provision as a function of biodiversity and land-use intensity. This graph was later updated by Braat and de Groot (2012) and me (Figure 1.2). The figure depicts a gradual trend of service provision for a wider range of land-use intensities. Five intensities of land use are distinguished, in order of increasing intensity: natural, light use, extensive, intensive and degraded. Unfortunately, the different intensities are neither defined nor characterised further in the original studies. They are a mix of ecological, land use and management terms and could therefore lead to confusion. Intensification of land use is generally associated with additional inputs such as fertilizer, pesticide and feed, and more dependence on technology and investments (Jangid et al. 2008, Erb et al. 2013). This means that changes in land-use intensity involve additional or different activities, i.e. management, that maximise the land use. Finding a way to generalise land-use intensity into distinctive steps is desirable, because each step can be characterised by different types of management activities. The different steps in land-use intensity would result into a typology of management regimes, i.e. a bundle of management activities that collectively serve a land-use purpose. With this in mind, I suggest formulating five general but logical levels of land management intensity, namely natural, low and high intensity, converted, and abandoned (Figure 1.2). I leave out terms like 'degraded' (which is a relative term that could be applied to many land-use intensities, and results from mismanagement), 'extensive' (which refers to spatial extent rather than intensity) and 'light use' (authors are unclear about how 'light use' relates to 'extensive' and 'intensive').

Braat et al. (2008) and Braat and de Groot (2012) merit using land-use intensity as a starting point to formulate a typology of different management regimes. With increasing land-use intensity, ecosystems are more affected by inputs, technology etc. Studies that relate land use to ecosystem services often refer to management-related factors to formulate categories of land-use intensity. Braat et al. (2008), for instance, also propose 'management scenarios' in which the spatial extent of eight dominant land-use types is projected for 2050. These land-use types include managed forests, extensive and intensive agriculture, artificial surfaces and cultivated grazing, and do not only refer to the land use (or cover) type but also to their history and how they are currently managed. The following section will discuss such management regime typology and what can be learned from earlier attempts to characterise land-use intensity and group management activities.

1.5 MANAGEMENT REGIMES AND STATES: BUNDLING MANAGEMENT ACTIVITIES AND THEIR EFFECTS

As noted by Mooney et al. (2009), the effects of ploughing, grazing, hunting, timber removal, river diversion, water extraction, polluting, fertilizer additions etc. are profound and becoming more intense. Thus, management activities should be characterised and grouped into comprehensive categories to account for their combined effects. The resulting typology should be general and flexible enough so that it can be applied in different contexts and ecosystem types. The need for such a typology has been expressed in the ecosystem services and land-system science literature. For land systems, Verburg et al. (2013b) and Erb et al. (2013) state that a structured analysis of land-use intensity is rarely conducted. Furthermore, ecosystem services research requires the quantification of ecosystem service provision by different 'management states' (De Groot et al. 2010b) or 'management regimes' (ICSU-UNESCO-UNU 2008). Management regimes and states differ substantially from each other and both studies probably mean the same.

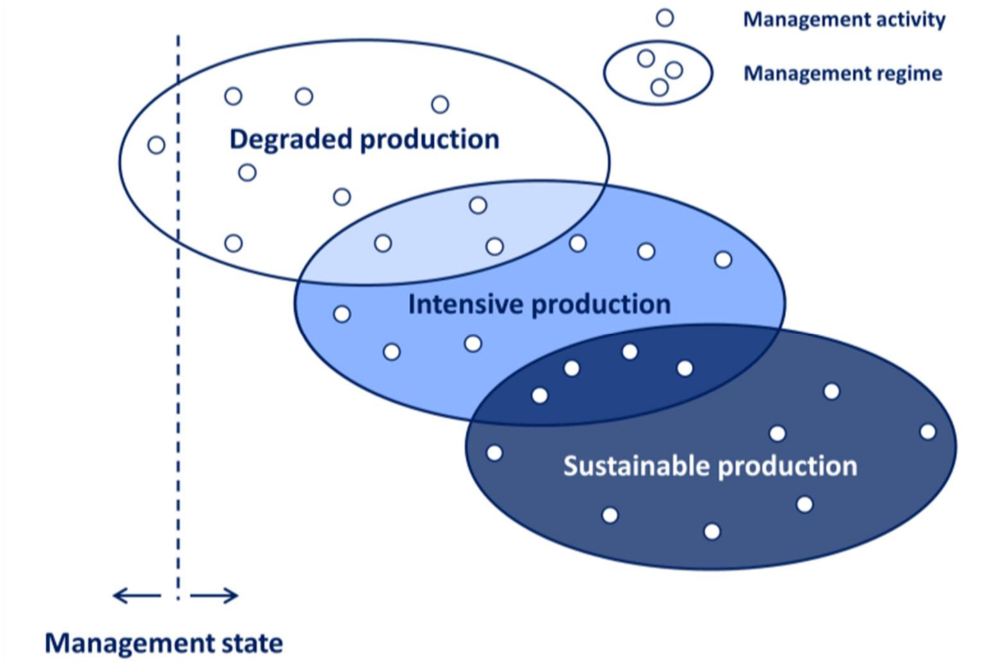

Figure 1.3: Illustration of the difference between management activities, regimes (bundles of activities), and states. A management state occurs at a moment in time, and results from a series of management activities. The degraded, intensive and sustainable production systems illustrate different management regimes with overlapping activities.

To avoid confusion, I illustrate the difference between three key concepts related to land management in Figure 1.3. A management regime is defined as the bundle of human activities that serve one or more land-use purposes. This bundle of activities results in distinguishable end-results of land use, cover and specific ecological and other characteristics of a given area. I define this as a management state, i.e. a 'snapshot' resulting from a series of management activities. At larger landscapes, multiple management regimes can co-occur simultaneously and the landscape's state is determined by all activities that take place.

Differences can occur at a local scale (i.e. per field with a specific management regime) because different intensities occur in each management regime. Consider, for instance, the management regimes in Figure 1.3. 'Intensive production' is characterised by activities such as converting original vegetation into cropland, lowering the water table, mixing the soils, and applying pesticides. In the case of more 'sustainable production', however, land cover is likely to be affected less heavily or at least differently. Limited conversion takes place under such management regime and crops are rain-fed, fertilizer use is low and no pesticides are used. Both systems can depend on irrigation, but more intensive systems require more intensive water management regimes, whereas the sustainable systems rely on rain-fed irrigation. The third production system in Figure 1.3, 'degraded production' also depends on irrigation, but other factors, such as soil degradation could contribute to even more intensive ways of extracting and applying water. Summarising, all three production systems have a similar purpose (food production) but largely different management activities. The combination of all management activities (i.e. the management regime) will result in clearly discernible management states. Using terms like 'degraded' and 'sustainable' would in this case not be problematic, as they have a clear reference point and refer to one land-use purpose, i.e. production. The term 'regime' is derived from literature that compares outcomes of different agricultural production systems. Each system is characterised by different chemical inputs and/or irrigation (e.g. Snapp et al. 2010). In the literature on flora and fauna management, the term is used for a very specific and localised range of treatments (Jones and Hayes 1999, Clegg et al. 2003). Additionally, 'regime shifts' in ecology indicate that ecosystems have been reorganised or transformed as a result of human activities or other driving forces (Scheffer et al. 2001, Folke et al. 2004). Such regime shifts are frequently referred to in the ecosystem service literature (e.g. Díaz et al. 2006, Bennett et al. 2009).

Below, I review the literature on both the conceptual and practice aspects of land-use intensity. Since land-use intensity is described by many authors through differences in management activities, this literature forms a useful starting point for developing a typology. Moreover, most of the reviewed land-use studies aim to study global land-use intensity, which automatically asks for a general, consistent regional typology that can be up-scaled to the global level. The findings below are described in chronological order of publication.

Foley et al. (2005) provide an extensive overview of how land use has changed throughout history. They describe stages in land use transition as going form pre-settlement to frontier, subsistence, intensifying and intensive land use, a notion that is based on De Fries et al. (2004). Foley et al. (2005) argue that increased irrigation, use of high-yielding cultivars, chemical fertilizers, and pesticides are the main factors with which you can describe intensifying land management. They go on to describe the impacts of these management practices in terms of global land cover, freshwater resources, forest resources, climate, air quality and occurrence of infectious diseases. An interesting addition is that Foley et al. (2005) include protected (recreation) areas as an increasingly important land-use type worldwide. They suggest that strictly natural areas might be declining, but that this can be counter-balanced by the establishment of actively protected (managed) areas, be it for biodiversity conservation and/or recreation purposes.

Ellis and Ramankutty (2008) offer an interesting perspective on how to analyse human impacts on global land cover. They propose so-called *anthropogenic biomes* and call for acknowledging that the terrestrial biosphere has been restructured by humans and biogeochemical cycles substantially altered. They distinguish land-cover types based on dominating land-use purpose and, when applicable, irrigation type (e.g. rain-fed or irrigate). See Table 1.2 for an overview of anthropogenic biomes. Ellis and Ramankutty (2008) furthermore propose considering population density as a proxy for (potential) ecosystem modification. The authors add that anthropogenic biomes can be assessed through observing changes in ecosystem structure and processes, such as net primary productivity (NPP), carbon and reactive nitrogen balance, and biodiversity. The classification by Ellis and Ramankutty (2008), for instance, distinguishes between remote, populated and residential pastures, which are all characterised by decreasing biodiversity, and increasing carbon emissions, reactive nitrogen, and NPP (Table 1.2).

The GLOBIO3 model has been developed to assess and project human-induced changes in global biodiversity (Alkemade et al. 2009). Drivers considered in the model are land-use intensity, land-cover change, fragmentation, climate change, atmospheric nitrogen deposition and infrastructure development. Alkemade et al. (2009) link land-cover classes to categories of land use and assign biodiversity values (i.e. the mean species abundance (MSA)) to these newly formed classes. The authors add sub-categories to general land-cover types, and distinguish between them based on land use intensity, thereby suggesting a wide range of management activities that could be considered. Snow and ice as well as bare areas (e.g. deserts and high alpine areas) are assumed to occur as 'primary vegetation' or permanently covered, so no sub-categories are assigned to these classes. For all natural land cover classes except for bare areas, primary vegetation is described as permanently covered, pristine or dominated by original vegetation. In addition, eleven categories are used in GLOBIO3 (Table 1.2). Descriptions of the categories feature a mix of land cover, land use, management, and finally the state of the remaining vegetation. Management activities are mentioned and assumed to have an impact on land cover but this is not explicitly investigated.

In order of increasing intensity, GLOBIO3 divides forests into primary, lightly used, and secondary forests, as well as forest plantations. Lightly used forests are characterised by extractive use, such as hunting, selective logging, but with minimum impact allowing for regrowth of naturally occurring species. Secondary and plantation forests are both characterised by vegetation removal, but in secondary forests re-growth occurs and trees are replanted in plantations. Scrublands and grasslands are divided into primary vegetation, livestock grazing and man-made pastures. The main difference between the latter two is that man-made pastures are forests or woodlands converted for grazing, whereas livestock grazing involves the replacing of wildlife with grazing livestock. Alkemade et al. (2009) furthermore distinguish between low- and high-input agriculture (land cover class: cultivated and managed areas). Low-input is assumed to feature subsistence farming, with low external input, whereas high-input resembles conventional, irrigation-based agriculture, dependent on high external input. Agroforestry is described as a cropland or forest mosaic, in which trees are kept as shade and wind shelter. Finally, built-up areas (belonging to artificial surfaces) are described as 80% built up.

The land-use or land-cover classes by GLOBIO3 offer useful insight into which management activities can generally be related to changing land-use intensities. Using land cover as the basis for a typology seems useful, since describing land use and management as additional sub-categories will result in a hierarchical classification. Alkemade et al. (2013) applied GLOBIO's classification for rangelands, i.e. grass-, scrub-, wood- and wetlands and deserts used for grazing. They assigned land-use intensity to rangelands based on relative stocking rates and developed a classification that is similar to mine: natural (natural stocking rates, similar to wildlife grazing), moderately used (higher stocking rate, different seasonal patterns in grazing), intensively used (very high stocking rate and higher impact on vegetation), man-made (converted, high degree of human management), and finally un-grazed, abandoned rangelands (no longer in use, overgrazed, no forests (yet) developed). These five classes describe a trajectory from natural all the way to converted ecosystem and abandoned land use, which seems realistic and applicable to a wide range of ecosystems and land cover types. The addition of abandoned land use is interesting; estimates suggest that an area the size of France was taken out of agricultural production globally between 1995 and 2005, and abandonment can be regarded as an encouraging option in places where net agricultural returns are low, land is severely degraded, or where re-wilding and habitat enlargement has positive impacts on biodiversity (Munroe et al. 2013). Land abandonment is a transitional stage rather than a static end-state, for instance between intensive agriculture and restored natural grasslands. Alkemade et al. (2013) also describe what could be seen as 'management states': changes in vegetation structure relative to natural rangelands as a result of grazing and, in the case of man-made grasslands, other human management.

De Groot et al. (2010b) describe categories of *management states*, which actually are a mixture of regimes and states. The categories are described through a mix of land cover, land use, management activities and ecological state. Their 'management states' are: wild/un-managed ecosystems, sustainably or extensively managed, degraded, intensively managed and developed. The categories largely compare to those presented in the COPI-study (see Braat et al. (2008) and Figure 1.2) and represent a similar range to the categories presented by Alkemade et al. (2013). However, the management states are not consistently described and are therefore difficult to distinguish: 'systems' are compared to 'ecosystems'; land use and management are used interchangeably; relative terms like 'sustainably managed' and 'degraded' are used to describe separate management states, although they could apply to other states as well, and; 'intensively managed' and 'developed' are difficult to discern as they both feature a degree of conversion. De Groot et al. (2010b) suggest that most management states could be evaluated based on pollution (i.e. impacts), land-use types or ecological state, but their typology would not enable a consistent comparison. Furthermore human activities that were mentioned include use of certain ecosystem services, resource use in relation to natural productivity, harvesting, constructing (permanent or limited) infrastructure, external inputs of energy and/or resources, land cover conversion and constructing cultivation systems. The categories presented by De Groot et al. (2010b), just like by Braat et al. (2008), offer useful and interesting examples, but a more consistent set of characteristics would be needed to systematically differentiate between different management regimes and states.

A classification of *land systems* is suggested by Van Asselen and Verburg (2012). The classification is intended for global-scale land change modelling; the authors classify combinations of land cover composition, livestock density, and land use intensity in a series of land systems, stating that current models generally reduce land management to a single, uniform management factor per region. Although Van Asselen and Verburg (2012) also fail to define or describe management, their study clearly shows what they consider the most important management factors. Land management is in their study limited to agricultural activities. This is odd because the land system classification is not limited to agricultural land. They describe several standard land-use types, but only consider the type and stocking intensity of livestock as management factors. Through these factors, the classification can account for the land-use intensity of various animal farming systems, namely extensive, medium, and intensive ones. However, a clear limitation is that other land systems, for instance related to forest and wetland systems, are lacking in the global classification. For instance, cropland systems are well-defined and many different ones are distinguished, but for forest systems only the simple distinction between 'dense forest' and 'open forest' (with livestock grazing) is made. Furthermore, Van Asselen and Verburg (2012) predict the location of land systems using location-specific factors such as include climatic, soil type, terrain (slope, altitude), vegetation (potential natural vegetation) and socio-economic variables (market influence and accessibility, and population density). Most factors could serve to indicate that management *state* of a land cover, whereas the socio-economic variables give more information on the context of management. The predictive approach is similar to that followed by Alkemade et al. (2013). Both studies assess land cover and define spatially explicit variables to predict what land use and supporting management activities might occur.

Erb et al. (2013) provide a conceptual framework for studying land-use intensity, based on a review of theoretical concepts and indicators available in the literature. The authors state that the scientific understanding of land-use change is insufficient to characterise the conditions under which sustainable land-use intensification could occur, because a clear definition is lacking, 'intensity' is an ambiguous term and land use studies usually disregard the complexity of systems' intensification processes. Erb et al. (2013) propose characteristics that should be monitored when assessing land-use intensity, including: a) inputs to the production system (cropping frequency, rotation length, techniques, types of regeneration etc.); b) outputs from the production system (yield, stocking density, felling rates etc.) and c) changes in ecosystem properties (biodiversity, NPP, carbon stocks, water and nutrient cycle, etc.). The latter reflect unintended outcomes of land use and represent powerful drivers of land system dynamics, and refer to what I call a 'management state'. An important difference with the previously described global land-use assessments is the proposed inclusion of output indicators. Erb et al. (2013) argue that the output (production per area and time) better reflects land-use intensity, because it represents the purpose better. This remains to be seen, as high output can just as well be an indication of fertile soil, favourable climate, or efficient rather than intensive management. The inclusion of stocking density as *output* indicator is also unusual, because it can also be considered as an input that affects land cover directly. Erb et al. (2013) furthermore argue that the difference between the actual and potential states of ecosystems could be a good measure to use as proxy to measure intensity and sustainability (sensu Haines-Young et al. 2012).

Table 1.2: Overview of classifications of management regimes and/or (corresponding) states, as presented by several conceptual studies and global-scale land use (system) assessments. Note that not all studies aimed to come up with such a classification; the table below is a result of own interpretation.

Reference	'Management regimes'	Management activities	'Management state' indicators
Foley et al. (2005)	Natural ecosystem Intensive cropland Restored cropland Protected natural recreation	Chemical fertilizer use Increased Irrigation Pesticide use High-yielding cultivars	Fresh water quality and quantity Land cover Infectious diseases
Ellis and Ramankutty (2008)	Irrigated Rain-fed Urban, populated areas Wildlands Remote areas	Irrigation Rain-fed irrigation Population density as proxy for ecosystem modification	Land cover NPP Carbon and reactive nitrogen balance Biodiversity
Alkemade et al. (2009), Alkemade et al. (2012)	Primary vegetation Lightly used natural forest Secondary forest Forest plantation Livestock grazing Man-made pastures Agroforestry ('mosaic') Low-input agriculture Intensive agriculture Abandoned, un-grazed Built-up areas	Hunting Selective logging Allowing vegetation regrowth Removing original vegetation Planting exotic trees Replacing wildlife by livestock Intercropping Rain-fed irrigation Fertilizer use Changing stocking density Grazing with seasonal patterns Abandoning land use	Biodiversity (MSA, relative to original vegetation) Land cover Vegetation structure Disturbance Fragmentation Nitrogen balance
De Groot et al. (2010b)	Wild/ un-managed Sustainably, extensively managed Degraded Intensively managed Developed	Restoration of ecosystem services Protection of biodiversity Resource and service extraction based on natural carrying capacity Abandoning intensive management External energy / resources inputs Conversion of original ecosystem Construction of infrastructure Cultivation	Biodiversity (species numbers, relative to reference situation) Pollution Degradation of vegetation Vegetation structure Natural productivity
Van Asselen and Verburg (2012)	Natural Mosaic Extensive cropland Medium cropland Intensive cropland Urban	Type of livestock Stocking density of livestock	Climate-related Soil type Terrain (slope, altitude) Land cover Vegetation cover
Erb et al. (2013)	None	Stocking density Fertilizer use Cropping frequency Rotation length Regeneration types Stocking density *Yield*	Biodiversity Nutrient cycle Water cycle Carbon stocks NPP Soil quality Actual and potential state

Most studies that are reviewed above either aim at the global scale or have a strictly conceptual purpose, i.e. to illustrate research questions. Because such studies require a generalising approach, I could distil important characteristics of universally applicable management regimes.

Table 2.1 summarises the classifications as proposed in the studies discussed before. I provide an interpretation of the classifications into management regimes, give an overview of the mentioned management activities and, if applicable, management states. Note that the table does in no way provide an overview of all management activities mentioned in literature. This review was initiated from the point of view of compiling classifications similar to management regimes.

Land-use intensities are mostly distinguished based on management factors that have a strong focus on agricultural production: stocking rate, fertilizer use and irrigation regime. Frequently occurring management regimes include *natural ecosystems*, *low* and/or *high intensity land use* of existing land cover, and finally *converted* or *abandoned land*. In addition, the studies suggest a multitude of indicators for assessing the management state or predicting the management regime of an area, but very few studies put the indicators to use. The consensus is that a typology of management regimes should be mirrored by that of management states with indicators of biodiversity, land cover, vegetation structure, and hydrological, carbon and water cycles. Many authors admit that more specific characteristics of both management regime and state would in fact be difficult to assess on a global scale (Ellis and Ramankutty 2008, Van Asselen and Verburg 2012, Verburg et al. 2013a).

As can be seen in Table 1.2, most studies suggest using ecological or production system properties in addition to management activities to measure land-use intensity. From an ecosystem-services perspective, assessing the management state is desirable, as indicators for management state can also indicate ecosystem service provision. By definition, management activities affect land cover directly and thereby influence an ecosystem's capacity to provide ecosystem services. From that perspective, it seems crucial to link management regimes with corresponding management states, i.e. appropriate ecological indicators for measuring management effects. Interestingly enough, many ecological studies generally classify vegetation condition in relation to human-induced or natural degradation (e.g. Milton et al. 1994, Snyman 1998, Thompson et al. 2009). Classifications of vegetation condition involve either unidirectional steps (from pristine to near pristine, lightly, moderately, and severely degraded) or more cyclical steps (intact, degraded, transformed, open, restored). Vegetation structure, regeneration capacity, composition and degree of invasive plant invasion are also used to indicate the state of an ecosystem because of pressure from grazing, agriculture, or natural causes (Snyman 1998, Euston-Brown 2006, Sigwela et al. 2009). The challenge for relating management with ecosystem services lies in the selection of comprehensive and consistent indicators of management state that can indicate ecosystem services provision. In addition, ecological studies generally fail to link vegetation condition to specific management activities or regimes.

Some authors have suggested additional factors that should be taken into account when classifying management regimes or states. Land-system studies increasingly regard areas as socio-ecological systems, and suggest including socio-economic indicators in addition to ecological and biophysical indicators when analysing management regimes (Erb 2012, Van Asselen and Verburg 2013). Several authors have suggested assessing ecosystem services and land use/management within administrative rather than ecological boundaries (e.g. Ellis and Ramankutty 2008, Bennett et al. 2009, Raudsepp-Hearne et al. 2010). This is motivated by the realisation that social processes (demand, policies, regulations, market force) shape the production and consumption of ecosystem services (Raudsepp-Hearne et al. 2010, Van Asselen and Verburg 2012). A landscape or larger region should be seen as a combination of different socio-ecological systems with each a different dynamics in terms of management, land use and ecosystem service provision (Raudsepp-Hearne et al. 2010, Peña-Cortés et al. 2013). Consequently, this typology of management regimes should reflect social processes, and indicators must be sought that best reflect these processes. Examples include different concessions (who is allowed to harvest or grow where), protection of resources, biodiversity and landscape (De Groot et al. 2010b, Chan et al. 2011, Peña-Cortés et al. 2013). Another way to account for social dynamics is to assign different management activities to regimes, and differentiate between classes with the general management regimes as well. Natural systems, for instance, can then be described as either strictly protected, i.e. no access, no extraction, or protected to allow for nature recreation and/or sustainable extraction of resources by local communities.

Based on the above, a management regime typology should include the following classes: *natural ecosystems*, *low intensity use*, *high intensity use*, *converted to different land use*, and *abandoned*’. Note that most *natural* ecosystems are also used by people, but with limited impacts and no infrastructure or technology. *Low* and *high intensity use* regimes are both typified by strong human influence, and support production of resources to humans. *High intensity use* systems, however, depend on infrastructure, technique, and additional inputs to extract and produce these resources, whereas this is not used in low intensity use systems. *Converted ecosystems* form an important category, as worldwide many forests and woodlands are converted to support intensive agriculture, aquaculture or urban expansion. The amount of ecosystem services provided can be extremely high in converted systems that have food production as land use. The question about these systems should be to what extent the provided services are dependent on human inputs as compared to natural processes, and how the provision of these products compares to that of the original system, in terms of diversity and sustainability. *Abandoned land*, finally, should be included in the classification in the context of converting low-value land into high-value land, or restoration of ecosystem services. For instance, degraded and unused areas could be attractive for agricultural development, ecosystem restoration, etc., perhaps even more than intact ecosystems. In the light of supporting global food production, and sustainable intensification, this is important to consider (Verburg et al. 2013c). Terms like ‘intensification’, but also ‘restoration’ and ‘degradation’ can be considered as transitions from one to another management regime, but are not a management regime or state by themselves. In addition, the term ‘sustainable’ should always be used in the context of the given management regime, land use and area. What is considered ‘sustainable’ in the one production system might differ completely in other locations, depending on environmental and climatic conditions. In that sense, organic agriculture should not always be considered as optimal and sustainable, just like intensive agriculture is not always unsustainable either. Sustainability depends on which management activities occur, and what the effects of those bundled activities (i.e. regime) are on the land cover.

1.6 OBJECTIVE AND RESEARCH QUESTIONS

In line with the above-mentioned research challenges, this thesis aims **to quantify the effects of management on ecosystem service provision.** In order to achieve this objective, my research requires suitable criteria for indicator selection, workable definitions, indicators for ecosystem service provision characteristics and a consistent typology of management regimes and states. This thesis will improve understanding of ecosystem service provision as well as the effects of land management on it. This improved understanding enables scientists to assess ecosystem services in a structural and consistent way, and helps decision makers and land managers to understand the effects of their management choices and activities.

The central question of this thesis is "What are the effects of management on ecosystem service provision?" This central question is addressed through the following research questions:

1. What are the key indicators to quantify ecosystem services provision?
2. How can the effects of management on ecosystem service provision be quantified?
3. How can management regimes be conceptualised to quantify their effects on ecosystem services?

To answer the research questions, I developed an indicator-based approach by advancing a framework for systematic indicator selection. 'Management' was included in the framework as a driving factor and clearly distinguishable steps of ecosystem service provision were defined. The framework was then applied in three case studies: 'Het Groene Woud' (a rural area in The Netherlands), mangrove ecosystems in Java (Indonesia) and semi-arid to dry sub-humid rangelands. These case studies were used to refine indicator selection, select indicators for those ecosystem services that had not been studied in the other case studies, and to refine the typologies of management regimes. Important differences between the case study areas relate to data availability, spatial scale and ecosystem services that are provided. The available data resulting from the case studies was integrated and used to quantify the effect of land management on ecosystem services provision.

In the first case study in 'Het Groene Woud', the indicator-selection framework was developed, refined, and tested. This indicator-based approach quantified and spatially modelled ecosystem service provision. Katalin Petz (a fellow PhD student) and I closely collaborated on this research and published together. Together, we presented the ecosystem service indicators and developed a stepwise methodology to select key indicators for assessing land management effects on ecosystem services.

In order to generalize and conceptualise the effect of different management regimes on ecosystem services, I developed a consistent typology of management regimes and corresponding states (research question three). This was done and tested for mangrove ecosystems in Java, Indonesia. The framework for indicator selection and typology of management regimes were combined to analyse the provision of ecosystem services per management regime.

In the third case study, I applied and refined the management regime typology to the context of semi-arid to sub-humid rangelands. These rangelands are increasingly used for livestock grazing and smallholder agriculture, but relatively few data are available on ecosystem services provision. The case study, therefore, served to test the applicability of my framework and management regime typology in data-scarce regions. I compiled recurring indicators from the literature for soil erosion and surface runoff, and incorporated these indicators into 'indicator interaction diagrams'. Quantitative information for the indicators was integrated into a comprehensive dataset and linked to indicators for management regimes. The effects of land management were analysed by comparing quantitative data for different management regimes.

1.7 OUTLINE OF THE THESIS

This introduction (Chapter 1) and the combined synthesis, discussion and conclusion of this thesis (Chapter 6) are written to connect the various independent scientific papers that are reprinted in Chapters 2 to 5.

Chapter 2 presents a framework and stepwise methodology to select indicators that assess the effects of land management and ecosystem services. This indicator selection is illustrated through a case study in 'Het Groene Woud', a rural landscape in the south of the Netherlands. The different steps identify, select and evaluate indicators for ecosystem properties, function and services.

Chapter 3 describes the application of the framework for the 'Groene Woud' case study, in which the effects of management on the provision of eight ecosystem services was analysed. Indicators and proxies were selected from those identified in Chapter 2 and used to quantify and map the area's ecosystem services. Special attention was paid to the role of green landscape elements for ecosystem service provision. In addition, a simple scenario analysis illustrated the effect of land-use intensification. Research for Chapters 2 and 3 was conducted together with Katalin Petz. Both contributed equally to the research.

Chapter 4 presents a case study on the consequences of coastal land-management decisions on mangrove ecosystem services in Java, Indonesia. Java's coasts are characterised by a combination of heavily degraded, modified or even converted mangrove ecosystems, and some remaining natural ecosystems. Chapter 4 presents a novel typology of management regimes for mangrove systems, based on (underlying) policy regulations, management activities and ecological characteristics. The provision of mangrove ecosystem services per management regime was compared through this typology and this offered valuable insights into the potential impacts of mangrove degradation, conservation, rehabilitation and land-use intensification.

Chapter 5 presents the results of a case study on the effects of different rangeland management regimes on soil erosion and surface water runoff in drylands. The case study illustrates how findings from studies in other semi-arid rangelands can be integrated and applied to an area for which data are scarce. Soil erosion and reduced water availability are two crucial problems for land managers in drylands but the effects of land use and management are largely unknown. I developed so-called 'flow diagrams' to summarise the key indicators for soil erosion and surface runoff and illustrate relations between the indicators. In line with the typology developed in Chapter 4, I constructed a management regime typology for semi-arid rangelands, which considers management activities such as livestock grazing and vegetation restoration. I related quantified indicators for soil erosion and surface runoff to management regimes and was able to show substantial differences between how erosion and runoff could be reduced by management.

Finally, the findings are synthesised and discussed in Chapter 6. This chapter will show the versatility of my research framework and the robustness and applicability of my management regime typology. I reflect on how each case study contributed to answering the research questions and conclude how my study's results support decision making. In addition, I reflect on how the framework for indicator selection, typology of management regimes, flow diagrams and the study's quantitative information can be integrated in an integrative approach to quantify the effects of management on ecosystem services. The conclusion shows that explicitly distinguishing management activities from land use, land cover and ecosystem services is a constructive approach.

This cascading weir might seem quite an 'unnatural' structure, but it plays a crucial role in restoring wetlands and, thus, capturing precious water and organic matter in the soils ('Klein Langkloof', South Africa). In ecosystem services science, the 'cascade-model' is used frequently to describe ecosystem service provision in three steps; ecosystem properties (1) together determine the capacity (2) of an ecosystem to provide ecosystem services (3).

2 FRAMEWORK FOR INDICATOR SELECTION TO ASSESS EFFECTS OF LAND MANAGEMENT ON ECOSYSTEM SERVICES

ABSTRACT

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 twelve properties indicators, nine function indicators and nine 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.

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 21: 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 (2005b) highlighted the importance of these ecosystem services for sustaining human wellbeing. The Economics of Ecosystems and Biodiversity study (TEEB 2010b) 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 (Fisher et al. 2008, Carpenter et al. 2009, 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 involves human activities that affect land cover directly (Kremen et al. 2007, Olson and Wäckers 2007, Verburg et al. 2009). Land management supports land use and includes ecosystem management (Brussard et al. 1998, Bennett et al. 2009). Land cover refers 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 ecosystem management only refers to managing an area so that ecological services and biological resources are conserved, while sustaining human use (Brussard et al. 1998, MA 2005b). Examples of land management include irrigation schemes, tillage, pesticide use, nature protection and restoration (Follett 2001, Bennett et al. 2009).

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 (Van Strien et al. 2009, Villa et al. 2009, De Groot et al. 2010b, Bastian et al. 2012).

Given these four challenges, a consistent and comprehensive framework for analysing ecosystem services seems required (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, formulating a comprehensive set of indicators (Niemeijer and de Groot 2008, Layke et al. 2012), that enables the assessment of land management effects on ecosystem services provision on different spatial scales, is important (Carpenter et al. 2009, Van Strien et al. 2009, De Groot et al. 2010b). With indicators, policy-makers and land managers can be provided with information, based upon which interventions can be identified, prioritized and executed (OECD 2001, Layke 2009a). 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 to systematically select indicators that can be used to analyse the link between land management and ecosystem services provision at multiple scales. To achieve this objective, we developed a consistent framework for indicator selection that 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 then illustrate its use for indicator selection. We then apply the framework in a case study to assess the effect of land management on ecosystem services. Characteristics of and interactions between indicators were studied, and all indicators were evaluated based on a set of criteria. The case study was done in the southern part of the Netherlands, where multiple ecosystem services are provided across different spatial scales.

2.2 METHODS

2.2.1 Framework

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 of the MA (2003), which was itself based on a Driver, Pressure, State, Impact, Response (DPSIR) framework. We adapted the frameworks by TEEB (De Groot et al. 2010a) and Haines-Young and Potschin (2010) for indicator selection. These are among the most recent and comprehensive ecosystem service assessment frameworks. The TEEB framework explains the link between biodiversity, ecosystem services and human wellbeing (De Groot et al. 2010a) and builds on several recent studies (MA 2003, Braat et al. 2008, Fisher et al. 2009). The TEEB-study calls for developing indicators for assessing the economic consequences of biodiversity and land-use change (Reyers et al. 2010). The stepwise so-called *cascade-model* by Haines-Young and Potschin (2010) is useful for assessing ecosystem services provision in a structured way, linking ecosystem properties to functions and services. Although the importance of land management is acknowledged in 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.

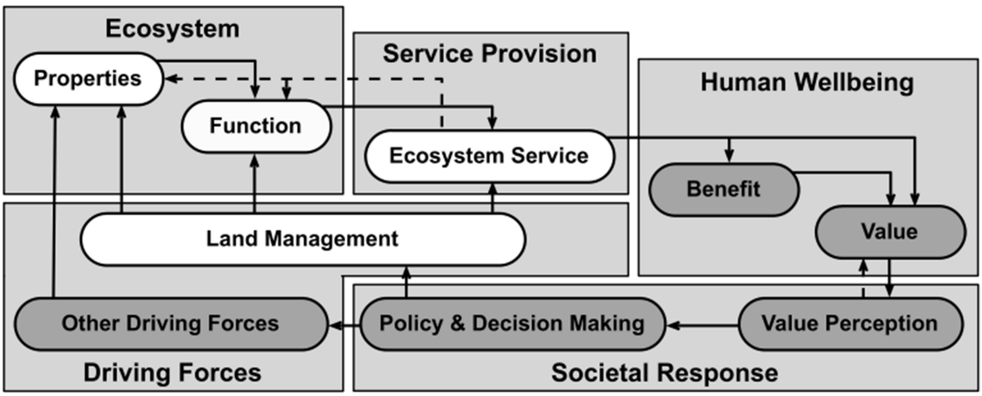

Figure 2.1: Research framework for selecting indicators to assess land management effects on ecosystem service provision. Based on Haines-Young and Potschin (2010), Kienast et al. (2009), De Groot et al. (2010a) and Hein (2010). The white boxes indicate the study's scope. Solid arrows indicate effects; dashed arrows indicate feedbacks.

Figure 2.1 shows the main elements of our framework: the driving forces, ecosystem, service provision, human wellbeing, 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, but we note that the interactions which we describe below can refer to ecosystems at multiple spatial scales, e.g. at plot, landscape, regional or even national scale (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 2005b). Although drivers like climate change or environmental pollution have an impact on the ecosystem, we only focus on the driving force land management. As described earlier, land management affects ecosystem properties and function (Kremen et al. 2007, Chen et al. 2011, Bastian et al. 2012), as well as the ecosystem service provided (O'Farrell et al. 2007, Edwards et al. 2011). 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 the 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. Examples of ecosystem functions include aerosols capture (Nowak et al. 2006) and carbon sequestration (Díaz et al. 2009). The ecosystem service contributes to human wellbeing, 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 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 (Riley 2000, De Groot et al. 2010b). 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 (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 (Schomaker 1997, Layke et al. 2012).

Finally, we considered data availability, credibility, and portability as other criteria. Data availability is especially essential if data and 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'

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.

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 (216000 inhabitants), 's-Hertogenbosch (140000), and Tilburg (200000) (CBS 2011). The area comprises intensively managed maize & grassland, rural settlements and patches of forest and heathlands (Figure 2.2). Due to its tranquillity, abundant forest patches and cultural historic elements, 'Het Groene Woud' offers many recreation opportunities (Het Groene Woud 2011). Agriculture has been an important economic activity in the area. A large part of the area is occupied by cropland (20%) and grassland (43%) (De Wit et al. 1999, Kuiper and de Regt 2007). A 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 is maintaining agricultural production while protecting biodiversity and increasing recreation opportunities (Petz and Van Oudenhoven 2012).

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)

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, 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 for ecosystem properties, function and service for each selected ecosystem service, and determined qualitative relations between them. Examples of these 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 CO_2 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 The Netherlands (Wesseling et al. 2008) and in the UK (Beckett et al. 2000, Bealey et al. 2007) .

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 relations (including feedbacks) between ecosystem properties, function and service. 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 (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 (Wesseling et al. 2008, Hein 2011).

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 their spatial scales and mapped the function indicators in order to spatially visualize the potential of the area for providing the service. We distinguished between landscape element, plot and landscape 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 plot scale 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 landscape scale.

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 twelve key indicators for ecosystem properties, nine for functions and nine for service provision (Figure 2.3).

Indicators for ecosystem properties could be 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.

The number of ecosystem properties indicators was highest. All functions depend on land cover and landscape structure, whereas vegetation characteristics influence all 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 properties indicators: water stored in vegetation to most (eight),

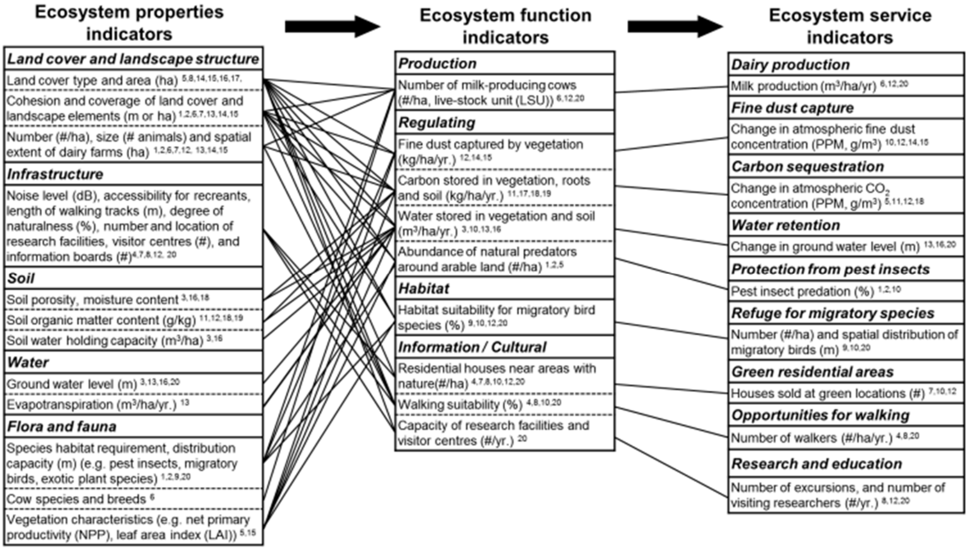

Figure 2.3: Overview of key properties, function and service indicators for nine ecosystem services in 'Het Groene Woud'. Units are given between parentheses. Lines indicate linkages between individual indicators. Typology of indicators is based on De Groot (1992), Kienast et al. (2009) and De Groot 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 (2009b); [13] Mulder and 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 Het Groene Woud (Accessed on January 20th, 2011, URL: *www.groenewoud.com*).

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, so 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: the example of three ecosystem services

Food provision: dairy production

Management for dairy production affects ecosystem properties, function and service provision (Figure 2.4). Applying pesticides and nutrients, the first land management indicator in Figure 2.4, influences several ecosystem properties. For instance, the net primary productivity (NPP) of grass can be enhanced by fertilizers (Jangid et al. 2008, Batáry et al. 2010). Veterinarian measures can influence the cows' capacity to produce milk through disease prevention and additional feeding. Mechanisation can affect the grassland area and farm size 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.

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 directly related 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 will have to be altered. This would require more nutrient application and mechanisation, increasing the number of cows or area of grassland or lowering the milk production.

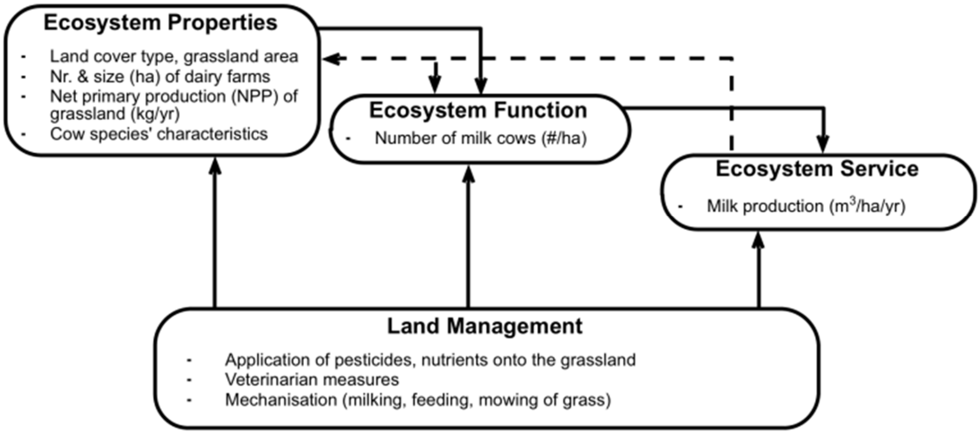

Figure 2.4: Framework with indicators for land management, ecosystem properties, function and services for the service milk production. Arrows indicate direct linkages between the boxes; the dashed line indicates feedback.

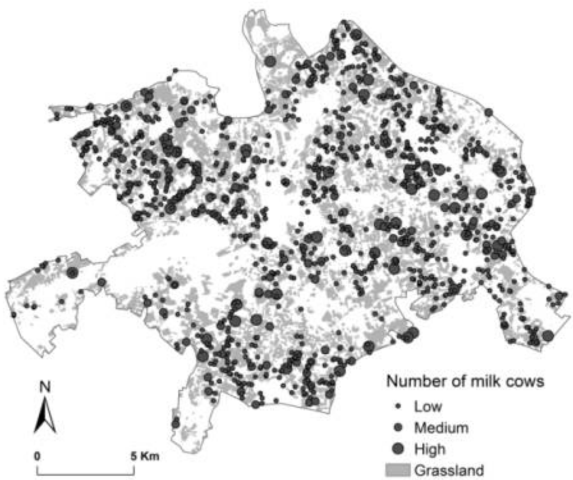

Figure 2.5: Map of 'Het Groene Woud', indicating where the service milk production can be provided. The service indicator 'number of milk cows' (dots) and function indicator 'area of grassland' (light grey area) were mapped. Land cover data by de Wit et al. (1999), milk cow data by Naeff and Schmidt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

The service dairy production is provided on grassland, which currently 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) scale, as its spatial pattern follows the allocation of the grassland across the landscape. Only a few parts of the area are currently 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 (Figure 2.6) involves selecting the location and planting (species choice) as well as maintaining forest plots and woody elements (Beckett et al. 2000, Oosterbaan et al. 2006, McDonald et al. 2007). 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 (Tonneijck and Swaagstra 2006).

Intensive agriculture and road traffic are the main fine dust emission sources in 'Het Groene Woud' (Oosterbaan et al. 2009). Local emission directly influences the amount of fine dust that can be captured by vegetation (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, such as point sources like 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).

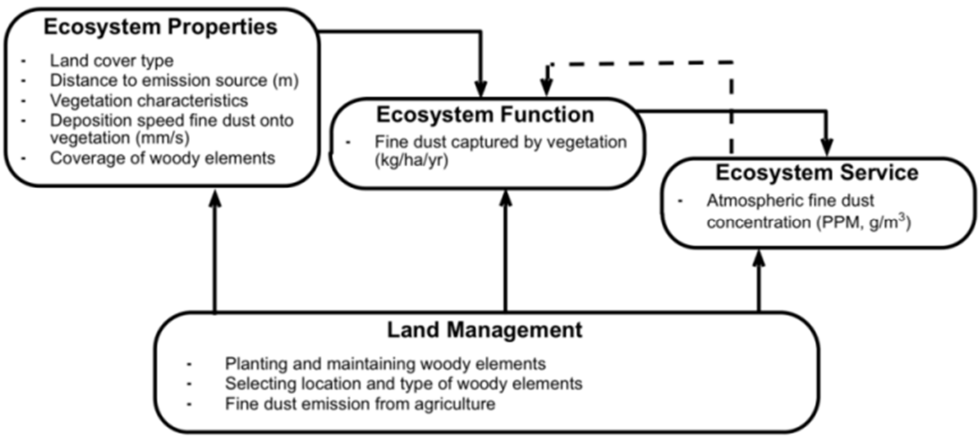

Figure 2.6: Framework with indicators for land management, ecosystem properties, function and service, for the regulating service fine dust capture. Solid arrows indicate direct linkages between the boxes; the dashed line indicates feedback.

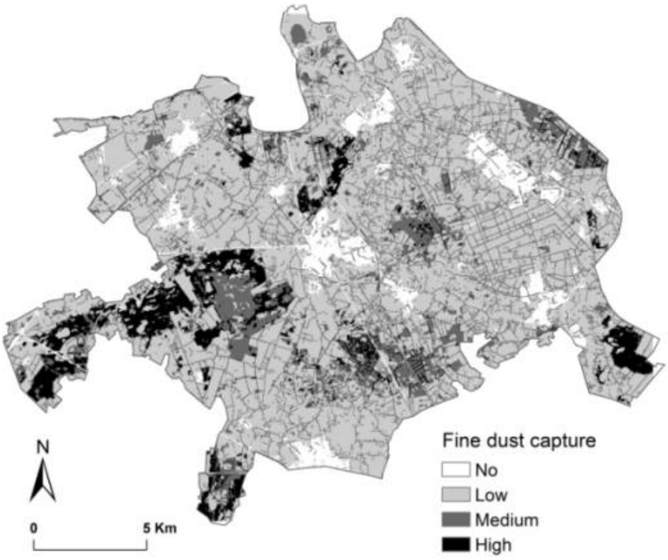

Figure 2.7: Map of 'Het Groene Woud', indicating where the service 'fine dust capture' can be provided. The function indicator 'fine dust captured' was mapped, based on the capacity of land cover, including woody elements, to capture fine dust. Forest areas (black) have a higher capacity to capture fine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

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 2.6. Fine dust capture can be measured on the scale of landscape element (e.g. tree-rows), plot (forest patch) and landscape (entire area). Figure 2.7 shows the spatial pattern of woody elements and forest plots across the landscape in 'Het Groene Woud'. The figure shows that all areas, except those with urban infrastructure (white on the map), contribute to fine dust capture in the area.

Opportunities for recreation: walking

Managing 'Het Groene Woud' to improve walking opportunities influences the area's ecosystem properties and functions (Figure 2.8Figure 2.). Developing and maintaining nature reserves, parks and green areas influences the area's degree of naturalness. It can also increase the length of walking tracks and the accessibility (Goossen and Langers 2000). Protecting and maintaining historical landscape elements improves 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 (Goossen and Langers 2000, De Vries et al. 2007).

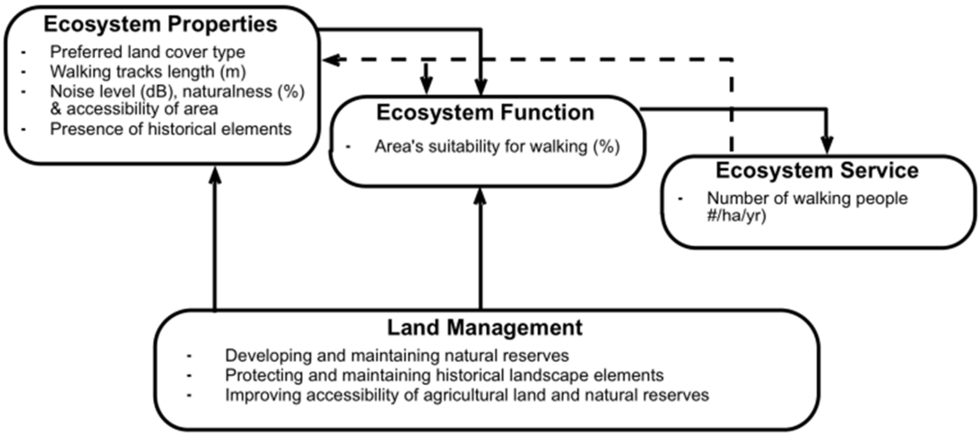

Figure 2.8: Framework with indicators for the land management, ecosystem properties, function and service boxes, for the cultural service opportunities for walking. Arrows indicate direct linkages between the boxes; dashed lines indicate feedbacks.

The 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 land over arable land, grassland or urban areas (Goossen and Langers 2000). The diversity of land cover is also highly appreciated by walkers (Van den Berg et al. 1998, De Vries et al. 2004).

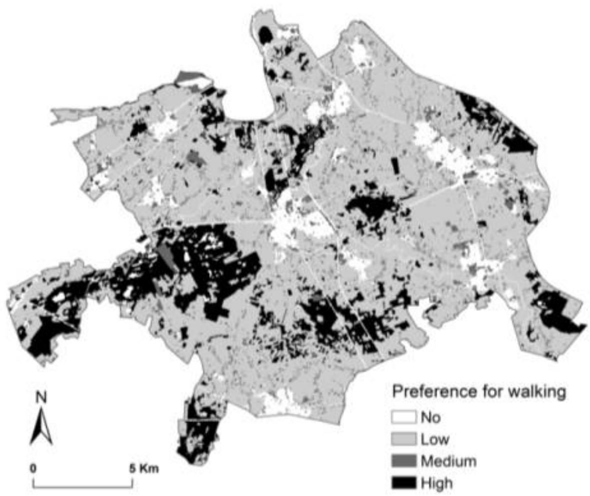

Figure 2.9: Map of 'Het Groene Woud', indicating where the service opportunities for walking can be provided. The properties indicator 'preferred land cover type for walking' was mapped. Forest areas (dark) are preferred most by walkers, compared to agricultural area (grey) and urban area (white). Recreation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

The actual service performance can be measured by the number of walkers (service indicator), which is directly related to the walking suitability. Naturally, an area with higher suitability is more likely to attract larger numbers of walkers (Goossen and Langers 2000, De Vries et al. 2004). 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 (entire region) scale. The map also indicates the distance from cities to potential walking areas. The majority of the area is either highly suitable or not suitable at all for walking.

2.4 DISCUSSION

2.4.1 Methods: framework and 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 wellbeing 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. (2009b), and others. However, our framework enables the assessment of how land management can affect ecosystems (state), and their services and human wellbeing (impact). This is among the biggest scientific challenges faced by scientists conducting ecosystem assessments (ICSU-UNESCO-UNU 2008, Carpenter et al. 2009).

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 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 (De Bello et al. 2009, Czúcz et al. 2011). Many studies have been carried out on impacts of land use on ecosystem services provision (Schröter et al. 2005, Fürst et al. 2010a, Richert et al. 2011, Barral and Oscar 2012) and on policy and land use planning in relation to ecosystem services (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) and the World Resources Institute (WRI) ecosystem services indicators database (Layke 2009b), 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-UNESCO-UNU 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 (UNEP-WCMC 2011, Layke et al. 2012). 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 properties 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 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, together also called state indicators, provide information on how much of a service an ecosystem can potentially provide in a sustainable manner (Layke 2009b, De Groot et al. 2010b). Service indicators, also called performance indicators, provide information on how much of the service is actually provided and/or used (Fisher and Turner 2008, Layke 2009b, De Groot et al. 2010b). 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 spatial scale services were provided and, additionally, at which spatial scale land management could affect the provision of these services. The consideration of multiple scales is important not only because service provision can occur at several scales, but also because the level of service provisioning and decision making might differ (Hein et al. 2006, Daily et al. 2009, Seppelt et al. 2012). The selected indicators could be linked to landscape element, plot, and landscape scale. Results showed that properties indicators and some function indicators could be linked to all three scales, whereas some function and all service indicators could only be linked to plot and landscape scales.

Our criteria (Section 2.2.2) 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 of the three spatial scales. 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.

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	Consi-stency	Compre-hensive	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 for 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

ACKNOWLEDGEMENTS

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Cows, pastures, cornfields, tree rows and forest elements make up this typically Dutch landscape. Cycling is the favourite Dutch recreation activity, and most of the 35.000 km of cycling tracks lead through landscapes that are 'tailor-made' for cyclists. Recreants' preferences have been extensively monitored in the Netherlands and landscapes like 'Het Groene Woud' are managed carefully, considering these preferences. Land managers are challenged to balance preferences for an attractive landscape with aims to maintain agricultural production and protecting nature.

3 MODELLING LAND MANAGEMENT EFFECTS ON ECOSYSTEM FUNCTIONS AND SERVICES: A CASE STUDY IN THE NETHERLANDS

ABSTRACT

Understanding the effects of land management on ecosystem services is essential for making management decisions. Current modelling approaches that aim to assist decision making generally do not distinguish between ecosystem functions and services, or include land management effects. Our objective was to model the effect of land management on multiple ecosystem services in 'Het Groene Woud', the Netherlands. Based on quantitative and spatial relationships, we mapped and modelled eight ecosystem functions and services. Next, three services 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 ecosystem services 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 ecosystem function and services maps can be used for regional management, because they provide location-specific quantitative information on ecosystems' capacity to provide services and on the service provision itself.

Based on:
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Corrigendum (2012). International Journal of Biodiversity Science, Ecosystem Services & Management 8 (3): 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). These changes have impaired the ecosystems' capacity to sustain food production and provide fresh water to humans, to provide a healthy habitat and shelter for animal and plant species, to regulate climate and air quality, and to prevent crops and humans to suffer from infectious diseases (MA 2005b, WRI et al. 2008). The contributions to human wellbeing by ecosystems are defined as ecosystem services (De Groot et al. 2010a). Over the years, evidence has mounted on the extent and value of ecosystem services provided globally (TEEB 2010b), as well as on their decline as a result of land management change and other drivers (Kremen et al. 2007, ICSU-UNESCO-UNU 2008). Land management is defined as the human activities that support land use and directly affect the land (Van Oudenhoven et al. 2012).

Information on the effects of management on ecosystem services is crucial for developing policies on sustainable land-use options (Nelson et al. 2009). However, quantitative empirical information on the capacity of a given ecosystem to provide multiple services is scarce and the biophysical characterization of ecosystem services is still not well established (Chan et al. 2006, Villa et al. 2009). One of the main challenges for current ecosystem services research is assessing the ecosystem service bundles provided through alternative management regimes (ICSU-UNESCO-UNU 2008, De Groot et al. 2010b).

Mapping and modelling can help to better understand the interactions between land management and ecosystem service provision (Daily et al. 2009, De Groot et al. 2010a). Mapping and modelling studies have largely focused on water, carbon sequestration, pollination, biodiversity, and recreation services (Egoh et al. 2008, Reyers et al. 2009, Maes et al. 2011). Most studies, however, do not distinguish explicitly between potential and actual service provision (i.e. ecosystem function and services) (Kienast et al. 2009, Lamarque et al. 2011). In most mapping and modelling studies, ecosystem services have been reduced to indicators with limited management and policy relevance (Willemen et al. 2008, Raudsepp-Hearne et al. 2010). Management alters ecosystem properties (i.e. the ecosystem's processes and structure) and, thus, changes land use and landscape structure and, consequently, also influences ecosystem services (Verburg et al. 2009, Van Oudenhoven et al. 2012). Current research often neglects land management effects on ecosystem services (Reyers et al. 2009, Hein 2010).

Therefore, our study aimed to model the effect of land management on multiple ecosystem services. We developed generic models in an *ArcGIS* spatial modelling environment, which we applied to 'Het Groene Woud', a typical Dutch landscape with many different land-use types and landscape elements. We used multiple indicators to quantify, map and model ecosystem services 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 ecosystem service provision under two simple management scenarios.

3.2 METHODS

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

'Het Groene Woud' (350 km^2) is located in the The Netherlands' southern province Noord-Brabant, amidst three densely populated cities: Eindhoven, 's-Hertogenbosch, and Tilburg (Figure 3.1). The three cities account for around 80% of the population of the region (roughly 650,000) (CBS 2011). 'Het 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. The main targeted sectors of the regional policy are agriculture, tourism/recreation, and nature, which have to be maintained, increased and conserved, respectively (Streekraad Het Groene Woud en De Meierij 2008, Het Groene Woud 2011).

In 2005, the area was declared a 'National Landscape', which means that new policies and initiatives should contribute to conserving the landscape's unique cultural-historical, natural and landscape features and not compromise local economic activities (Kuiper and de Regt 2007). The regional management strategy aims to improve landscape heterogeneity, multi-functionality and connectivity of green and blue landscape elements (Kuiper and de Regt 2007, Blom-Zandstra et al. 2010). Regional policy and management are closely linked through the local council, which 'translates' policy options into management plans (Streekraad Het Groene Woud en De Meierij 2008, Het Groene Woud 2011). Large segments of the area are included in the Dutch Ecological Main Structure and European Natura 2000 networks (Blom-Zandstra et al. 2010). Habitats and biodiversity are preserved through ecological zones (Bredenoord et al. 2011). The Kampina Nature Reserve is a biodiversity hotspot and important recreation area (Figure 3.1). The case study area was selected because of the link between policy and regional landscape management.

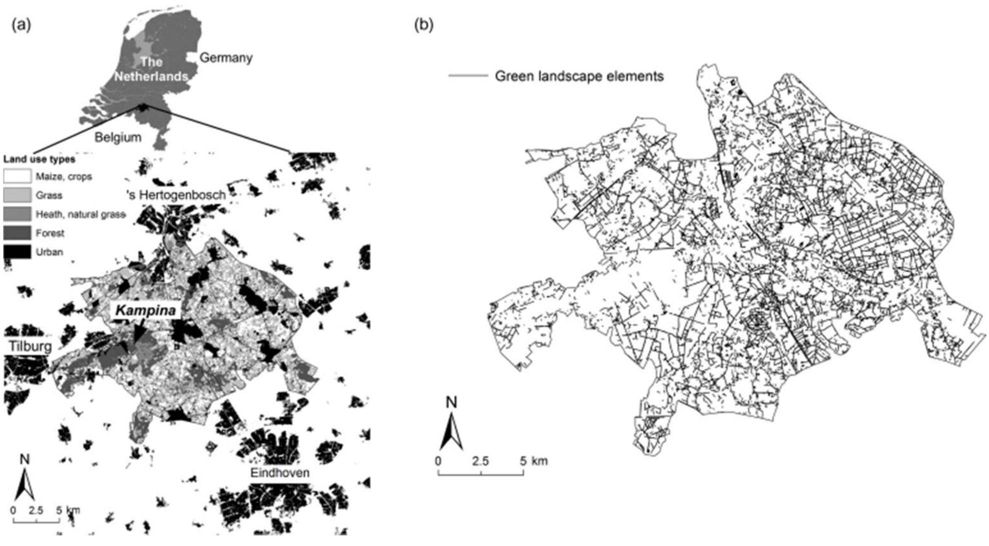

Figure 3.1: The two maps indicate the main land use types (a) and location of green landscape elements (b) in the study area. Data source: De Wit et al. (1999) and Grashof-Bokdam et al. (2009a).

3.2.2 Methodology

We used the following steps to quantify and model ecosystem functions and services: (1) selecting ecosystem services; (2) selecting indicators and quantifying functions and services; and (3) modelling functions and services. Finally, we also analysed how ecosystem services would change under alternative management scenarios. Our approach follows the stepwise framework introduced in Chapter 2 (Van Oudenhoven et al. 2012, see Figure 3.2 for the adapted version).

Selecting ecosystem services

We selected ecosystem services that had been mentioned by local sources (websites and brochures), stakeholders (regional council members, scientists and farmers) or the scientific literature and reports (e.g.Bianchi et al. 2008, Oosterbaan et al. 2009, Blom-Zandstra et al. 2010). In line with the TEEB typology (De Groot et al. 2010a), we selected food (milk), raw materials (fodder), air quality regulation (PM10), climate regulation (carbon sequestration), pollination, biological control, lifecycle maintenance and opportunities for recreation. The selected services represent all four ESS categories (provisioning, regulating, habitat and cultural) and reflect the three main sectors that are targeted by regional policy.

Indicator selection and quantification of ESFs and ESSs

We identified ecosystem properties, functions and services 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 (Niemeijer and de Groot 2008, Reyers et al. 2010, Van Oudenhoven et al. 2012). Examples of relevant land management components include land-use type, landscape pattern, crop type and noise level. Information on indicators and data was 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. A complete overview of all indicators and relationships can be found in Appendix 1.

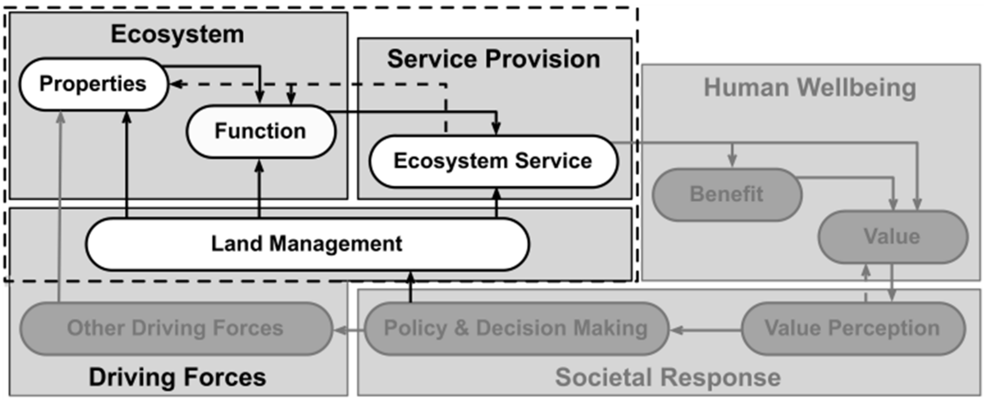

Figure 3.2: Framework for assessing land management effects on ecosystem services. The white boxes in the dotted box indicate the scope of our research. Solid arrows indicate effects; dashed arrows indicate feedbacks. Adapted from Van Oudenhoven et al. (2012).

Food production (milk): about 43% of the area is grassland used by dairy cows (De Wit et al. 1999, Kuiper and de Regt 2007). The amount of milk that can be produced (service indicator) is dependent on the grassland area in combination with the number of milk-producing cows (function). 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, but we did not quantify these inputs. We assumed that the area on which maize is cultivated determines the amount of maize that can be produced. 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 (McDonald et al. 2007, Hiemstra et al. 2008). PM10 is particulate matter with a diameter of 10 μm or less (Beckett et al. 1998, Bealey et al. 2007). Local agriculture and traffic account for 8% of the total PM10 emission (444 ton yr^{-1}) in 'Het 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 $ha^{-1} yr^{-1}$) captured by vegetation (function) decreases atmospheric PM10 concentration (Beckett et al. 1998, Bealey et al. 2007, McDonald et al. 2007). We only used the capture of vertically deposited PM10 as a function 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) was used. We adjusted this to the average PM10 concentration of 26 $\mu g/m^3$ 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 (service) 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 (Beckett et al. 1998, Bealey et al. 2007). 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 % (Stewart et al. 2002, Tonneijck and Swaagstra 2006, Bealey et al. 2007). 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 to local PM10 concentration reduction, due to the fact that 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 Mton CO_2, whereas agricultural grasslands emit 4.2 Mton CO_2 and urban areas emit 0.2 Mton CO_2 annually (Brandes et al. 2007, Schulp et al. 2008). The amount of carbon sequestered (function) leads to a decreasing atmospheric CO_2 concentration (service) (Schulp et al. 2008, Carol Adair et al. 2009). We used country-level carbon sequestration data (ton C $ha^{-1} yr^{-1}$) 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 for heath and natural grass too (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 (Gohar and Shine 2007, Environmental Protection Agency 2011)

Pollination: several crops, such as beets and various vegetables, depend on natural pollinators in 'Het Groene Woud' (De Wit et al. 1999). Pollination by wild bees is of great economic importance to farmers that cultivate pollinator-dependent fruits and vegetables (Priess et al. 2007, Gallai et al. 2009). The abundance of pollinators within a given proximity of croplands (function) affects crop yield (service) (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 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) (Steffan-Dewenter and Tscharntke 1999, Priess et al. 2007). The service itself, the crop yield can be provided only in areas with pollination-dependent crops. We assumed that the pollination service follows the pollinator abundance, which means that at the maximum fruit set of 60% the yield is 100%.

Biological control: many crops that are grown in 'Het Groene Woud', such as wheat, maize and various vegetables, can be severely affected by pests, mainly insects (Gurr et al. 2003, Bianchi et al. 2006). We considered biological control the predation of insect pests by natural predators. The abundance of natural predators (function) can cause decreasing numbers of pests (service) and thereby decrease damage to crops (Foster et al. 2004, Clough et al. 2007, 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 service indicator for biological control. Bianchi et al. (2006, 2008) 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*) (Bianchi et al. 2005, Baveco and Bianchi 2007) 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: 'Het Groene Woud' 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 (function) is related to the occurrence of species (service). 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, through establishing 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 (function) largely determines how many people can walk (service). 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 (Van den Berg et al. 1998, Goossen and Langers 2000, De Vries et al. 2007). 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 in areas with a walking suitability of at least 60%.

Ecosystem function and service modelling

The above-described relationships served as a base for modelling each function and service:

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 an example of the climate regulation model overview. The LGN3+ 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. The resolution of all maps was 25 x 25 m. Appendix 1 provides an overview of the indicators used and relationships between them.

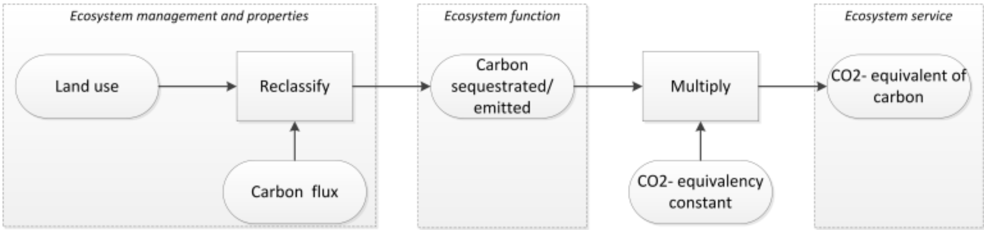

Figure 3.3: Schematic overview of the climate regulation model. Round boxes indicate inputs/outputs, and square boxes indicate processes or tools used in ArcGIS 9.3 to derive outputs.

Scenario analysis: shift to extensive or intensive land management

We developed two scenarios: (1) *Intensive agriculture* and (2) *Functional nature protection*. We quantified milk production, PM capture and recreation opportunities under the two scenarios. Our scenarios were based on the *Suitable Nature* and *Functional Nature* scenarios developed by PBL (2011) as part of the Dutch 'Nature Outlook'. The two scenarios were based on the main land use types (cropland, grassland and forest), sectors mainly targeted 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 and local PM10 emission (Table 3.1).

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 assumed that existing green landscape elements would be maintained but not further expanded. This is in line with the *Functional Nature* scenario, which assumes increased involvement of local stakeholders in decision making and increased awareness of and attention to the benefits of nature, both in financial and non-monetary terms (PBL 2011). We

Table 3.1: Land management characteristics of 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 (3100 ha) converted into grassland (16,500 ha in total)	No changes in land use areas (14,400 ha grass, 5400 ha forest)
Conventional milk production (→ 8000 L milk cow^{-1} yr^{-1})	Switch to organic milk production (→ 6600 L milk cow^{-1} yr^{-1})
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 (~5100 ha) 31% vegetation cover

therefore assumed no changes in PM10 emissions, in the total area of different land use types, and in the coverage of green landscape elements.

We quantified the three services under the two scenarios using the management variables described in Table 3.1, the relationships specified above and quantitative outputs of our ecosystem service models.

3.3 RESULTS

3.3.1 Modelled ecosystem functions and services

In this section, numbers and maps are shown for eight quantified and modelled ecosystem functions and services. We only provide separate maps of function and services if the spatial patterns of the function and service maps differed.

Food production (milk)

Around 14,400 ha of grassland provide grazing area for 7200 milk cows (Figure 3.4a). A conventionally kept cow can produce 8000 L of milk per year (LEI and CBS 2010) and an organic cow 6600 L (LEI and CBS 2010). Based on that, roughly 57,600 kL of 'conventional' and 47,520 kL of organic milk are produced yearly from the cows that feed on grass.

Raw materials (fodder)

Maize is cultivated on 5500 ha (Figure 3.4b). The average silage maize yield in 2010 was 45 ton ha^{-1} (CBS 2011), which means that 250,000 ton yr^{-1} maize is produced in 'Het Groene Woud'.

Air quality regulation

Annual PM capture by coniferous forests, deciduous forests and heathland, natural grass and green elements is 94kg (high), 54 kg and 27 kg ha^{-1} yr^{-1}, respectively. The other land-use types capture less than 15 kg ha^{-1} yr^{-1} (low) and we assumed that urban areas capture no fine dust (Figure 3.4c). In total, 644 ton PM10 can be captured by vegetation annually, which means that the total amount of PM10 emitted within the area (444 ton) can be captured by vegetation. The 31% vegetation cover (forest, heath, natural grassland and green elements) in 'Het Groene Woud' is estimated to contribute to a 10-15% reduction of the local PM10 concentration.

Climate regulation

Carbon sequestration rates for grassland (0.18 ton C $ha^{-1} yr^{-1}$), cropland (-0.25 ton C $ha^{-1} yr^{-1}$), urban area (0), forest, heath and natural grass (1.1 ton C $ha^{-1} yr^{-1}$) were used, where negative numbers indicate carbon emission (Kuikman et al. 2003, Schulp et al. 2008) (Figure 3.4d). The corresponding CO_2-equivalents of the carbon sequestrated or emitted are 0.66, -0.92, 0, 4.037 ton CO_2-equivalent, 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, so 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).

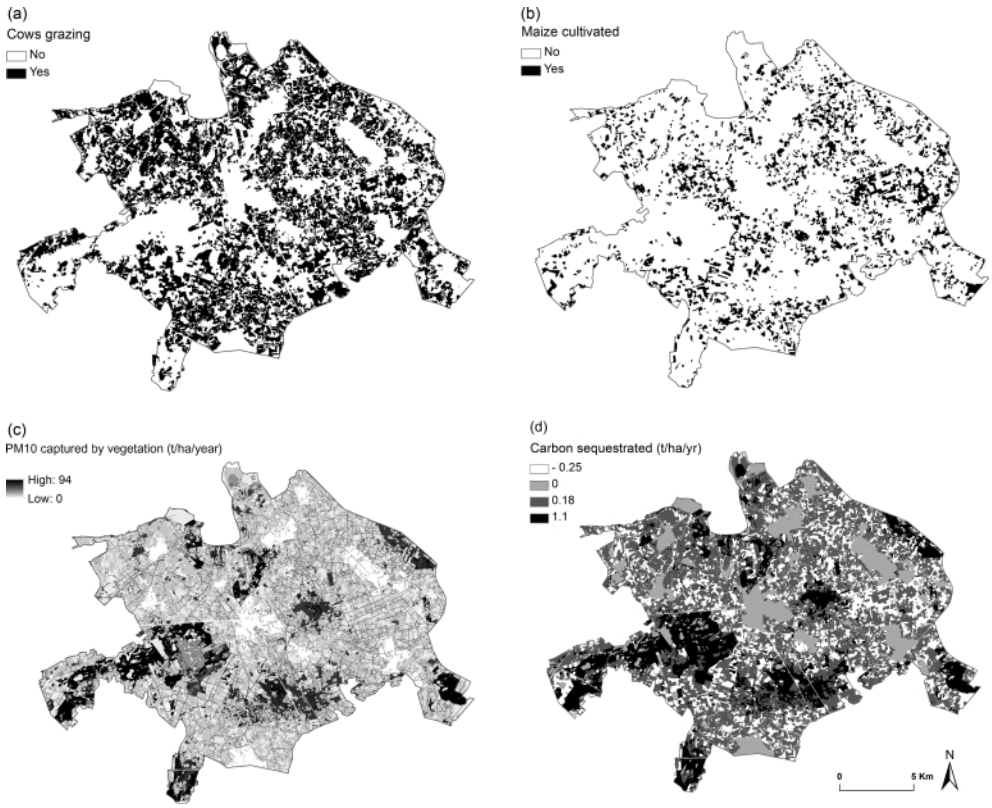

Figure 3.4: Ecosystem function maps for four ecosystem services: milk production (a), fodder production (b), air quality regulation (c) and climate regulation (d). Ecosystem service maps are not included for these services, because they show a similar spatial pattern to the ecosystem function maps.

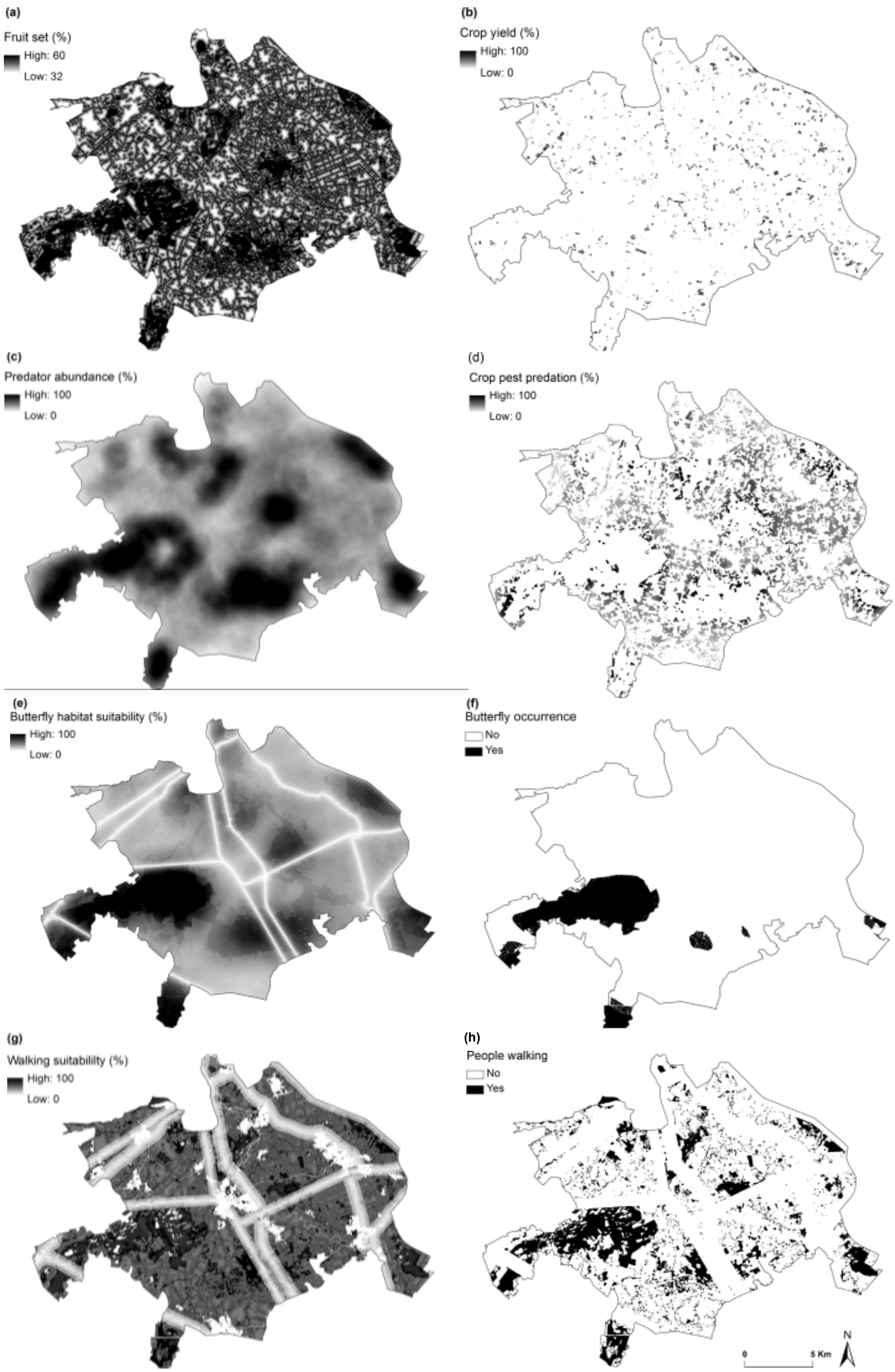

Figure 3.5: Maps of modelled pollination function (a) and (b), biological control function (c) and service (d), lifecycle maintenance function (e) service (f), and opportunities for recreation function (g) and recreation service (h).

Biological control

Pest predation in croplands follows the abundance curve of natural predators of insect pests, with highest predation possible in croplands near forests and green elements (Figure 3.5c and d). The service is only provided in areas that can be affected by agricultural pests.

Lifecycle maintenance

Butterflies occur in closed woody habitats and live primarily in non-fragmented forests in 'Het Groene Woud'. 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

Combined forest and heathlands with low noise levels provide the highest suitability for walking (100%), whereas noisy areas along roads and railways are the least suitable (0%) (Figure 3.5g). About 60% of the local residents walk regularly, which amounts to 390,000 people. Assuming that people walk only in areas with at least 60% suitability, walking would occur at 19% of the area (6265 ha). This leads to a walkers' density of 62 ha^{-1}. 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 'Het Groene Woud' (Figure 3.5h).

3.3.2 Scenario analysis: shift to extensive or intensive land management

The outcomes of the *Intensive agriculture* and *Functional nature protection* scenarios were quantified for milk production, air quality regulation and opportunities for recreation (Table 3.2).

More milk could be produced under the *Intensive agriculture* scenario (66 ML yr^{-1}) as compared with the *Functional nature protection* scenario (47.5 ML yr^{-1}). This is the result of the increasing grassland area (15%) and the larger number and higher productivity of conventionally kept cows (8250) compared with organically kept cows (7200). More PM10 could be captured (644 vs. 359 ton yr^{-1}) and the area with high walking suitability (above 60%) is largest in *Functional nature protection* (6362 vs. 4360 ha). The higher PM10 capture in *Functional nature protection* is caused by maintaining green elements and forest areas, and constant PM10 emissions. All locally emitted PM10 (444 ton yr^{-1}) could be captured. We assumed that a 10-15% decrease of local PM10 concentration can be achieved by 25% vegetation cover (Stewart et al. 2002, Bealey et al. 2007). *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. *Functional nature protection* would result in larger area with high walking suitability than *Intensive agriculture*, which 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^{-1} and in *Functional nature protection* is 61.3 ha^{-1}.

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

Ecosystem service	Function/ service	Scenario 1. Intensive agriculture	2. Functional nature
Milk production	Function	8250 milk cows (conventional)	7200 milk cows (organic)
	Service	66 ML milk yr^{-1}	47.5 ML litre milk yr^{-1}
Air quality regulation	Function	396 ton yr^{-1} PM10 captured	644 ton yr^{-1}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	390,000 walkers 89.4 walkers ha^{-1}	390,000 walkers 61.3 walkers ha^{-1}

3.4 DISCUSSION

3.4.1 Modelling the effects of land management on ecosystem services

Indicators and methods for modelling ecosystem services

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

Similar 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 storage or 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 function to service indicators or air quality to other service. We could not directly link data on fine dust capture capacity to changes in atmospheric PM10 concentration, but assumed this relationship based on the literature. Although research has shown 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 'Het Groene Woud' and stated that NH_3 was an uncertain component to be modelled at landscape scale, as a result of heavily fluctuating local 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 high uncertainties involved we did not use this information.

Pollination and biological control were modelled before with agent-based models and hence with a focus on animal behaviour (Kremen et al. 2007, Lonsdorf et al. 2009, Kareiva et al. 2011). 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 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 highly crop-specific.

Lifecycle maintenance can be measured and modelled through species richness (Chan et al. 2006), mean species abundance (Alkemade et al. 2009), and habitat rarity and habitat integrity (also referred to as fragmentation) (Tallis et al. 2011). Similar to Tallis et al. (2011), we established quantified and distance relationships between land management and ecosystem properties related to habitat suitability for mapping lifecycle maintenance. Scientific literature only supports the positive effect of nature protection on species (habitat). We assumed a 20-30% habitat suitability increase due to nature protection for this case study. The indicator choice also determined 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 (Figure 3.5a and e).

Recreation was measured and modelled before through involving 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 a proxy for recreation, due to the activity's popularity in the study area. We studied recreation rather than tourism, because walking trips would be regarded as tourist activities if a night were spent in an accommodation in the area (Henkens et al. 2005, CBS 2010). Therefore, motives and indicators for tourism could be different than for recreation. A diverse landscape has a positive effect on recreation (Van den Berg et al. 1998). Nevertheless, the 10-30% walking suitability increase due to high landscape diversity was an assumption made for this case study. Furthermore, we did not consider other aspects of landscape diversity, such as topography and water ways.

Quantified outputs of ecosystem services models

The function and service maps provide location-specific information about the effect of land management on ecosystem service provision. The reliability and accuracy of the models and uncertainty of the results depend on the quality of the input data and relationships. For example, information on fodder production was directly derived from statistics of maize production. We used national aggregated, yearly updated statistics, which give a rough approximation of fodder production. Using regional, location-specific data might have led to more accurate results. Similarly, the climate regulation function map is directly derived from country-level land use-specific carbon sequestration data. Carbon sequestration by different land-use types shows a similar trend with the results of similar studies (Chan et al. 2006, Swetnam et al. 2011), namely that forests sequester the highest amount of carbon compared to other land covers. For milk production we compared the modelled number of cows (7200) with 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 'Het Groene Woud' than the provincial average we used and, therefore, more cows can be kept in reality. Although the 165 ton km^{-2} average milk production in 'Het Groene Woud' (calculated as non-organic milk produced/total area) is relatively low, it falls within the 100-500 ton km^{-2} range indicated on the national milk production map (in 2008) (van Oostenbrugge et al. 2010).

For air quality regulation and climate regulation, we mapped ecosystem functions by using land-use specific data of PM10 capture and carbon sequestration. The reliability and accuracy of these results also depend on the quality of input data. The actual contribution of PM10 capture to a lower PM10 concentration and the actual contribution of carbon sequestration to a lower CO_2 concentration were difficult to estimate. In other words, it proved to be difficult to make the link to what we defined as the actual service. 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 comparable area to 'Het Groene Woud' (a 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 them up 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.

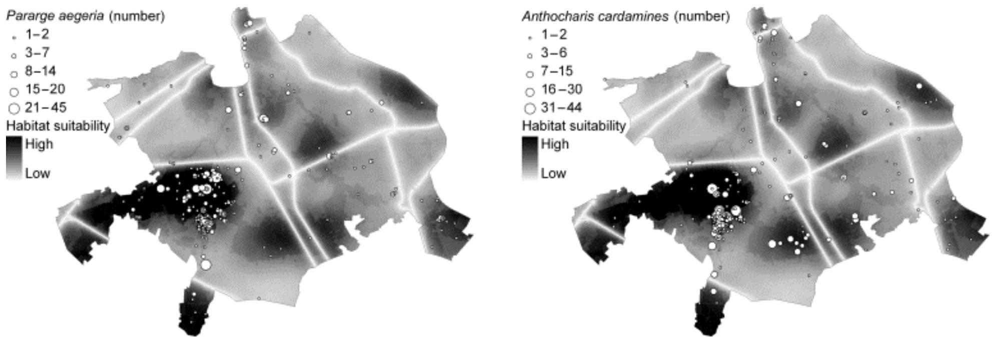

Figure 3.6: Empirical data on the occurrence of *Pararge aegeria* (left) and *Anthocharis cardamines* (right) overlaid with the modelled habitat suitability. The majority of butterflies of both species (64% and 58%, respectively) were found in areas with suitability higher than 50% (black).

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

We compared the walking suitability map with a national map on attractiveness for walking (Goossen and Langers 2000) and a general attractiveness map of Dutch landscapes simulated with the GLAM-2 (second version of a GIS-based landscape appreciation model) (De Vries et al. 2007). The Kampina Nature Reserve scores the best in all three studies. On our walking suitability map the negative effect of roads and railways is much more visible (Figure 3.5g and Figure 3.7). These similarities and differences can be attributed to the assumptions used in the methodology, as well as the indicator choice. Common indicators were land-use preference and noise level (Goossen and Langers 2000, De Vries et al. 2007). 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%. Our analysis was done at a higher resolution than used in GLAM-2 (De Vries et al. 2007) and by Goossen and Langers (2000). About 75% of all walks occur within 20 km from dwelling places (CBS 1997), which implies that the entire area is attractive for walking.

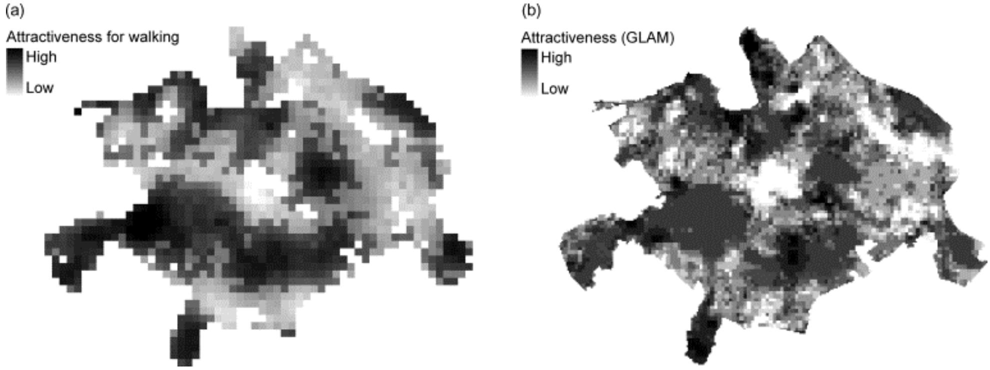

Figure 3.7: Maps of attractiveness for walking (a) based on indicators from Goossen et al. (1997), and Goossen and Langers (2000); and landscape attractiveness (b) based on the GLAM-2 model by (De Vries et al. 2007).

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. Performing a uniform uncertainty assessment on all services is generally difficult, because the methods to assess validity and uncertainty differ per service.

3.4.2 Scenario analysis

Our 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 ‘Het Groene Woud’. Protecting and increasing the extent of green landscape elements seems realistic, considering the current regional spatial policy. The area’s current landscape configuration is the result of a local, bottom-up initiative: nature managers, farmers, and municipalities worked 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).

The *Intensive agriculture* scenario results in high milk production, but at the cost of recreation and air quality regulation. The high recreant density in only a limited area suitable for walking would be undesired for local stakeholders and other 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, 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 management-related variables for the three ecosystem services, and consequently illustrating how this stepwise modelling approach can facilitate making land management decisions. We showed that land management for the optimization of one service has an effect on multiple services, because management often targets and alters ecosystem properties, such as 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 ‘Het 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 multi-functionality, connectivity of green landscape elements and the full implementation of the EHS network are target points of the current and future regional management strategy (Kuiper and de Regt 2007, Opdam et al. 2009, Blom-Zandstra et al. 2010). 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 green landscape elements play an important role in the provision of multiple ecosystem services. Therefore, a 10% increase of green elements (which could be done if the local council agrees) could contribute to increase landscape multi-functionality and service provision in 'Het Groene Woud'.

3.5 CONCLUSION

Our study's aim was to model the effect of land management on multiple ecosystem services. We used multiple indicators to quantify, map and model ecosystem services in 'Het Groene Woud'. Our maps of ecosystem function and services show a clear trade-off between services 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 service provision, 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 is essential to better understand land management effects on ecosystem service provision and is a first step towards bundling services. Our scenario analysis offered a preview of how this bundling can be done in a simple way, while still yielding useful results. The societal relevance of the study lies in its implication in regional management and policy. Further research in 'Het Groene Woud' and similar landscapes should focus on assessing more dynamic aspects services, for instance water and nutrient (nitrogen, phosphorus and carbon) balances. These balances are 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 and integrative approach. Therefore, we suggest combining the stepwise approach we applied with dynamic and qualitative approaches to get a more complete overview of the bundle of ESSs that can be provided.

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Aquaculture ponds make up much of Java's coastline. Mangroves have often been perceived as 'wastelands' by land managers and decision makers and, as a result, thousands of hectares of mangroves have been converted for aquaculture. Analysing and communicating the 'true value' of mangroves can contribute to more informed management decisions. Important ecosystem services associated with mangroves include coastal protection, raw materials, carbon storage and, ironically, fish and shrimp provision.

4 EFFECTS OF DIFFERENT MANAGEMENT REGIMES ON MANGROVE ECOSYSTEM SERVICES IN JAVA, INDONESIA

ABSTRACT

Over half of Indonesia's mangroves have been degraded or converted to aquaculture. We assessed the consequences of management decisions by studying the effects of different management regimes on mangrove ecosystem services in Java, Indonesia. Our novel typology of management regimes distinguishes five categories: *natural*, *low intensity* and *high intensity use* mangroves, mangroves *converted for aquaculture* and *abandoned aquaculture* systems. Eleven specific management regimes were developed, based on their legal status, management indicators and ecological characteristics. Seven ecosystem services were selected: food, raw materials, coastal protection, carbon sequestration, water purification, nursery for fish and shrimp, and nature-based recreation. We reviewed key ecosystem properties underpinning service provision and identified state and performance indicators. Ecosystem service provision was estimated and scored for each management regime by relating the regimes' ecological characteristics with ecosystem service indicators. *Natural* mangroves scored highest for most services, except for food. High fish and shrimp production by *aquaculture* regimes occurs at the expense of other ecosystem services. Rehabilitating *aquaculture* systems into *plantations* and *silvo-fisheries* reverses this loss, while still providing shrimp or raw materials. Transitions between management regimes were illustrated to show consequences of management decisions. Our findings can assist local decision makers to make better informed management decisions.

Based on:

A.P.E. van Oudenhoven, A.J. Siahainenia, I. Sualia, F.H. Tonneijck, S. van der Ploeg, R.S. de Groot, R. Alkemade and R. Leemans (2014). Effects of different management regimes on mangrove ecosystem services in Java, Indonesia. *Submitted*

4.1 INTRODUCTION

Indonesia has the largest extent of mangroves in the world (Spalding et al. 2010). Mangroves occur in tidal forests and include both the trees and their ecosystems (Alongi 2002, Giesen et al. 2006 and others). In this paper, 'mangroves' are ecosystems dominated by mangrove vegetation. Mangroves are progressively pressured by humans and their socio-economic developments. The Indonesian mangroves' extent declined by a quarter from their original 4.5 million hectares extent in the 1980s (Giesen et al. 2006, Spalding et al. 2010). Mangroves are mainly converted into aquaculture (Giesen et al. 2006), although expansion of urban areas and agriculture (including oil palm and rice paddies), coastal erosion and timber extraction also contribute (Giesen et al. 2006, Walters et al. 2008). The construction of aquaculture ponds is often fuelled by governments, private sector investments and development agencies, like the World Bank and the Asian Development Bank (Walters et al. 2008).

Scientists and non-governmental organisations have often emphasised the importance of mangroves to humans and the consequences of mangrove conversion (e.g. Rönnbäck 1999, Barbier et al. 2011). Ignoring important mangrove ecosystem services and their values in policy and management decisions is the major reason for the continuing mangrove conversion and degradation (Barbier et al. 2011). Ecosystem services are the contributions of ecosystems to human wellbeing (TEEB 2010a). Considering the (economic) consequences in terms of ecosystem services gained or lost is critical because most ecosystems, and especially coastal ecosystems, face the risk of being converted to provide other marketed services (Chan et al. 2011). Ecosystem services of mangroves include fuel wood and timber, food, coastal protection and nursery for fish and crustaceans (Rönnbäck et al. 2007, Barbier et al. 2011). Rather than quantifying ecosystem service provision in non-monetary terms (e.g. biophysical, intrinsic values, human dependence), the economic value of ecosystem services is usually emphasised (Schröter et al. 2014b). Economic valuation of ecosystem services offers insight into their potential values (e.g. Barbier et al. 2011, Brander et al. 2012) but generally ignores differences in biodiversity and other environmental and socio-economic properties. Economic valuation could benefit from quantifying the complex interactions between and effects of human activities on ecosystem processes and their services (Barbier et al. 2011).

The economic valuation literature, however, largely ignores the effects of management activities and the consequent land uses on mangrove ecosystem services (Rönnbäck 1999, Barbier et al. 2008). Management activities determine land use and directly affect land cover, i.e. natural vegetation, soils, cropland, water and human structures (Van Oudenhoven et al. 2012). Examples of management activities include fishing, replanting mangrove trees, aquaculture and constructing ecotourism facilities. Land use refers to the purpose of management activities (e.g. fish production, timber production, conservation) and can be influenced by legislation, socio-economic development, local traditions etc. (Verburg et al. 2013a) Management regimes are defined as 'the bundle of human activities that serve land-use purposes'. Knowing the effects of management regimes on mangrove ecosystem services is important to inform policy makers and land managers, and allow them to better plan and manage land use. Empirical evidence is needed to support management because many management assumptions have not been tested or verified (Carpenter et al. 2009).

This study assesses the consequences of management decisions in Java's mangroves, Indonesia, by studying the effect of different management regimes on mangrove ecosystem services. Java was chosen because this island is heavily impacted by management activities and different land uses, and many government decisions are first implemented here. Our extensive literature review characterized key indicators for seven key mangrove services: food, raw materials, coastal protection, carbon sequestration, water purification, nursery for fish and shrimp, and nature-based recreation (Section 4.3). We developed a typology of eleven management regimes divided over five broad categories. This typology applies to mangrove systems in the context of Indonesian legislation and Javanese management practices and ecological characteristics (Section 4.4). We related indicators of management regimes and ecosystem services. The consequences of each management regime for ecosystem service provision are assessed and compared in Section 4.5. Our broader discussion and conclusion in Section 4.6 and 4.7 should stimulate local managers and Indonesian decision makers to make better informed management decisions.

4.2 METHODS

4.2.1 Research framework

Many factors influence management decisions, but policy and decision making is the most important factor (Figure 4.1). A stepwise, iterative review of Indonesian policy documents and the literature provided the insights on management activities in mangroves (Section 4.2.2). Driving forces other than management (e.g. climate and seasonality) are also considered for some services. Driving forces affect ecosystem properties, which underpin service provision. The typology of management regimes helps to comprehensively understand the linkage between all these factors (Figure 4.1). Section 4.2.3 describes the assessment of ecosystem properties underpinning, and 'state' and 'performance' indicators of ecosystem service provision. Finally, Section 4.2.4 describes the relation between management regimes, the other drivers and ecosystem service indicators.

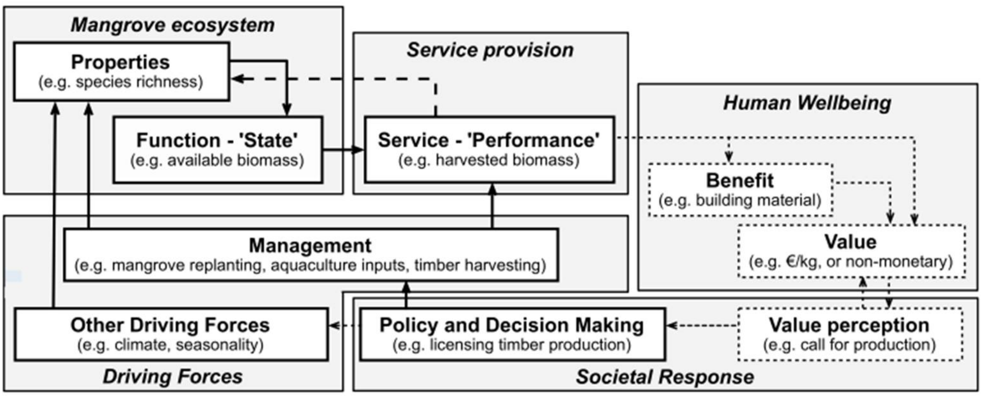

Figure 4.1: Research framework, adapted from Chapter 2 (Van Oudenhoven et al. 2012). Examples between parentheses refer to raw materials provision. Solid arrows indicate direct linkages; dashed arrows indicate feedbacks. Boxes and arrows with dotted outlines were not considered in the analysis.

4.2.2 Developing the management regime typology

The typology of mangrove management regimes was based on the scientific literature and Indonesian legislation, which ensured consistency with the Javanese context. Various global land-use and conceptual studies (e.g. Alkemade et al. 2009, De Groot et al. 2010b, Verburg et al. 2013a) classify ecosystems into natural, intensively used, converted and abandoned, or variations thereof, but their classifications are generic, consider few management activities and have not been developed for coastal ecosystems. Five broad categories of management regimes were first defined, by combining the reviewed classifications with examples from studies in Indonesia and South-East Asia (references in footnotes of Table 4.3). We developed specific regimes based on a) policy status, b) management activities and aquaculture indicators, and c) ecological characteristics (Table 4.1). Eleven resulting management regimes are further described in Section 4.4 and Table 4.3 and 4.4.

To ensure that the management regimes would apply to the Javanese context, we conducted a rapid field assessment between December 2012 and January 2013 in Pemalang, Banten Bay and Banyuwangi. We conducted interviews with experts, local stakeholders, district government representatives and scientists. We observed species richness, mangrove age, height and related indicators in mangrove systems and aquaculture inputs and harvests in the other regimes. Only *eco-certified aquaculture* and 'ideal' *silvo-fishery* regimes were not present. The field assessment helped to confirm our assumptions on management activities and aquaculture management indicators, and ecological characteristics of each management regime.

Table 4.1: Proposed characteristics of mangrove management regimes. Management activities and biophysical characteristics in italics were ignored for our typology of management regimes.

A) Policy status	Source
Jurisdiction of an area; Ministries of Forestry, Fishery, Agriculture or district bureau of Spatial planning	Forestry act No. 41/1999, 'Guidelines for (..) management models' (GMMM) by the Ministry (Min.) of Forestry (2012), Presidential Decree No.73/2012, Sualia et al. (2013)
Ownership status of an area	Peña-Cortés et al. (2013), Sualia et al. (2013)
Targeted ecological and/or economic function	Forestry act No. 41/1999, GMMM by Min. of Forestry (2012), Presidential Decree No.73/2012, Sualia et al. (2013)
Activities that are allowed or forbidden	Government regulation (Reg.) No. 28/2011, Min. of Forestry Reg. No. 3/2004, Sualia et al. (2013)
B1) Management activities	**Source**
Fishing (with nets, lines, boats)	Gilbert and Janssen (1998), Manson et al. (2005)
Hunting (monkeys, birds)	Sualia et al. (2013), Walters et al. (2008)
NTFP harvesting	Forestry act No. 41/1999, GMMM by Min. of Forestry (2012), Presidential Decree No.73/2012, Walters (2004)
Timber harvesting	Forestry act No. 41/1999, GMMM by Min. of Forestry (2012), Walters (2004, 2005b), Sualia et al. (2013)
Construction of recreation facilities	GMMM by Min. of Forestry (2012), Knight et al. (1997)
Recreational visits by tourists	Gilbert and Janssen (1998)
Replanting of mangrove	Forestry act No. 41/1999, Min. of Forestry Reg. No. 3/2004, Presidential Decree No.73/2012, Sualia et al. (2013)
Aquaculture effluent and waste disposal	Knight et al. (1997), Primavera et al. (2007)

B2) Aquaculture management indicators	**Source**
Natural or artificial stocking	Gilbert and Janssen (1998)
Artificial fertilizer, pesticide, antibiotics use	Gautier (2002), Rönnbäck (2001)
Stocking density	Gautier (2002), Rönnbäck (2001)
Size of aquaculture ponds	Gautier (2002), Primavera et al. (2007), Rönnbäck (2001)
Water exchange technique	Kusmana et al. (2008), Primavera et al. (2007)
Natural or artificial feed	Gilbert and Janssen (1998), Rönnbäck (2001)
Aeration of aquaculture ponds	Kusmana et al. (2008)
C) Ecological and biophysical characteristics	**Source**
Number of true mangrove species	Snedaker and Lahmann (1988), Primavera (1998),
Average diameter at breast height (d.b.h)	Komiyama et al. (1996)
Maximum height of mangrove trees	Bengen (2003), Komiyama et al. (2008)
Maximum age of mangrove trees	Berger and Hildenbrandt (2000), Clough et al. (1997)
Maximum perimeter of mangrove trees	Manson et al. (2005), Mumby et al. (2004)
Maximum root length of mangrove trees	Farnsworth and Ellison (1996)
Undergrowth	Matthijs et al. (1999)
Nr. of seedlings and saplings	Clarke and Allaway (1993), Primavera (1998)
Temperature of substrate, water	Middelburg et al. (1996)
Soil substrate	Middelburg et al. (1996), Schrijvers et al. (1995)

4.2.3 Indicator selection for mangrove ecosystem services

Seven mangrove ecosystem services were selected based on their relevance for Indonesian policy and stakeholders in Java: food (i.e. fish and shrimp), raw materials, coastal protection, carbon sequestration, water purification, nursery for fish and shrimp, and nature-based recreation.

A literature review was conducted for each ecosystem service to determine key ecosystem properties, and 'state' and 'performance' indicators. Ecosystem properties are the set of ecological and biophysical conditions, processes and structures that underpin the ecosystem's capacity to provide ecosystem services (Van Oudenhoven et al. 2012). We only collected information on ecosystem properties that were studied in the context of ecosystem service assessments. Indicators for the 'state' and 'performance' (see Figure 4.1), respectively, indicate the ecosystem function or capacity to provide services, and the actual service provision (De Groot et al. 2010b, Van Oudenhoven et al. 2012). We also included other drivers and non-ecological factors that determine service provision in our overview, which is provided in Section 4.3. We collected recurring information from the ecosystem services literature (i.e. confirmed by multiple sources) rather than conducting an exhaustive review. Review papers on mangrove ecosystem services were consulted first and further information was obtained from their references. This information was furthermore verified by reviewing citing studies. References are provided in footnotes of Table 4.2.

4.2.4 Analysing ecosystem service provision per management regime

We related information on management activities, aquaculture management indicators and ecological characteristics (Table 4.1) with underpinning ecosystem properties and state and performance indicators for ecosystem service provision (Table 4.2).

Although few ecosystem service studies explicitly mentioned 'management regimes', we used descriptions of ecology and management from study-site descriptions for assigning them to a management regime. We always considered both ecological characteristics and management indicators when assessing ecosystem service provision per management regime. Quantitative results were preferred, but these were rarely available for all regimes. Moreover, qualitative information proved more reliable and consistent for especially regulating services. We specify per service if information on state and/or performance indicators could be related to management regimes. When multiple sources provided quantitative information, the full range of possible outcomes was presented. Results were interpolated for regimes with no or little quantitative data from their adjacent regimes. A full description of ecosystem service provision for all management regimes is provided in Appendix 3, including references.

To compare the service provision per management regime, quantitative and qualitative information on ecosystem services was integrated using a scoring system ranging from -3 to +3. Scores were related to the highest and lowest possible result for each ecosystem service. Negative scores indicate disservices resulting from a certain management regime, such as CO_2 emission instead of sequestration, water pollution instead of purification and increased flood risk instead of coastal protection. The robustness of the results was also determined based on availability of multiple sources and multiple indicators, and applicability to the Javanese management and ecological context. Results that were interpolated, based on few indicators, weakly linked to management regimes, and/or of limited applicability to the Javanese context are considered uncertain.

4.3 INDICATORS FOR MANGROVE ECOSYSTEM SERVICE PROVISION

This section describes key ecosystem properties underpinning the provision of seven mangrove ecosystem services and driving forces that determine these ecosystem services, as summarised in Table 4.2. The table also provides 'state' and 'performance' indicators for each service. We explain service provision below and provide additional references in Table 4.2.

4.3.1 Fish and shrimp provision

Mangroves provide various foods, such as fish, crustaceans, fruits and vegetables. We limit our study to fish and shrimp but summarise other food uses per mangrove species in Table S2.1 (Appendix 2). The available stock is a common state indicator for fish and shrimp provision; the actual harvest a performance indicator (Table 4.2). Harvest and stock are measured in relation to mangrove area or aquaculture pond size. Natural and artificial production involves crucially different underlying ecological properties and management. Natural provision depends on ecological and biophysical characteristics, regulating services and the nursery service of mangroves and adjacent ecosystems (Sheridan and Hays 2003). Ecosystem properties underlying natural production are provided in Table 4.2. They mostly include properties related to the nursery service, e.g. food abundance and

Table 4.2: Drivers, ecosystem properties underpinning service provision and state and performance indicators of mangrove ecosystem services. References in footnotes.

Ecosystem Service	Drivers of service provision (including management)	Ecosystem properties underpinning ecosystem service provision	State indicator (unit)	Performance indicator (unit)
Food (fish and shrimp)	Nursery service[2,3,4]; coastal fishing intensity[2,4]; aquaculture inputs[2,]	Nutrient availability[1,2,3]; water quality[2]; predation[3]; trophic subsidy[2,3]; physical subsidy[2,3]	Available stock (kg yr^{-1}; kg ha^{-1} yr^{-1})[5,6]	Actual harvest (kg yr^{-1}; kg ha^{-1} yr^{-1})[6,7,8]
Raw materials (tree biomass)	Climate and seasonality[10,14]; protection status of area[13,14]; harvesting methods[14]; desired end-use[10,14]; proximity user to forest[13,14]	Species richness[9,10]; tree density[9,10,11], diameter [10,11], height[9,10,11], age[9,10] and productivity[9,10,11]; fraction dead wood, litter[10]; soil substrate type[10,11]; forest size[9,10,11]; inundation and flooding pattern[10,11,12]	Available tree biomass for human use (ton yr^{-1}; ton ha^{-1} yr^{-1})[9,10]	Actual tree biomass harvested for human use (kg yr^{-1}; kg ha^{-1} yr^{-1})[9,15]
Carbon storage and sequestration	Long-term protection[10,12]; climate[12,16]; restoration[12,20]; temperature[12,16]; hydrological management[12,16,20]; distance from seaward edge[20]	Soil and sediment type[12,16,17]; soil depth[17,18,19,20]; organic matter content[12,19,20]; soil inundation[12,20]; tide[12,16,20]; tree diameter[10,11,16,18], age and size [6,17,20]; stem density[16,17,20]; riverine inputs[12,17]; species richness[17,18,20]; nutrient availability[12,17,18,20]	Carbon storage (ton ha^{-1}) [10,16,20]	Difference between carbon stocks of intact and impacted mangrove forests (ton ha^{-1} yr^{-1})[12,20]
Coastal protection (wave attenuation and storm surge protection)	Wave period and height[25,28]	Extent or width of forest[22,23,24]; species richness[22,24]; structural diversity[22,23,24]; tree age[22,24]; water depth[26,28]	Projected area of mangroves (m^2)[22,23]; width of mangrove greenbelt (m) [22,23]	Wave height reduction rate (m^{-1}) [27,28]; wave energy dissipation[27,28]
	Topography[24,26]			Surge reduction rate (m^{-1}) [29]
Water purification (N & P removal)	Biomass harvest[30,31]; mitigated sediment disturbance[30,31,33]; nutrient output aquaculture system[30,31,32]	N and P requirements trees[30,31,32]; litterfall[30]; biomass accumulation[30,31]; physically stable sediment[30,33]; mangrove area[30,31]; plant density, structure[31,32]; photo-synthesis rate[31,33]; water salinity[31,33]; flow speed[30,31]; clay mineralogy, iron content[30,33]; redox status[33]	Potential N and P removal (mg ha^{-1} yr^{-1})[30,31]	Actual N and P removal (kg ha^{-1} yr^{-1})[30,31,34]
Nursery service	Mitigation of pollution, overfishing and other pressures[3,5,35]	Nutrient trapping [5,25,35]; tidal mixing[3,35,36]; water inflow[3,36]; turbidity[3,36]; roots [3,5,25]; diverse trophic niches[3,35,36]; retaining immigrating larvae and juveniles[3,5,35,36]; intact hydrological cycles[3,35]	Relative contribution to fish, shrimp stock(%)[5]; mangrove-dependent juveniles maturing (%)[3]	Fish, shrimp harvest per area of mangrove (kg ha^{-1} yr^{-1})[7,35]; relative contribution to harvest (%)[5,35,36]
Nature-based recreation	Infrastructure[37,38,39]; recreation facilities[38,39,40]; travel distance[39,40]; crowdedness[39,42]; skyline disturbance[38]	Flora and fauna, land cover, land use, and/or cultural element with stated preference[37,38]; condition of ecosystem[37,38,39]	Potential number of visitors (# yr^{-1}; # ha^{-1} yr^{-1}) [37,38,42]	Actual number of visitors [38,40,42]; boat hires[40,42]; booked trips[38,40,42]

1 Mumby et al. (2004); 2 Rönnbäck (1999); 3 Sheridan and Hays (2003); 4 Naylor et al. (2000); 5 Manson et al. (2005); 6 Rönnbäck et al. (2003); 7 Kathiresan and Rajendran (2002); 8 Aburto-Oropeza et al. (2008); 9 Bosire et al. (2008); 10 Ong (1993); 11 Sukardjo and Yamada (1992); 12 Mcleod et al. (2011); 13 Ewel et al. (1998), 14 Walters (2005a); 15 Walters (2005b); 16 Alongi (2012); 17 Bouillon et al. (2008), 18 Donato et al. (2011); 19 Kauffman et al. (2011); 20 Kauffman et al. (2013); 21 Clough et al. (1997); 22 Massel et al. (1999); 23 Quartel et al. (2007); 24 Vo-Luong and Massel (2006); 25 Walters et al. (2008); 26 Zhang et al. (2012); 27 Mazda et al. (2006); 28 McIvor et al. (2012a); 29 McIvor et al. (2012b); 30 Gautier (2002); 31 Li et al. (2008); 32 Primavera et al. (2007); 33 Robertson and Phillips (1995); 34 Jackson et al. (2003); 35 Baran (1999); 36 Pauly and Ingles (1999); 37 Puustinen et al. (2009); 38 Van Oudenhoven et al. (2012); 39 Boon et al. (2002); 40 Satyanarayana et al. (2012); 41 Rönnbäck et al. (2007); 42 Knight et al. (1997).

predation (see section 4.3.6). Artificial production, i.e. fish and shrimp farming, depends mostly on additional artificial and natural inputs and involves inclosing the fish or shrimp stock in a secure system and providing it all nutritional and disease preventive requirements (Naylor et al. 2000). Aquaculture management inputs are described in Section 4.4.4. We note that aquaculture also depends on ecosystem services, such as nursery service (seedlings), erosion prevention and coastal protection (Rönnbäck 1999, Naylor et al. 2000).

4.3.2 Raw materials

Raw materials from mangroves can be harvested from leaves, bark, wood and dead wood (Walters 2005b). We limit our study to biomass in general and consider available biomass for human use a state indicator and the actual harvest indicates the performance (Table 4.2). Biomass harvest is considered sustainable if remaining below the forests' productivity (Ong 1993, Bosire et al. 2008). Raw material uses include fuel wood, fodder, tannin and construction material (Bosire et al. 2008, Walters et al. 2008). The strength and durability make mangrove wood suited for use in construction. However, diameter, growing form and stem length ultimately determine raw materials' use (Walters 2005b). Because these properties differ per species, mangrove species richness is a suitable proxy for raw material provision (Walters 2005b). Species richness depends on climate and ecological and biophysical conditions, such as inundation and soil substrate (Table 4.2). Table S2.1 in Appendix 2 summarises raw materials use per species, but we did not quantify actual use. Drivers of raw material provision include protection status, desired end use and the location of mangrove forests (Walters 2005a).

4.3.3 Carbon storage and sequestration

Mangroves are productive and biogeochemically active ecosystems, and as such represent important sinks of carbon in the biosphere (Ong 1993, Walters et al. 2008). Carbon storage and sequestration involve different time scales and processes. Carbon storage (ton ha^{-1}) can be considered a state indicator for carbon sequestration. Actual sequestration (ton ha yr^{-1}) is rarely measured, but can be estimated by calculating the difference between carbon storage of intact mangrove forests and impacted forests (Mcleod et al. 2011, Kauffman et al. 2013). However, meaningful carbon sequestration takes decennia, if not millennia (Mcleod et al. 2011).

Mangroves store carbon within living biomass both aboveground (leaves, stems, roots, branches) and belowground (fine roots), within non-living biomass (litter and deadwood) and as organic matter in their sediments (Mcleod et al. 2011, Alongi 2012). Carbon storage of living aboveground biomass is determined by age and size, species richness, tides etc. (Table 4.2). Mangrove forests show continued increase of photosynthesis levels for up to a century, before reaching a dynamic equilibrium (Ong 1993, Clough et al. 1997). Belowground carbon accounts for 49-98% of the total carbon stock in mangroves, and over 75% of belowground tree carbon can be found in dead, rather than live, roots (Donato et al. 2011, Kauffman et al. 2013). Soil and (dead) root carbon pools increase with increasing tree age (Donato et al. 2011, Alongi 2012). Belowground carbon has rarely been measured and a weak correlation exists between above- and belowground storage (Bouillon et al. 2008, Donato et al. 2011). While living biomass reaches a dynamic equilibrium, waterlogged mangrove soils continuously accumulate carbon (Mcleod et al. 2011, Alongi 2012). Differences in below- and aboveground carbon are further explained by ecosystem properties such soil depth, organic matter content, basal area etc. (Table 4.2). Carbon also accumulates in mangrove soils when silt, clay and organic particles are captured in mangrove ecosystems. This depends on forest floor properties that are influenced by climate, soil, sediment type and riverine inputs (Mcleod et al. 2011, Kauffman et al. 2013; Table 4.2).

Carbon sequestration depends on additional factors compared to carbon storage. Active management (restoration, hydrological management) and long-term protection of vegetation and soil is required to optimise and maintain long-term soil carbon accumulation (Alongi 2012). Vegetation clearance will expose mangrove soils and result in immediate release of carbon that has been sequestered over millennia (Ong 1993, Mcleod et al. 2011).

4.3.4 Coastal protection

Mangroves provide coastal protection by buffering waves and storm surges, and elevating soil surface in response to sea level rise (Mazda et al. 2006, McIvor et al. 2012a). We did not consider soil surface elevation (i.e. coastal erosion prevention and soil accretion) in this study, because it involves poorly understood and complex processes (Alongi 2008, 2012).

Wind and swell waves result of tides, wind and storms (Massel et al. 1999). Wave attenuation, i.e. wave height reduction, is caused by mangroves acting as an obstacle for the oscillatory water flow in waves. Wave energy is dissipated and wave height reduced because the water flow has to change direction and faces friction (Mazda et al. 2006). The wave height reduction rate (performance indicator) is indicated as the initial wave reduction over a horizontal distance travelled (m^{-1}). No significant results have been found for wave attenuation >70 cm, because of technical and practical difficulties to carry out measurements (McIvor et al. 2012a). Important ecosystem properties include species richness, structural diversity and mangrove tree age (Table 4.2). High diversity of branches, roots and trees is more likely to buffer wave impacts (Massel et al. 1999, Quartel et al. 2007). The mangrove' width and projected area are frequently used as a state indicator, i.e. to determine potential wave attenuation (McIvor et al. 2012a).

Storm surges are movements of sea water onto land caused by strong winds (McIvor et al. 2012b). Storm surge reduction rates (performance) are harder to establish than for wave attenuation and available information is limited to US-based studies (McIvor et al. 2012b). Factors influencing storm surge protection are similar as for wave attenuation (Table 4.2), but their predictive value is lower; contrary to wave attenuation, relationships between underpinning factors and storm surge reduction rates are not linear due to topographical influences (Zhang et al. 2012).

4.3.5 Water purification

Mangroves trap, transform and export nutrients, pollutants and sediments from natural and human sources (Robertson and Phillips 1995). We consider water purification by mangroves as the uptake of (inorganic) nitrogen (N) and phosphorus (P) from aquaculture discharge water. Conversely, N and P emission can be seen as a 'disservice' (Jackson et al. 2003). Actual nutrient removal (kg ha yr^{-1}) can be seen as a performance indicator, and potential removal a state indicator (Table 4.2). N and P emission in discharge water are mostly measured per ha of pond, whereas uptake is mostly measured per ha of mangrove. Mangroves lower excess nutrient concentrations by storing them in its roots, stems and leaves, and adsorption by stable sediments (Robertson and Phillips 1995, Li et al. 2008). Nutrient removal is measured in relation to mangroves' N and P requirements to support net primary productivity, provided that sufficient mangrove area is present and that accumulated biomass is harvested, retained or recycled within sediments (Table 4.2). Moreover, mangrove productivity depends on forest structure as well as species- and age-specific photosynthesis rates (Li et al. 2008). N and reactive P can furthermore be immobilised in sediments, which is mostly dependent on clay mineralogy, iron content, undisturbed sediments etc. (Robertson and Phillips 1995, Li et al. 2008). Finally, water salinity, water flow speed, and plant density and structure increase nutrient retention time (Table 4.2).

4.3.6 Nursery service

Mangroves provide nursery ground or living habitat to fish and crustaceans, thus supporting capture fishery and, to some extent, aquaculture (Rönnbäck 1999, Walters et al. 2008). Nursery can be provided through shelter, food and refuge, and spawning opportunities (Walters et al. 2008). The fraction of mangrove-dependent juvenile species that mature into adults is a common state indicator, but it has rarely been quantified (Sheridan and Hays 2003). Other indicators include the stock or harvest per area of mangrove (state and performance, respectively) and the relative contribution of mangroves to a given harvest (Table 4.2). Reviews, such as by Sheridan and Hays (2003) show that most studies fail to empirically relate the amount of juveniles that are recruited in mangrove areas with the extent to which they mature into adults that can be caught. However, some have provided useful underlying ecosystem properties can be distilled from the literature (Table 4.2).

Mangroves form integrated ecosystems with sea-grass beds, un-vegetated shallows and coral reefs, and fish and crustaceans can be either short-term or longer-term residents of mangrove ecosystems (Rönnbäck et al. 1999, Sheridan and Hays 2003). Differences in ecosystem dependence and residence time complicate measurements of mangroves' actual contribution (Sheridan and Hays 2003). Models have related mangrove area with fish and shrimp catches (e.g. Pauly and Ingles 1999), but local estimations require calibrations based on long-term measurements of harvests and mangrove cover. Important processes and properties required for nursery include high nutrient productivity (nutrient trapping, tidal mixing and freshwater inflow), turbidity and presence of roots (refuge), structural and biological diversity, hydrodynamic cycles retaining immigrating larvae and juveniles, and mitigation of pollution and other pressures (Table 4.2). The importance of these characteristics differs between and among crustaceans and fish species (Sheridan and Hays 2003).

4.3.7 Nature-based recreation

Nature-based recreation involves recreational activities related to nature or natural elements. For mangroves they include diving, bird watching and hiking. Tourism involves tourists spending a night on location, whereas recreation describes the activities (Puustinen et al. 2009). A state indicator for recreation could be the potential number of visitors for a specific recreation purpose and actual visitor numbers indicate the performance (Table 4.2). An area's suitability for recreation determines whether recreation is possible and suitability can therefore be used as a proxy for the state indicator (Van Oudenhoven et al. 2012). The suitability for recreation depends on a number of ecological and other factors (Table 4.2). Recreation occurs because certain flora and fauna, land use or culturally important features are appreciated by people. Such features include rare plants and animals, unspoilt views and traditional agriculture (Puustinen et al. 2009; Table 2). Recreation generally requires additional facilities and organisation as well as infrastructure such as roads, parking lots and walking bridges (Boon et al. 2002, Puustinen et al. 2009). Signs and information boards help to create awareness of potential users (Satyanarayana et al. 2012). Recreants might be discouraged by lacking facilities, crowdedness, travel distance, high noise levels, damaged or polluted ecosystems, mosquitoes etc. (Boon et al. 2002, Van Oudenhoven et al. 2012; Table 4.2). However, we note that recreants' preferences are personal and location-specific, and have rarely been quantified and standardised for mangroves and most other ecosystems.

4.4 MANAGEMENT REGIMES FOR MANGROVES IN JAVA, INDONESIA

We distinguish five broad categories of mangrove management regimes: *natural* (purpose: preserving biodiversity and ecological and biophysical functions), *low intensity use* (purpose: production of natural resources), *high intensity use* (purpose: rehabilitation and sustained food and raw materials production), converted to aquaculture (purpose: fish and shrimp cultivation) and *abandoned aquaculture* (no purpose). Eleven specific management regimes are divided over the five main categories (Table 4.3). We italicize the management regimes to emphasize that they are part of our typology. Table 4.4 provides an overview of all management activities, aquaculture indicators and ecological characteristics for each management regime. Differences are further explained below.

4.4.1 Natural mangroves

Natural mangroves have recognised ecological and/or biophysical functions that should be formally preserved (Forestry act No. 41/1999). The Forestry act divides forests into *protection*, *conservation* and *production* forests based on their associated ecological, biophysical and/or economic function. Mangrove forests under *protection* and *conservation* fall within *natural mangroves*, whereas *production* falls under *low intensity use mangroves* (see 4.4.2).

Protection forests serve to protect biodiversity and ecological and physical functions. Ecological functions include nursery and source of genetic resources, physical functions include coastal protection and preventing saltwater intrusion. *Protection* is more strictly enforced compared to the *conservation* regime as the local governments are responsible for it (Sualia et al. 2013). Policies related to protection also apply to greenbelts and bordering areas. Local inhabitants are permitted to hunt on unprotected animals and gather NTFP at low intensity, i.e. collection of deadwood and other materials without inflicting damage to the vegetation (Table 4.4). Fishing occurs only around mangroves. Furthermore, permits are issued for activities related to science, education and research and development. Maintaining the natural integrity of protection forests might require the restoration of water flows and removing invasive plant species (Lewis III 2005). Recreational visits are also allowed, but no infrastructure is in place to support recreation. The management state of mangroves under protection in Java is characterised by four or more mangrove species. Trees have a maximum age of 20-30 years and corresponding heights and perimeter of higher than 30 m and 5-70 cm, respectively (Table 4.4).

Table 4.3: Typology of management regimes for mangroves in Java, Indonesia

Management regime*	**Description**
NATURAL MANGROVES	
Protection of ecological and physical functions	Management aims to preserve ecological and biophysical functions and biodiversity. Local governments are responsible for protection. Management activities include hunting on unprotected animals, low intensity NTFP harvesting, fishing and facilitating research.
Conservation of biodiversity and local culture	Management aims to conserve biodiversity and ecological functions, natural resources, and local culture. Management activities include facilitating recreation and tourism, hunting on unprotected animals, low intensity NTFP harvesting and fishing.
LOW INTENSITY USE MANGROVES	
Production of forest products	Management aims at utilizing mangroves' economic function, which is mainly NTFP and timber production. Management activities include timber harvesting, high intensity NTFP harvesting, replanting mangroves, enabling recreation, and fishing.
Unprotected	There is no formal protection in place, due to remoteness or abandonment. Management activities can include timber harvesting, low intensity NTFP harvesting and fishing.
HIGH INTENSITY USE MANGROVES	
Plantation	Management aims at mangrove rehabilitation, to slow down deforestation rate and restore ecological and economic functions, thereby increasing people's prosperity. Management activities include high intensity NTFP harvesting, recreation, fishing and planting mangroves.
Silvo-fishery	Management combines aquaculture and mangrove replanting and aims to rehabilitate mangroves to reduce deforestation rates, restore ecological and economic functions, thereby increasing people's prosperity. Management activities include high intensity NTFP harvesting, recreation, cultivating shrimp, crab and fish, maintaining dykes and replanting mangroves.
MANGROVES CONVERTED FOR AQUACULTURE	
Eco-certified aquaculture	Aquaculture that follows guidelines related to animal health and welfare, food safety and quality, environmental integrity and social responsibility. Mangrove rehabilitation and protection of greenbelt is required. Guidelines for eco-certification are currently being developed. Management activities include use of artificial stock, high seed density and some fertilizer.
Extensive aquaculture	'Traditional' aquaculture in large ponds, with use of mixed stock, low seed density, limited fertilizer and pesticide, and natural feed. Water exchange occurs through natural tides.
Semi-intensive aquaculture	Aquaculture with use of artificial stock, low to medium seed density, fertilizer, pesticides and mixed feed. Water exchange occurs through water pumps and pedal wheels.
Intensive aquaculture	Aquaculture in small ponds with use of artificial stock, high seed density, fertilizer, antibiotics, pesticide, and formulated feed. Water exchange occurs through water pumps and pedal wheels.
ABANDONED AQUACULTURE	
Abandoned aquaculture	Management activities have been abandoned, due to depletion. No regulations apply.

* Main categories are based on Stevenson (1997), Gilbert and Janssen (1998), Macintosh et al. (2002), Alkemade et al. (2009). Specific management regimes are based on Gilbert and Janssen (1998), Sofiawan (2000), Rönnbäck (2001), Bengen (2003), Primavera et al. (2007), Kusmana et al. (2008), Walters (2005b), Barbier et al. (2011) and Indonesian policy documents: Forestry Act No. 41/1999, Government Regulation No. 10/2010 and No. 28/2011, Ministry of Forestry Regulation No. 3/2004 and 'Guidelines for development of mangrove management models' by the Ministry of Forestry (2012)

Table 4.4: Management activities, aquaculture indicators and ecological characteristics of management regimes for mangrove areas in Java, Indonesia

Management regime	Management activities					Aquaculture management indicators					Ecological characteristics of mangrove trees							
	Recreational visits (Y/N)	Fishing (Y/N)	Timber harvest-ing (Y/N)	NTFP harvest intensity	Mangrove replanting (Y/N)	Avg. pond size (ha)	Origin stock	Stock density (m^{-2})	Origin additional feed	Fertilizer or pesticide use	Avg. species #	Avg. d.b.h (cm)	Max. height (m)	Max. age (yr)	Max. perimeter (cm)	Max. root length (m)	Undergrowth	Seedling, sapling #
Protection	Y	Y	N	Low	N	-	-	-	-	-	≥4	17-22	≥30	20-30	50-70	>1.5	Clear	Low
Conservation	Y	Y	N	Low	N	-	-	-	-	-	3-4	12-16	≥30	12-19	30-50	>1.5	Shrubs	Medium
Production	Y	Y	Y	High	Y	-	-	-	-	-	3-4	<13	≤30	10-16	<40	<1.5	Shrubs	Medium
Unprotected	N	Y	Y	Low	N	-	-	-	-	-	3-4	<13	≤30	10-16	<40	<1.5	Shrubs	Medium
Plantation	Y	Y	N	High	Y	-	-	-	-	-	≤3	<11	<20	7-10	<35	<1	Shrubs	High
Silvo-fishery	Y	N	N	High	Y	>1.5	Nat.	1-3	Nat.	P	≤3	<11	<20	7-10	<35	<1	Shrubs	High
Eco-certified aquaculture	Y*	-	N	-	Y	0.1-1	Nat., A	10-50	Nat.	F / P	≤2	<7	10-20	<10	<20	-	No	High
Extensive aquaculture	N	-	N	Low	N	1-10	Nat., A	1-3	Nat.	F	≤2	<3	10-20	4-6	<10	-	No	High
Semi-intensive aquaculture	N	-	N	-	N	1-2	Nat., A	3-10	Nat., A	F / P	≤2	<3	10-15	<4	<10	-	No	Medium
Intensive aquaculture	N	-	N	-	N	0.1-1	A	10-50	A	F / P	1	<2	10-15	2-4	<5	-	No	Low
Abandoned aquaculture	N	N	N	-	N	-	-	-	-	-	≤2	<1	<1	1-2	3	-	Stumps	Low

Note: Y/N = Yes / No; - = not applicable; Nat. = Natural; A = Artificial; F = Fertilizer; P = Pesticide; Stock density = shrimp; d.b.h. = diameter at breast height.
Sources for indicators of management: Bengen (2003), Gilbert and Janssen (1998), Kusmana et al. (2008), Macintosh et al. (2002), Primavera et al. (2007), Rönnbäck (2001), Sofiawan (2000), Stevenson (1997), Sualia et al. (2013), Walters (2005b), and Indonesian policy documents: Forestry Act No. 41/1999, Government Regulation No. 10/2010 and No. 28/2011, Ministry of Forestry Regulation No. 3/2004 and 'Guidelines for development of mangrove management models' by the Ministry of Forestry (2012). Sources ecological characteristics: Bengen (2003), Matthijs et al. (1999), Middelburg et al. (1996), Kusmana et al. (2008) and Schrijvers et al. (1995).

Mangroves under *conservation* have unique recognised ecological, economic and biological characteristics and fall under the jurisdiction of the Ministry of Forestry. Their main purpose is the preservation of biodiversity, natural resources and local culture. In accordance with government regulation No. 28/2011, conservation forests include forest reserves, hunting parks, national parks and recreation parks. Recreation facilities (e.g. walking tracks, information centres) are constructed and maintained to promote nature-based recreation and tourism. Local communities are permitted to hunt on unprotected animals, gather NTFP at low intensity and fish around the forest. Mangrove trees are younger when compared to protection forests and all ecological characteristics consequently score lower (Table 4.4). Ecological characteristics of conservation forests range widely, because Java also has many 'young' conservation forests, which are characterised by a similar policy context and management activities but lower maximum age and corresponding characteristics (Table 4.4).

4.4.2 Low intensity use mangroves

Low intensity use mangroves are natural or replanted forests that are used for NTFP harvesting and timber extraction (Forestry act No. 41/1999). They can either be actively managed by communities (through private ownership), by local or regional governments, or, due to lacking protection, freely used for NTFP harvesting and timber extraction. Management activities may not significantly change the ecosystem or involve construction of permanent infrastructure. We distinguish between mangrove production forests and unprotected mangroves.

Production forests have a formally recognised economic function in timber and NTFP production. High intensity NTFP harvesting occurs, which involves intensive management to produce and process NTFP. Replanting of trees is compulsory if forests' ecological integrity has been affected due to management activities. Local people are permitted to hunt and fish. Tourists also visit production forests, although no infrastructure exists to accommodate them. The most mature mangrove trees in Javanese *production* forests can reach sixteen years, with corresponding height and perimeter of <30m and <40cm, respectively. In addition, fewer mangrove species (3-4) and more undergrowth and seedlings can be expected, compared to natural mangrove areas (Table 4.4; Bengen 2003).

Unprotected mangroves do not fall under any formal jurisdiction or land-use purpose. It is a highly diverse management regime that includes formerly abandoned, restored or left-alone mangrove areas. *Unprotected* mangroves also include mangroves that are gradually restoring because of un-intentional protection, for instance due to civil unrest or being difficult to reach or exploit. Low intensity harvesting and limited timber cutting takes place, due to the combination of weakly enforced regulation and limited accessibility. We assume *unprotected* mangroves to have similar ecological characteristics to production forests, but expect higher variability (Table 4.4)

4.4.3 High intensity use mangroves

High intensity use mangroves are formally regarded as rehabilitation sites (Presidential decree No. 73/2012). They are characterised by a combination of forested, converted and/or restored mangroves and their main purpose is the sustainable fish or timber provision combined with mangrove restoration or conservation. We distinguish between *plantation* and *silvo-fishery*.

Mangrove plantation generally involves 'silviculture', i.e. the controlled sustainable establishment and growth of mangrove forests to meet landowners' needs (Walters et al. 2008). Ministry of Forestry regulations apply to *plantations* in Indonesia. Mangroves can be planted or regrown due to controlled regeneration (Bosire et al. 2008), with the purpose to provide raw materials, support fisheries, aquaculture and tourism, or to enhance coastal protection (Walters et al. 2008). Fishing and high intensity NTFP harvesting take place in mangrove *plantations*. Tourists visit *plantations* for fishing and additional out-door activities, such as birding, boating and hiking. No timber harvesting occurs, and mangroves are replanted when needed. *Plantations* in Java are characterised by young trees (maximum age 7-10 years) with maximum heights and perimeters of <20 m and<35 m, respectively (Table 4.4).

The goal of *silvo-fishery*, according to Ministry of Forestry Regulation No. 3/2004, is to rehabilitate the mangroves' ecological and economic functions, i.e. to provide services such as coastal protection and nursery without causing economic losses to shrimp aquaculture. Regulations from the ministries of Forestry, Regional Spatial Planning and Fishery apply. Benefits of *silvo-fishery* include a) stronger embankments, b) fodder for livestock, c) nursery for shrimp and crabs, d) coastal erosion prevention, e) salt water intrusion prevention and f) coastal protection (Sofiawan 2000, Bengen 2003, Sualia et al. 2013). Our *silvo-fishery* management regime provides all formally targeted ecological and economic functions in a natural way (Bengen 2003). This 'ideal' option is illustrated in Figure 4.2 (Bengen 2003) but we note that it is virtually absent in Java. In fact, most Javanese 'silvo-fisheries' more resemble *extensive* or *semi-intensive aquaculture* systems. Important reasons for the lack of 'ideal' *silvo-fisheries* in Java include limited knowledge on optimal *silvo-fishery* management and the relatively small size of ponds in or around which mangroves are replanted. Four variations of *silvo-fishery* 'models' are officially recognised by Ministry of Forestry Decree No. 3/2004. They can be divided into systems with mangroves planted inside ('Type 1') or outside the ponds ('Type 2'). Most versions provide few of the desired ecosystem services, due to differences in where mangroves are planted, the presence of water in- and outlets, etc.

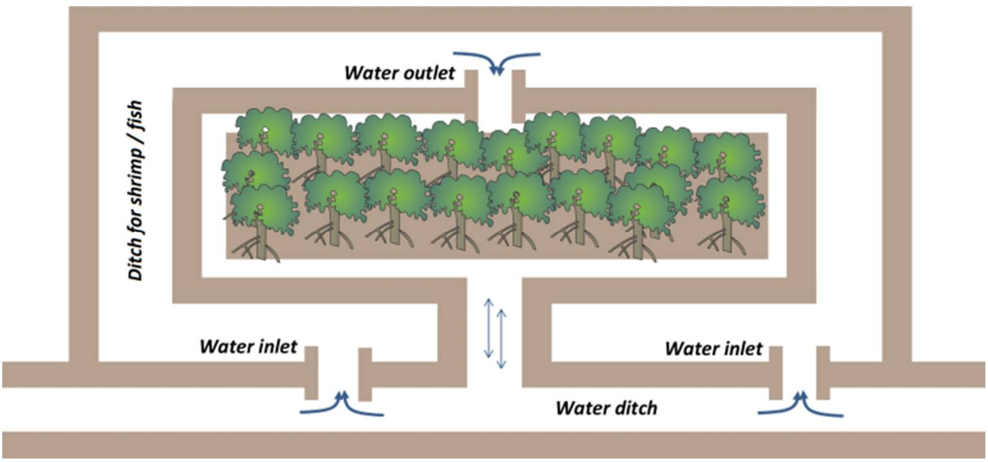

Figure 4.2: Illustration of the 'ideal' silvo-fishery option, with a two-gate water inlet system, a separate mangrove area inside the pond, and a separate ditch for fish. Source: Bengen (2003).

'Ideal' *silvo-fishery* systems are enclosed by dykes, and a large ditch surrounds a centrally located patch of mangrove trees. Natural tidal movement provides water circulation, which is stimulated by two water inlets (Figure 4.2). A water outlet directs the effluent through the mangroves, thus removing excess nutrients from the discharge water. Only natural shrimp stock is added and no additional feed or fertilizer is used. Limited pesticide use has been noted in Java (Bengen 2003, Sualia et al. 2013). *Silvo-fishery* ponds are generally 1.5 ha or larger but pond size can vary as usually aquaculture ponds are rehabilitated that have been constructed before. Recreational visits are quite common in silvo-fishery sites and mainly focus on recreational fishing, boardwalks, and environmental education. Furthermore, NTFP are harvested at high intensity. The amount of mangrove cover per pond is quite variable, but an assumed ideal pond-mangrove ratio is 60:40 (Bengen 2003, Bosma et al. 2014). Ecological characteristics of *silvo-fisheries* are similar to that of mangrove plantations (Table 4.4).

4.4.4 Mangroves converted for aquaculture

Aquaculture ponds are owned or rented by the private sector instead of communities or governments. Landowners follow regulations stipulated by ministries of Environment, Public Works, Agriculture and Fishery, which often leads to 'creative' interpretation or combination of these regulations (Sualia et al. 2010). In addition, the Regional Spatial Planning bureau and other local district offices might overrule certain regulations by ordering the conversion of mangroves or expansion of aquaculture. We limit the *converted* category to *aquaculture* only because other land-use options can generally be found more inland. We distinguish between *eco-certified, extensive*, *semi-intensive* and *intensive aquaculture*, based on stocking density and the levels of feed input and fertilizer use. Characteristics of aquaculture below are mostly based on Rönnbäck (2001), unless stated otherwise, and apply to shrimp and fish aquaculture. Where possible, we distinguish between shrimp and fish aquaculture (Table 4.4).

Requirements for *eco-certified aquaculture* are currently being developed by the Aquaculture Stewardship Council (ASC) in Indonesia, which follows up on certification systems such as the AAC Standard US (Ministry of Fishery, Indonesia) and Global Gap, a European standard focusing on food safety. We note that the following requirements are the result of personal communications and reviewing scattered information in Indonesian grey literature. *Eco-certified aquaculture* is currently absent in Java and, similarly to *silvo-fishery*, represents a desired situation. Eco-certification requirements will focus strongly on biodiversity and mangrove rehabilitation. Apart from engaging in sustainable and 'neat' management, landowners take part in ex-situ mangrove restoration to improve biodiversity and greenbelt maintenance. In-situ replanting (i.e. in and around ponds) could also be made compulsory. Raw materials harvesting is allowed nowhere. *Eco-certified aquaculture* is similar to *intensive aquaculture* in terms of pond size (0.1-1 ha), stock density (10-50 m^{-2}) and feed. However, shrimp seeds must be of native species raised in natural hatcheries. No artificial feed is allowed and only natural pesticides are used for pest control (Table 4.4).

Extensive aquaculture systems are operated where land is inexpensive. They are usually rented by local individuals (Sualia et al. 2013). Ponds vary greatly in size (1-10 ha). Limited infrastructure is required and construction occurs in a less destructive manner compared to other aquaculture options. Pond owners rely on the tides to provide most of the food for the shrimp, but pesticides as well as fertilisers or manure are added. 'Pre-preparation ponds' are often used so that plankton can flourish before stocking the ponds with fish or shrimps. Stocking occurs naturally or artificially with a low seed density (Table 4.4). This can result in poly-culture ponds, and it depends on the farmer if additional fish are being kept or removed from the system. Remaining mangrove trees are frequently pruned and used for limited raw material use (Table 4.4).

Semi-intensive aquaculture aims to increase the production of fish from pond systems beyond natural feed supply and stocking densities. Production occurs with artificial stocking, natural and artificial fertilisers and supplementary feeds, mostly natural and some formulated (Table 4.4). This entails considerable construction and management and high stocking density, compared to extensive farming. Water is exchanged artificially, and aeration of shrimp ponds occurs through pumping and using pedal wheels. The pond size is four to five times lower than in *extensive aquaculture*. Remaining mangrove trees are frequently pruned and used for limited raw material use (Table 4.4).

Intensive aquaculture systems are subject to developed infrastructure, hatchery and feed industries. Farmers use large quantities of commercial food (supplements) and chemical fertilizers, pesticides and antibiotics. The stocking density is very high, and no other species are cultured together in the pond (Table 4.4). Pond sizes range from 0.1 to one ha. *Intensive aquaculture* uses pumped seawater, and is often located beyond the intertidal and the natural mangrove setting. Pedal wheels and pumps are used to control water flows. It has to be stated that *intensive aquaculture* is quite rare in Java and the rest of Indonesia as compared to semi-intensive aquaculture (Sualia et al. 2010).

4.4.5 Abandoned aquaculture

Abandoned aquaculture sites have been impacted by and abandoned after unsustainable aquaculture exploitation, without any plan to restore either the mangroves or the aquaculture (Stevenson 1997). We consider *abandoned aquaculture* a separate management regime, because no formal management is in place, ownership is generally absent or unknown and the ecological and biophysical condition is much worse of compared to *converted to aquaculture* management regimes. General reasons for disuse include flood damage, shrimp disease and poor water quality (Stevenson 1997). Shrimp aquaculture farms are most often abandoned compared to other aquaculture. Abandoned aquaculture sites are difficult to generalise, due to difference in duration of abandonment and intensity of former land uses. Remnants of dykes and pumps have remained, and soils will be impacted by acid sulphate due to traces from excess nutrient and pesticide use. The surface area of abandoned sites is sometimes used for alternative purposes, such as housing, agriculture, and storage. Forest regrowth has not occurred and grasses and shrubs mostly occupy the area (Table 4.4). Note that, if managed and protected correctly, regeneration of mangrove species could be possible, depending on the pollution levels, inundation periods and inflow of mangrove seedlings (Stevenson 1997).

4.5 EFFECTS OF MANAGEMENT REGIMES ON MANGROVE ECOSYSTEM SERVICES

Information on ecosystem service provision by all management regimes will contribute to better, more informed decision making. Section 4.5.1 describes mangrove ecosystem service provision per management regime on and Section 4.5.2 uses these results to describe the consequences of transitions from one management regimes to another.

4.5.1 Mangrove ecosystem service provision per management regime

Table 4.5 integrates all scores of ecosystem service provision and provides an overview of ecosystem service provision per management regime. Detailed information on how these scores were determined is given in Appendix 3, as well as additional references.

The results suggest that mangroves under the *protection* regime score highest for all ecosystem services except food and *conservation* mangroves only score lower for raw materials. Conversely, mangroves *converted to aquaculture* received the maximum score for food production, but this coincides with low or even negative provision of all other ecosystem services (Table 4.5). Such disservices are carbon emission, risk of storm surges and water pollution. *Production* mangroves are managed for raw materials provision (i.e. wood and NTFP). This service scores similar or lower than in *natural* mangroves because we considered available biomass as indicator for the potential service provision. The actual raw materials harvest might be higher in *low intensity use* mangroves, because timber harvest is not allowed in *natural* mangroves. All other services in *low intensity use mangroves* score somewhat lower than in *natural mangroves* (Table 4.5).

Combining mangrove rehabilitation and production of raw materials (*plantation*) and shrimp (*silvo-fishery*) are land-use purposes of *high intensity use* mangroves. Remarkably, service provision in *plantations* scores similar or even higher (water purification) than in *production mangroves*, despite the limited species richness and lower age of mangroves in plantations (Table 4.5). Shrimp production in *silvo-fisheries* also coincides with low to medium provision of other ecosystem services.

Due to the absence of natural features*, abandoned aquaculture* systems provide disservices in the form of carbon emission, risk of wave and storm surge impact, and nutrient emissions (Table 4.5). Disservices result from exposed and oxidizing sediments as well as remaining aquaculture structures (Robertson and Phillips 1995, Kauffman et al. 2013, Winterwerp et al. 2013). Other ecosystem services are not provided.

The indicators mangrove species richness, mangrove age, root length and structural diversity account for decreasing service provision, going from *protection* to *plantation mangroves* (Table 4.5). All scores, expect for nature-based recreation, could be established in relation to these ecological characteristics (see Appendix 3), which makes them excellent indicators for assessing management effects on ecosystem services. Finally, differences between *silvo-fisheries* and *aquaculture* systems are mostly due to higher and gradually increasing aquaculture inputs. Absence of mangroves combined with dyke construction, inputs, digging up sediment and draining ponds contribute to lower ecosystem service provision in *aquaculture* systems. *Silvo-fishery* systems provide somewhat lower amounts of shrimp compared to *aquaculture* systems, but *aquaculture* management results in varying degrees of carbon emission (Kauffman et al. 2013), risk of storm surges (Winterwerp et al. 2013) and water pollution (Robertson and Phillips 1995).

Table 4.5: Scores for ecosystem service provision in mangrove management regimes in Java, Indonesia. Circles (●/○) indicate positive and diamonds (◆/◇) indicate negative ecosystem service provision, whereas a dash (-) indicates that no ecosystem service is provided. Closed shapes (●/◆) indicate high certainty and open shapes (○/◇) low certainty. Section 4.5.1 and Appendix 3 explain the underlying information for this table.

	Scores for ecosystem service provision						
Main management category Management regime	**Food**	**Raw materials**	**Carbon storage & sequestration**	**Coastal protection**	**Water purification**	**Nursery service**	**Nature-based recreation**
NATURAL MANGROVES							
Protection	○○	●●●	●●●	○○○	●●●	○○○	○○○
Conservation	○○	●●	●●●	○○○	●●●	○○○	●●●
LOW INTENSITY USE MANGROVES							
Production	○	●●	○○	○○	●●	○○	○○
Unprotected	○	○○	○○	○○	○○○	○○	○
HIGH INTENSITY USE MANGROVES							
Plantation	●	●●	●●	○○	●●●	○○	○○
Silvo-fishery	●●	○	○	○	○○	○○	○○
MANGROVES CONVERTED FOR AQUACULTURE							
Eco-certified aquaculture	○○○	○	-	◇	◇◇◇	-	○
Extensive aquaculture	●●	○	◇◇	◇◇	◇◇	-	-
Semi-intensive aquaculture	●●	-	◇◇	◇◇	◇◇	-	-
Intensive aquaculture	●●●	-	◇◇	◇◇	◆◆◆	-	-
ABANDONED AQUACULTURE							
Abandoned aquaculture	-	-	◇	◇◇	◇◇	-	-

However, just 18 (23%) of all 77 results in Table 4.5 are judged as ‘certain’, whereas the remaining 59 (77%) were considered ‘uncertain’. ‘Certain’ results are supported by multiple sources, based on multiple indicators and applicable to the Javanese context. Interpolated results, based on few indicators, weakly linked to management regimes, or limited applicable to the Javanese context are considered ‘uncertain’. The high number of uncertain results is caused by lacking empirical evidence in the literature on service provision per management regime. Results on potential rather than actual service provision were also more prevalent in the literature. Most certain results could be established for ecosystem services (e.g. food, raw materials and carbon sequestration) in *plantations* and *natural mangroves* in general, although in case of the latter it was often difficult to distinguish between *conservation* and *protection mangroves*. Despite many uncertain scores, the literature clearly indicates that substantial differences occur between management regimes in terms of ecosystem service provision.

4.5.2 Possible transitions between management regimes and their effects on ecosystem services

Management decisions generally involve choices between management regimes as well as transitions from one regime to another (Peña-Cortés et al. 2013). We used the added up scores from Table 4.5 to illustrate possible effects of transitions between our management regimes (Figure 4.3).

Mangrove conversion to *aquaculture* has been prevalent over the last decades in Java (Sukardjo 2009). Figure 4.3 suggests that converting mangroves for aquaculture could lower total ecosystem service provision scores substantially. This conversion can occur immediately, whereas aquaculture abandonment, natural regeneration and mangrove rehabilitation (Figure 4.3) are gradual processes. Aquaculture abandonment is poorly studied, but our figure indicates that abandoned systems could eventually recover to regimes with high ecosystem service scores. Mangrove rehabilitation (i.e. from converted mangroves to silvo-fisheries or plantations) could increase ecosystem service scores within up to ten years. The findings in Figure 4.3 should be treated with caution, because trade-offs between multiple management regimes and transitions between regimes over time have not been quantified in the literature and some transitions have not yet been observed in Java. Transitions between management regimes require additional investments, management activities and time, which were not considered in our analysis. However, Figure 4.3 offers an illustrative way of comparing management regimes as 'steady states' and explaining potential consequences of management decisions. Our policy review showed that terms such as 'intensification', 'conversion', 'degradation', 'sustainable management' and 'rehabilitation' feature frequently but are often ill-defined. Our findings show important useful differences between management regimes and are useful to stimulate discussion on the consequences of management decisions.

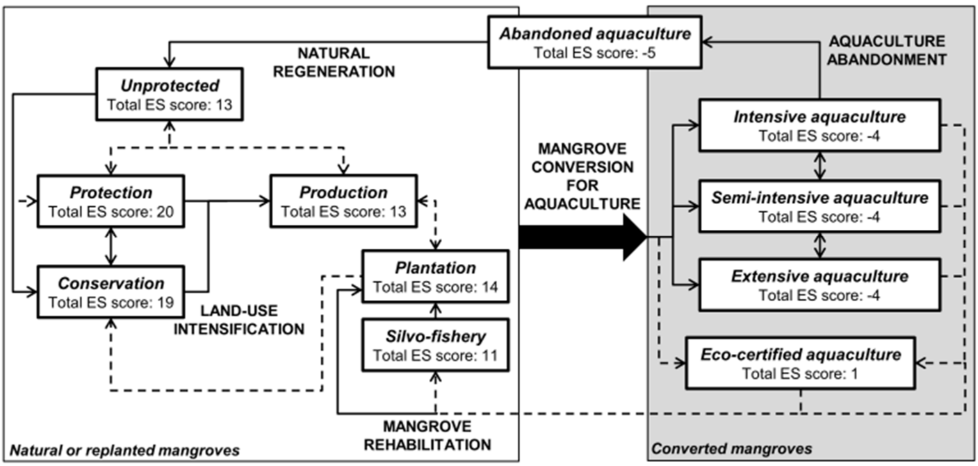

Figure 4.3: Possible transitions between management regimes in Java's mangroves. Ecosystem service (ES) scores of Table 5 were added up for each management regime. They grey area on the right indicates mangroves converted for aquaculture and the white area on the left indicates natural or replanted mangroves. Dotted lines between regimes indicate transitions that could not be observed in Java yet. Terms in capital letters indicate transitions.

4.6 DISCUSSION

This study aimed to assess the consequences of management decisions in Java's mangroves, Indonesia, by studying the effect of different management regimes on ecosystem services. We related characteristics from novel management regimes to ecosystem properties underpinning seven ecosystem services and indicators for potential and actual service provision. Our results suggest that management regimes aiming to preserve biodiversity, coastal protection and other regulating services (*conservation* and *protection*) can also result in providing large amounts of food and raw materials. Moreover, rehabilitating mangroves in former *aquaculture* systems (*plantation* and *silvo-fishery* regimes) can result in the provision of all seven ecosystem services and, thus, reverse negative impacts from *aquaculture* systems. Our results not only indicate large ecosystem service losses between *natural* mangroves and mangroves *converted for aquaculture*, but also suggest that *aquaculture* systems continue to provide disservices over time in the form of carbon and nutrient emissions, and increased flood risk.

Our typology of management regimes corresponds with the Javanese context and reflects relevant local management and land-use planning aims, because the eleven regimes are clearly defined by a land-use purpose and firmly rooted in Indonesian legislation. Decision makers can assess the consequences of these aims by considering the ecosystem services provided per management regime. Currently, Javanese policy makers are discussing our management regimes and related ecosystem service scores in the context of silvo-fishery and sustainable aquaculture regulations, which signifies an opportunity to influence decision making (c.f. Sualia et al. 2013). The management regime typology could be applied to other regions and countries if adjusted for local policies, management and ecological conditions. In other Indonesian regions our management regimes will have to be based on different ecological characteristics, although most policy and management aspects will likely be similar.

Our paper integrates information from various scientific disciplines into policy-relevant findings. However, several important assumptions and simplifications had to be made for the analysis. The following sections discuss the management regime typology, the ecosystem services assessment and, finally, the management implications of the study's findings.

4.6.1 A management regime typology for analysing management effects on ecosystem services

Assessing ecosystem service provision for different management regimes has been proposed as a major research challenge by many (e.g. ICSU-UNESCO-UNU 2008, Carpenter et al. 2009, De Groot et al. 2010b). However, currently available broad 'management regimes' lack consistent terminology, and specific and quantifiable characteristics to describe management regimes unambiguously (Braat et al. 2008, De Groot et al. 2010b, Van Asselen and Verburg 2012). Thus, we omitted inconsistent categories (e.g. 'degraded' and 'unsustainable') and introduced specific management regimes that enable comparing ecosystem services per regime. We also included 'converted' systems, which have mostly been described as either more intensive land-use systems (Verburg et al. 2013a) or 'degraded' systems (Braat et al. 2008). Some argue that converted systems produce economic goods rather than ecosystem services (c.f. Schröter et al. 2014b). We included converted mangroves in our typology and assessment because they are the main cause for mangrove loss and their outcomes should be compared to the benefits derived from differently managed systems (Rönnbäck et al. 2003). Alkemade et al. (2009) and Verburg et al. (2013a) both classified land-use intensity by simple management indicators (e.g. irrigation and grazing intensity). However, these were global studies and used generic indicators, whereas our typology is the first to develop a full range of specific ten management characteristics and indicators, and eight ecological characteristics. Our typology used the local variation in legislation and management activities. Moreover, the easily measurable ecological characteristics served to both verify local management regimes and to quantify the provision of ecosystem services. Integrating literature on land-use science, mangrove forest and aquaculture management, and mangrove ecology was required for our typology. Finally, we conducted a field assessment to verify the management regimes. Our typology is firmly rooted in scientific literature and Javanese legislation, and enables a consistent indicator-based comparison of ecosystem service provision for multiple management regimes.

Although most management regimes are common in Java, we note that *unprotected mangroves*, *silvo-fisheries* and *eco-certified aquaculture* are unusual regimes in Java. *Unprotected mangroves* are not listed in official policy documents but could be observed throughout Java, especially where aquaculture had been abandoned. Our literature review yielded nine different *silvo-fishery* variations that are found throughout Indonesia. Interestingly, the formally recognised variations (four out of the nine) are unable to provide all desired ecosystem services (Sofiawan 2000, Bengen 2003). We therefore only considered the 'ideal' variation as our *silvo-fishery* management regime (Sofiawan 2000). This regime is virtually absent in Java, but was used as a reference regime to be considered for decision making. More quantitative research on ecosystem services by silvo-fisheries is urgently needed to assess their potential as mangrove rehabilitation sites (Rönnbäck 2001, Bengen 2003). *Eco-certified aquaculture* is also non-existent in Java, but guidelines are currently being developed. This regime could provide in-situ and ex-situ benefits as replanted mangrove trees in aquaculture ponds could improve nursery, water purification etc. and mangrove trees planted outside the ponds could strengthen greenbelts and consequently provide coastal protection (McIvor et al. 2012a). Although the regimes are unusual, we consider them relevant to Java's context and future land-use planning. By assessing their potential ecosystem services we highlight their importance.

The management regime typology is based on legal, socio-economic and ecological characteristics, and, thus, captures multiple aspects that determine decisions on land use and management (Ghazoul 2007, Peña-Cortés et al. 2013). The typology does not consider other important drivers of land use, management and ecosystem services, such as spatial extent, climate change, (inter)national markets illegal management activities and political instability. Most ecosystem services (e.g. coastal protection, nursery) require a minimum amount of mangrove cover, but this could not be specified in our typology (McIvor et al. 2012a). Currently, policy regulations specify a minimum width for mangrove greenbelts (Sualia et al. 2013) but spatially explicit management regimes were difficult to develop because of lacking spatial guidelines for management regimes. We assumed that management regimes in *natural* and *low intensity* mangroves would be sufficiently large to provide multiple ecosystem services. Comparing management regimes of different sizes could help to support these claims and would contribute to more informed spatial planning. Indonesia has suffered from extreme climate events such as tsunamis and tropical storms (Cochard et al. 2008). Java has thus far been spared compared to other islands, but coastal mangroves are continuously affected by climatic factors and this could affect management activities (Ghazoul 2007). We partly account for this by relating management regimes to coastal protection services and also explain the influence of climatic factors on carbon storage and other ecosystem services (Mcleod et al. 2011). The choices for management regimes are probably influenced by national and foreign shrimp and timber markets, because Indonesia is a major international player with large potential for both markets (Sukardjo 2009, Bosma et al. 2014). We partly accounted for this by considering legislation that covers licensing to private and governmental institutes. We acknowledge the strong influence of the international market on the demand for shrimp and timber products, which could result in expansion of *aquaculture* or *production* regimes, but consider this relevant for management decisions rather than our typology. We did not account for illegal fishing and timber harvesting in our regimes, but these practices are assumed to pressure especially *natural* and *low-intensity use mangroves* (Ewel et al. 1998, Walters 2004). The political situation in Java is highly dynamic as legislations change swiftly and local legislation can be overturned by district or national legislation and vice versa (Sualia et al. 2013). However, we consider this more relevant for applying and monitoring management decisions than for the development of our typology. All *natural*, *low intensity use* and *high intensity use* management regimes are based either on existing forestry regulations or on drafted mangrove forest management regulations. Changes in this legislation are highly unlikely to result in a different typology. Most of the above discussed drivers affect our management regimes only indirectly or relate to management decisions rather than our regimes. Our typology includes direct drivers and covers realistic regimes that are considered realistic and long-lasting land-use purposes. Moreover, the typology is primarily a tool to analyse management effects on ecosystem services and not a precise account of Java's coastal systems, although policy makers are now considering our management regimes in their planning discussions.

4.6.2 Indicator-based analysis of ecosystem services per management regimes

We compiled a comprehensive set of indicators by integrating qualitative and quantitative information on drivers, ecosystem properties, and 'state' and 'performance' indicators (Section 4.3). Although some studies have collected indicators for multiple mangrove ecosystem services (e.g. Barbier et al. 2011), few have also applied the indicators in a quantitative or qualitative ecosystem service assessment. We evaluated the scientific consensus on important ecosystem service indicators, rather than all available indicators. Moreover, our review was based on ecosystem services literature and limited disciplinary studies. Future research could benefit from integrating more ecological research to quantify underpinning ecosystem properties. More empirical evidence is especially needed on the actual use of ecosystem services and trade-offs between services. Surprisingly, research is mainly limited to use preference rather than actual use (Walters 2005b, Rönnbäck et al. 2007). Trade-offs between ecosystem services, such as between raw material harvest and carbon storage, and fishing and nursery service are also understudied (Sheridan and Hays 2003, Alongi 2012). Future research could contribute to better information on the consequences of human decisions and can build on the set of ecosystem service indicators we compiled. Our analysis was limited by the selected ecosystem services, but this selection was made in dialogue with decision makers. We, consequently, ignored poorly studied but important other mangrove ecosystem services, such as other foods (Table S2.1, Appendix 2), water provision for aquaculture ponds, medicinal resources (Table S2.1, Appendix 2), salt water intrusion and spiritual enrichment (Rönnbäck et al. 2007, Walters et al. 2008). Because most of these services are provided by *natural* and *low intensity use mangroves*, we consider or current results as underestimating total ecosystem service provision.

We assessed ecosystem service provision per management regime by relating characteristics of our management regimes with ecosystem service indicators. Combining qualitative and quantitative indicators enabled a comprehensive comparison of service provision per management regime, including services for which little empirical evidence exists, such as carbon storage, coastal protection and nursery service. We argue that differences of these regulating services are better explained by qualitative information (i.e. traits) because complex ecological processes underpinning service provision have not been sufficiently quantified. Moreover, our ecosystem service scores per regime integrate and quantify qualitative findings. If we had only considered quantitative indicators, our analysis would have excluded the coastal protection and nursery services and would have been more limited for the other services. The key indicators for assessing management effects on ecosystem services were mangrove age (and related height, diameter etc.), species richness and structural diversity. These indicators could be used for monitoring management regimes and imply that ecosystem service provision per management regime will change over time. The high amount of uncertain results is the result of lacking empirical studies, but we are confident that our approach has resulted in finding robust relative differences between most management regimes.

Policy-relevant research of mangrove management and ecosystem services could benefit from more systematic integration of ecological research with land use, economic and management research (c.f. Peña-Cortés et al. 2013, Verburg et al. 2013a). This integration is relevant because mangroves are continuously pressured by humans and ecological research has been conducted for decades. Following our research approach, future research should focus on quantifying and modelling all linkages between management and ecosystem properties, ecosystem properties and mangroves' capacity to provide services, and, finally, the socio-economic and cultural value of mangrove ecosystem services (Figure 4.1). These future results could be mapped per management regime based on the various ecological and management indicators. Furthermore, our approach and management regime typology can facilitate a more integrated valuation of mangrove ecosystem services for each management regime (Barbier et al. 2011).

Most ecosystem services research in mangroves has focused on comparing provision of few services (e.g. wood, shrimp and carbon storage) in two or three regimes. Examples include *natural mangroves* compared to *plantations* (Ong 1993, Rönnbäck et al. 2007, Bosire et al. 2008) and comparisons of different *aquaculture* systems (Gautier 2002, Rönnbäck et al. 2003). The most comprehensive study was done by Gilbert and Janssen (1998), who analysed multiple services provided through different 'management alternatives' (i.e. strategies). They assigned these strategies to a mangrove area by altering basic indicators and spatial configurations. They suggested alternatives that correspond to our management regimes, such as 'preservation' (*conservation*), 'subsistence forestry' (*protection*) and 'aqua-silviculture' (*silvo-fishery*). A major difference, however, is that their 'management alternatives' formed spatially explicit scenarios for ecosystem service provision. Moreover, the alternatives were based on unclear methods and linked to very few measurable indicators. Gilbert and Janssen (1998) were able to conduct much empirical research on fish, raw materials and other provisioning services, but combined expert judgment and general assumptions for coastal protection, biodiversity and ecotourism. Because Gilbert and Janssen (1998) based their final conclusions on the value of marketed ecosystem services only, they conclude that *aquaculture* systems are the most preferred alternative, while *conservation* and *preservation* alternatives generate substantially less value. Our study compared all ecosystem services that were relevant for decision making and, consequently, 'valued' the importance of services such as coastal protection, carbon sequestration and water purification as equally important as food and raw material provision.

4.6.3 Implications for decision makers

Our typology of management regimes offers a range of different choices related to land-use planning and management. Decision makers can assess the consequences of these choices by considering the ecosystem services provided per management regime. We integrated findings on multiple ecosystem services, most of which are currently not yet considered in decision making. The results in Table 4.5 and Figure 4.3 highlight crucial differences between *natural* mangroves and mangroves *converted for aquaculture*, as well as the potential benefits of rehabilitating *aquaculture* systems. A clear advantage of communicating the results as in Figure 4.3 would be that transitions can occur relative quickly in mangroves, as compared to other ecosystems (Lewis III 2005, Walters et al. 2008). We integrated novel findings on understudied carbon emission (Kauffman et al. 2013) and flood risk (Winterwerp et al. 2013) of *aquaculture* systems, which both suggest substantial risks. These risks, as well as water pollution risk, could also be mitigated by 'hard management', i.e. constructing permanent aquaculture ponds (no sediment upwelling or drainage) and dams (flood prevention). Both management practices involve considerable costs and require constant maintenance (Winterwerp et al. 2013).

Comparing the results in Table 4.5 shows the integrated consequences of land-use purposes, but management decisions are mostly taken based on criteria, such as economic profits, biodiversity protection and employment opportunities (Ghazoul 2007, Peña-Cortés et al. 2013, Bosma et al. 2014). We therefore recommend using a multi-criteria decision analysis to identify the most desired management regimes (Schwenk et al. 2012). For example, *aquaculture* systems provide food to many but economic returns to a few individual managers and investors, whereas the disservices affect all stakeholders, including pond owners and local inhabitants. More balanced management decisions could be taken if criteria such as health, safety, employment were considered in addition to economic returns.

Our findings provide decision makers with new and comprehensive information to take better, more informed decisions on management. The findings apply to Java, which had little remaining mangroves in the 1980s but saw a gradual recovery on locations where rehabilitation, natural regeneration or active protection occurred, Because mangrove ecosystem services depend mostly on mangrove tree age, species richness and structural diversity, our findings suggest that rehabilitation and long-term protection of mangrove systems can result in steadily increasing provision of multiple ecosystem services.

4.7 CONCLUSION

We analysed the effects of different management regimes on ecosystem services in Java's mangroves, to assess the consequences of management decisions. We related a comprehensive set of qualitative and quantitative indicators of seven mangrove ecosystem services to a novel typology of mangrove management regimes. The regimes capture the full range of possible management activities and land-use purposes in Java's mangroves, because it was based on Indonesian legislation and locally verified management activities. Ecological characteristics were also established and were used to relate the management regimes to our ecosystem service indicators to quantify and assess ecosystem service provision per management regime.

Natural mangroves provide the most ecosystem services and score the best for all services but fish and shrimp. Different intensities of *aquaculture* provide high amounts of fish and shrimp but this is due to artificial inputs and occurs at the expense of all other ecosystem services. Rehabilitation of *aquaculture* systems can reverse this loss of ecosystem services, while still providing shrimp or raw materials. The management regimes represent clear goals for decision makers, but we recommend conducting a multi-criteria decision analysis to identify the most desirable management regimes.

Our approach facilitates the further integration of ecological information in ecosystem services assessments, which is important to come to more precise and appropriate quantification and valuation studies of different mangrove systems. However, further empirical research is needed on actual ecosystem services use rather than potential provision, the underpinning ecosystem properties and trade-offs between ecosystem services.

Our typology is comprehensive and relevant, and could also be applied to other regions. Our results are currently being considered by decision makers in the context of developing sustainable *aquaculture* and mangrove rehabilitation plans.

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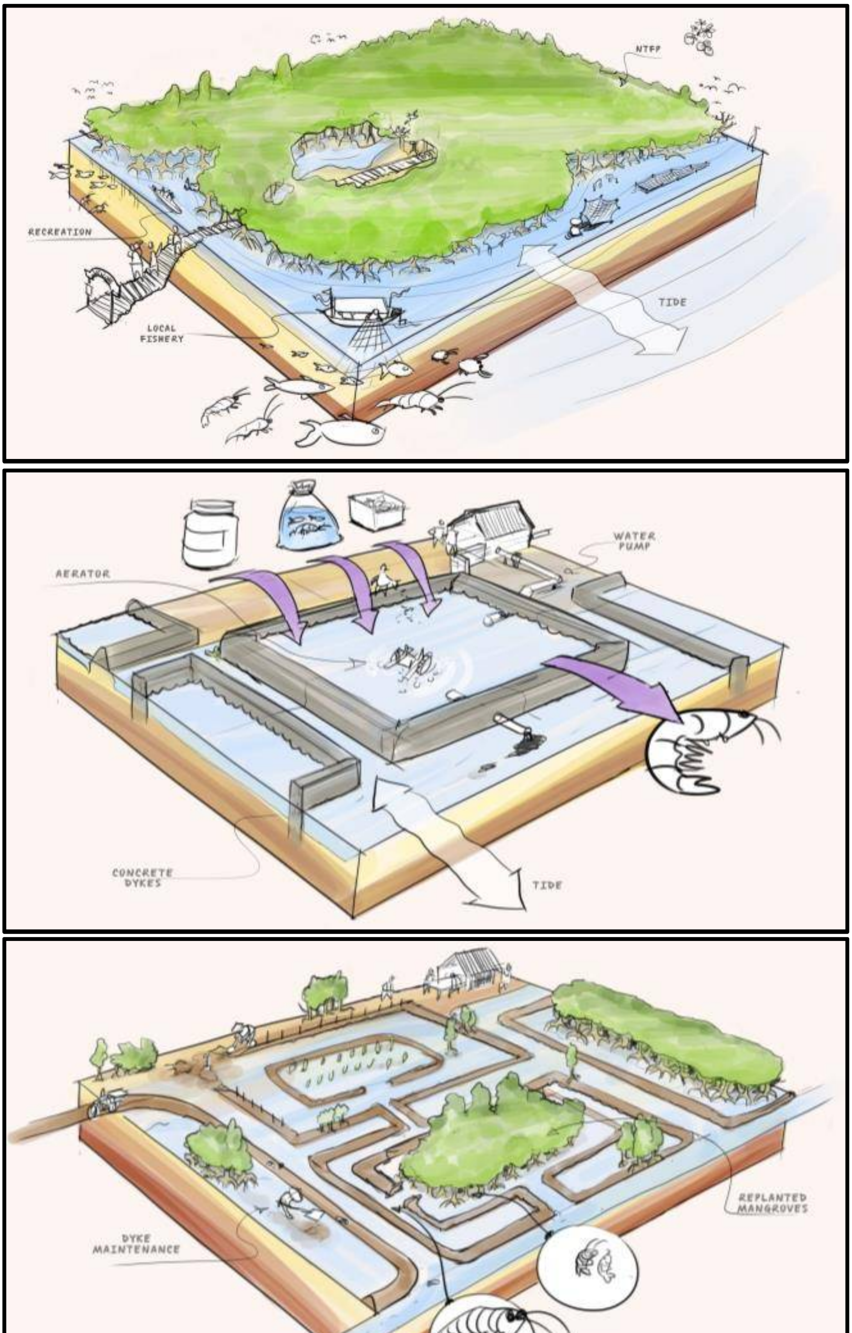

Illustrations of the management regimes Conservation, Intensive Aquaculture *and* Silvo-fishery *(from top to bottom). Illustrations by Joost Fluitsma, JAM Visual Thinking.*

A fence line separates grazed (right) from no longer grazed (left) on this semi-arid rangeland in the 'Baviaanskloof' (Eastern Cape, South Africa). The land on the right has been altered by long-term grazing, the other side has been left to recover for about thirty years after heavy grazing. Albeit slowly, canopy cover and organic matter contents have increased substantially on the abandoned rangelands, whereas soil erosion and surface water runoff on the grazed lands make the land unsuitable for further grazing and most other land uses.

5 EFFECTS OF DIFFERENT MANAGEMENT REGIMES ON SOIL EROSION AND SURFACE RUNOFF IN SEMI-ARID TO SUB-HUMID RANGELANDS

ABSTRACT

Over one billion people's livelihoods depend on dry rangelands through livestock grazing and agriculture. Livestock grazing and other management activities can erode soils, increases surface run-off and reduces water availability in semi-arid and sub-humid rangelands. We studied the effects of different management regimes on soil erosion and surface runoff in semi-arid to sub-humid rangelands. Eleven management regimes were assessed, which reflected different livestock grazing intensities and rangeland conservation strategies. Our review yielded key indicators for quantifying soil erosion and surface runoff. The values of these indicators were compared per management regime. Mean annual soil loss values in the *natural ungrazed*, *low intensity grazed, high intensity grazed rangelands* and *man-made pastures* regimes were, respectively, 717 (SE=388), 1370(648), 4048 (1517) and 4249 (1529) kg $ha^{-1} yr^{-1}$. Mean surface runoff values for the same regimes were 98 (42), 170 (43), 505 (113) and 919(267) m^3 $ha^{-1} yr^{-1}$, respectively. Canopy cover correlated negatively, while slope correlated positively to soil loss and runoff in all management regimes. Further analyses suggest that livestock grazing abandonment and *exotic plantations* reduces soil loss and runoff. Our findings underline that soil erosion and surface runoff differ per management regime, and that conserving and restoring vulnerable semi-arid and sub-humid rangelands reduce these risks.

Based on:

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Dataset available with author

5.1 INTRODUCTION

Drylands cover about 41% of the Earth's land surface and are inhabited by more than two billion people, of which 90% live in developing countries (UN 2011). Over one billion people in areas rely directly on drylands for their livelihoods, mostly through livestock grazing (65%) and agriculture (25%) (MA 2005a, UN 2011). Half of the world's livestock is supported by drylands' natural productivity (MA 2005a). The aridity index (AI) (i.e. the ratio between annual precipitation (P) and annual potential evapotranspiration (PET)) characterizes drylands, which occur in areas where AI is 0.65 (i.e. PET is at least 50% larger than P) (Middleton and Thomas 1997). Drylands are thus limited by soil moisture resulting from low rainfall and high evaporation.

Twelve to seventeen major types are distinguished, aggregated into four 'broad' biomes: desert, grassland, Mediterranean scrubland, and dryland woodlands (MA 2005a). These biomes largely follow the aridity gradient: AIs of hyper-arid, arid, semi-arid and sub-humid drylands range, respectively, from less than 0.05, 0.05 to 0.2, 0.2 to 0.5 and 0.5 to 0.65 (Middleton and Thomas 1997). In this study we focus on semi-arid and sub-humid drylands and will refer to them as 'rangelands', unless specified differently.

Land degradation is a common threat to semi-arid and sub-humid rangelands. Population increase, climatic variations and human activities (i.e. management) drive land degradation (MA 2005a, UN 2011). Degradation refers to reduced or lost biological or economic productivity and complexity of both natural and managed rangelands (MA 2005a). Approximately one fifth of all rangelands are currently suffering from degradation (MA 2005a).

Rangelands face the biggest risk of degradation (UN 2011) . Rangelands are dominated by grasses, forbs, shrubs and dispersed trees (Westoby et al. 1989). Rangelands are often associated with grazing and managed by ecological or low intensity management (Jouven et al. 2010). Most rangelands are grazed by livestock but natural rangelands also exist without livestock (Jouven et al. 2010). Semi-arid and sub-humid rangelands cover 56 million km^2 globally and are sensitive to management effects and climate variability (UN 2011). Sub-humid rangelands are, due to their higher water availability, increasingly used for intensive livestock grazing and cropping. Semi-arid rangelands, especially in the Mediterranean, have been grazed since the late 1900s. This relatively 'recent' disturbance has resulted in a transition from grass-dominated to shrub-dominated rangelands and led to increased rain-induced soil erosion and increased surface runoff.

The effects of rangeland management and land-use change on degradation and productivity are poorly understood (UN 2011). Preventing soil erosion and runoff is crucial to reverse degradation and improve productivity. MA (2005b) and TEEB (2010b) acknowledged this by including soil erosion prevention and water flow regulation as important ecosystem services (i.e. the contributions to human wellbeing). Both ecosystem services depend on similar underlying ecological characteristics (Marques et al. 2007, Fu et al. 2011). Soil erosion prevention reduces loss of productive land, downstream water pollution, clogging of waterways, flood risk and improves productivity (Snyman 1999, Fu et al. 2011). Reducing surface runoff provides similar benefits, as well as constant water availability to vegetation, decreased sedimentation and nutrient loss (Narain et al. 1997, van Luijk et al. 2013). Rangeland management is a crucial factor to consider because it negatively or positively affects soil erosion and runoff.

This study assessed consequences of management decisions in semi-arid and sub-humid rangelands by studying the effects of different management regimes on soil erosion and surface runoff. Management regimes are 'the bundle of human activities that serve land-use purposes' and reflect different land-use intensities. We limited our analysis to regimes that graze livestock at different intensities, as well as rangeland restoration and conservation. We developed a comprehensive typology of management regimes in these semi-arid and sub-humid rangelands and identified indicators to quantify soil erosion and runoff. By comparing different management regimes we identified regimes with the least erosion and optimal runoff.

5.2 METHODS

5.2.1 Indicator selection for soil erosion and surface runoff

A literature review that started with well-cited review papers on soil erosion and surface runoff yielded common indicators for soil erosion and runoff as well as indicators that characterized distinctive rangeland management regimes. Kosmas et al. (1997), Cantón et al. (2001), Fu et al. (2009), García-Ruiz (2010) and Fu et al. (2011) listed recurring indicators to quantify soil erosion and runoff. A further review confirmed which of these indicators were frequently used in quantitative assessments. Several publications that cited these five review papers focused on management effects on soil erosion and/or runoff and helped to find the necessary data.

5.2.2 Developing a management regime typology

The five review studies also informed our management regimes typology of semi-arid and sub-humid rangelands. Our approach was similar to Chapter 4's approach on mangroves. Each management regime results in distinguishable land use activities, land cover and specific ecological and socio-economic characteristics. Land use is the purpose for which humans change land cover to their own benefit (Fresco 1994, Verburg et al. 2011) and consists of a series of different activities. Land cover refers to all physical biotic and abiotic components that make up landscapes, including vegetation, soils, cropland, water and human structures (Young 1994, Verburg et al. 2009). Our management regime typology included: natural, low intensity use, high intensity use, converted and abandoned. Moreover, management regimes are assumed to be hard to reverse and transitions from one regime to another are only possible through substantial management actions (Westoby et al. 1989).

We only considered studies dealing with livestock grazing and nature conservation (e.g. restoration, protection, abandoning grazing) in rangelands and converted rangelands in semi-arid or sub-humid areas. The studies' aridity zone was verified using a 10' 'Global map of aridity' (FAO 2014). Locations were approximated when limited information was provided. When aridity zones mentioned in the study's site description did not match ours, we used the original description if the study sites were located between two aridity zones or if the study was conducted before 1990. We ignored the study's description if it was located more than 300 km away from the claimed aridity zone. We reviewed suitable studies for different management regimes. Indicators of management regimes are summarised in Table 5.1.

Table 5.1: Indicators used for developing management regimes in semi-arid and sub-humid rangelands

Indicator	**Short description** ***Categories plus abbreviations***	**Sources**
Stocking rate	Stocking rate relative to rangeland's grazing capacity. Low (L) is below the grazing capacity (<50%), high (H) is around grazing capacity and overgrazed (O) is much above grazing capacity.	McIvor et al. (1995); Dormaar and Willms (1998); Oztas et al. (2003)
Rangeland condition	Rangeland's ecological status (botanical composition and cover, and plant successional state), and its productivity, nutritive value and palatability. Categories: *Poor (P), Good (G), Degraded (D)*	Snyman (1999); Fynn and O'Connor (2000); Lechmere-Oertel (2003)
Vegetation cover	Vegetation cover in response to livestock grazing. This descriptive indicator includes mature vegetation, grass, invasive woody species and bare soil.	Stringham et al. (2003); Puttick et al. (2011); Manjoro et al. (2012)
Exclosing or enclosing	Exclosing involves disabling livestock grazing with fences and enclosing enables more localised grazing. Fences or natural barriers are used. Categories: *Exclosing (Ex), Enclosing (En)*	Dormaar and Willms (1998); Reeder et al. (2004); de Aguiar et al. (2010)
Intercropping	The occurrence of trees combined with grazing land. Trees can be natural or planted and grazing land can include rangelands and sown pastures. Categories: *Yes (Y), No (N)*	McIvor et al. (1995); Narain et al. (1997); de Aguiar et al. (2010); Gelaw et al. (2014);
Soil treatment	Treating the topsoil layer to optimise livestock grazing. Examples include removing topsoil, removing weeds, ploughing and mulching. Categories: *Yes (Y), No (N)*	Simanton et al. (1985); Mwendera and Saleem (1997)
Vegetation removal	Removing unwanted vegetation, that inhibits grass production. Examples include woody, unpalatable and water-consuming species. Categories: *Yes (Y), No (N)*	Simanton et al. (1985); McIvor et al. (1995); de Aguiar et al. (2010)
Restoring or planting	Planting exotic (Ex) or natural (Nat) vegetation to reduce erosion, increase wood production etc.	Narain et al. (1997); Andreu et al. (1998)
Sowing grass	Sowing nutritious species with the aim to maximise nutrient intake of livestock. Categories: *Yes (Y), No (N)*	McIvor et al. (1995); Narain et al. (1997)
Fertilizers, pesticides, herbicides use	Using fertilizers (F), herbicides (H) and/or pesticides (P) to improve grass productivity.	Carlson et al. (1990); Narain et al. (1997)

Several assumptions were made to apply the indicators to a large variety of rangeland ecosystems and to cope with different ways how rangeland management and land use are described. No quantitative ranges were determined for stocking rate intensities, because these depend on local factors that differ throughout the world's rangelands. Intermediate classes between low and high, and high and overgrazing were ill-defined and highly variable, and thus not considered. Many studies also report the 'rangeland condition' and/or vegetation cover in response to different intensities of grazing without referring to actual stocking rates. These indicators are frequently used in traditional rangeland ecology studies (e.g. Snyman 1997, Puttick et al. 2011). Rangeland condition and/or vegetation cover approximate stocking rates. For instance, poor, good and degraded rangelands could generally be linked to low, high and overgrazed stocking rates (Snyman 1997, Fynn and O'Connor 2000). Rangeland condition involves ecological status (i.e. botanical composition and cover, and plant successional status, productivity, nutritive value and palatability) (Snyman 1999). Water use efficiency, above-ground production and basal cover generally decrease when rangelands degrade and the vegetation deteriorates (Snyman 1999, Puttick et al. 2011).

Vegetation cover, which is a qualitative indicator, can be used to describe distinctive management-induced ecological states (Westoby et al. 1989). Although semi-arid and sub-humid rangelands comprise many different ecosystems, we used the rangeland ecology literature to confirm the rangelands' management-induced vegetation cover.

For the 'exclosing/enclosing livestock' indicator we assumed that roaming wildlife would be excluded. Exclosure or enclosure's duration was only considered when distinguishing between abandoned and conservation rangelands. The indicator 'intercropping' applies to both sown pasture systems and natural rangelands. We considered intercropping when trees were combined with high stocking rates. Low stocking rates generally exclude enclosing grazing areas and thus by default occur on rangelands were trees could occur naturally. Finally, most studies mention fertilizers, pesticides or herbicides only as a pre-treatment prior to experiments or new grazing regimes. Very few studies mention them as current activities, except for pasture management.

We created eleven different management regimes based on the indicators in Table 5.1 and the literature's descriptions of 'management regimes'. We cross-tabulated the regimes and indicators to develop our unique typology (Table 5.2). Each management regime is characterised by at least two differing indicators from other regimes.

5.2.3 Indicator selection for soil erosion and surface runoff

Data and supporting information were retrieved from the reviewed literature. When available, we gathered both unique measurements (e.g. soil loss kg ha^{-1}) and averages (e.g. soil loss kg ha^{-1} kg yr^{-1}). Graphical data was extracted using Plot Digitizer (version 2.6.3). All data was stored. Data that was linked to multiple indicators was entered as one data entry. Apart from quantified indicators, we also registered location (coordinates, country, location description), measurement date, average temperature, annual precipitation and biome type (IMAGE and TEEB typologies), ecosystem (TEEB), land cover (GLC 2000) and land use (IMAGE-GLOBIO and our own typology).

We assigned a management regime to each data entry based on matching indicators mentioned in the study. We used 'stocking rate' as a guiding indicator and rangeland condition or vegetation cover to either verify the mentioned stocking rate, or to indicate the stocking rate indirectly. However, qualitative terms for stocking rate, such as 'high', 'heavy', 'low', 'overgrazed' and 'degraded', were used inconsistently. We only took a given stocking rate when it was related to the rangeland's grazing or carrying capacity (c.f. Table 5.1). Quantitative stocking rates were sometimes used to compare intensities within studies. Data entries that could not be linked to management regimes (e.g. bare soil and cropland measurements), were excluded from our analysis. Additional literature ensured that all management regimes had quantitative information on both soil erosion and surface runoff. We used Web of Science™ and Google Scholar™, and used the management regimes' names or synonyms as keywords combined with *erosion* and *runoff*. Results were sorted by relevance and the top-50 papers were selected and checked for quantitative data. In total, we retrieved data from fourteen studies that quantified soil erosion (141 data entries), eleven on surface runoff (73 data entries) and seventeen that quantified both soil erosion and surface runoff (205 data entries). The forty-two studies were divided over twenty-six papers. Figure 5.1 shows the locations reported in the papers. Analyses were done with 267 data entries on soil erosion measurements and 283 on surface runoff measurements.

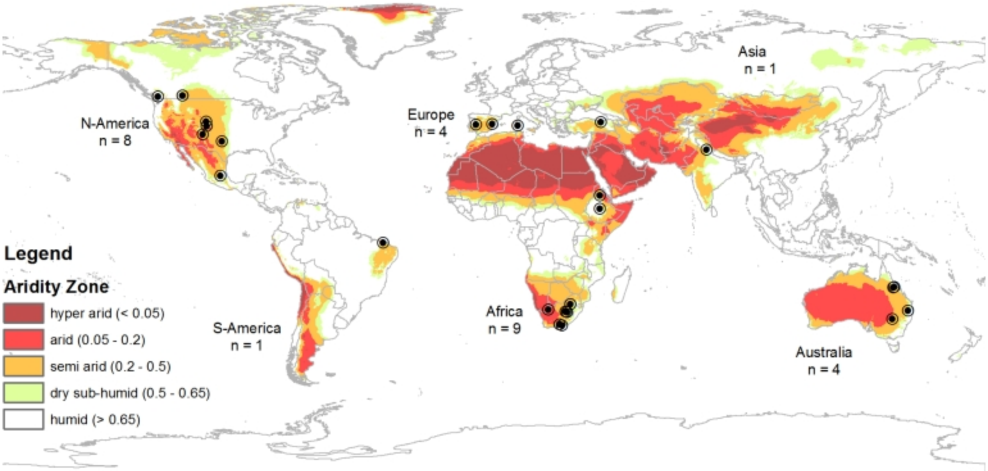

Figure 5.1: Analysed studies' reported locations on the Global Aridity Map (FAO 2014).

Data was analysed with SPSS Statistics version 22. Mean values were obtained for all quantitative indicators per management regime. Differences between means were not further tested due to large differences in number of studies and data entries per management regime. We conducted a Spearman's rank order correlation between some underlying indicators and soil loss and runoff to identify key indicators that might explain differences in soil loss and runoff.

Further analyses were performed with only the studies that compared soil loss and surface runoff per management regime. Mean values for soil loss and surface runoff were obtained per study and differences between means were subjected to a T-Test ($p < 0.05$). Mean values were averaged when multiple studies had quantified the same management regimes. We used eight out of thirteen studies on soil loss and three out of eleven on surface runoff for this analysis.

5.3 MANAGEMENT REGIME TYPOLOGY

We developed eleven management regimes for semi-arid and sub-humid rangelands and considered livestock grazing and nature conservations as land-use purposes. The five broad categories of management regimes (Table 5.2) are defined as follows:

- *Natural rangelands* are not grazed by livestock and have a recognised high biodiversity value or ecological function. Management activities are limited to nature protection (fences, patrolling etc.) but vegetation and soils are undisturbed. The category includes *natural ungrazed* (e.g. Launchbaugh 1955, Andreu et al. 1998) and *conservation rangelands* (e.g. Reeder et al. 2004, van Luijk et al. 2013).
- *Low intensity use rangelands* are managed to support low intensity livestock grazing or restored natural vegetation. Management activities do not involve infrastructure construction and modify vegetation cover. Rangelands include *low intensity grazed* (e.g. McIvor et al. 1995) and *restoration rangelands* (e.g. Andreu et al. 1998).
- *High intensity use rangelands* are managed for optimised livestock grazing. Management activities include intensive grazing, introducing highly palatable grass species, intercropping trees and pastures, and using chemical inputs to optimise grass productivity.

The category includes *high intensity grazed* (e.g. Mwendera and Saleem 1997) and *overgrazed rangelands* (e.g. Oztas et al. 2003) and *silvo-pastures* (e.g. McIvor et al. 1995, de Aguiar et al. 2010).

- *Converted rangelands* are systems in which the original vegetation has been cleared and replaced to serve another land-use purpose, such as livestock grazing and tree plantation. Management activities can be sowing grass, planting trees, irrigating, applying pesticides and herbicides, and ploughing. This category includes *man-made pastures* (e.g. McIvor et al. 1995) and *exotic tree plantations* (e.g. Narain et al. 1997).
- *Abandoned rangelands and pastures* have been used intensively or unsustainably, and are currently without a land-use purpose. This category includes both recovering *abandoned rangelands* (e.g. Descheemaeker et al. 2006) and irreversibly *degraded abandoned rangelands* (e.g. Muñoz-Robles et al. 2011).

Most management regimes are characterised by distinctively different stocking rates or livestock management (Table 5.2). Moreover, increasing stocking rates coincide with increasing additional management efforts and inputs, such as vegetation removal, soil treatment and applying herbicides. Along the typology, vegetation cover changes from mature vegetation (*natural rangelands*) to more grassy species (*low intensity grazed*) and introduced grass and herbaceous species (*high intensity grazed*). *Overgrazed rangelands* are characterised by bare soils and woody encroachment, whereas *abandoned degraded rangelands* are suffering from desertification and increased woody encroachment (Puttick et al. 2011, Manjoro et al. 2012). Rangeland degradation involves irreversible changes in both soils and vegetation (Fynn and O'Connor 2000).

Several other management regimes aim to restore rangelands' productivity and/or original biodiversity and additional indicators are needed to distinguish regimes that aim to reverse land degradation and restore or conserve rangelands. For instance, *conservation* and *restoration rangelands* are both 'exclosed', but active restoration management (i.e. replanting, removing alien vegetation etc.) only takes place in the latter. In fact, *conservation rangelands* could be described as 'undergrazed', since neither livestock nor free-roaming wildlife grazes here. *Silvo-pastures* aim to increase the rangelands' productivity through intercropping with trees. Exotic *tree plantations* only aim to reverse soil erosion but could actually have other negative effects on rangelands' productivity and biodiversity. *Abandoned rangelands*, finally, are characterised by a shorter period of not grazing and could either be used for livestock grazing or proceed to become conservation rangelands.

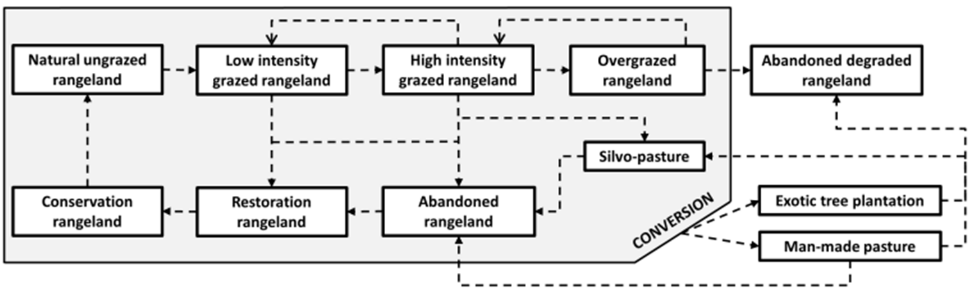

Figure 5.2: Possible transitions between management regimes of semi-arid and sub-humid rangelands. Based on Bellamy and Brown (1994), Stringham et al. (2003) and Alkemade et al. (2013).

Table 5.2: Short description and management indicators of management regimes in semi-arid and sub-humid rangelands. Management indicators are further explained in Table 5.1, acronyms are repeated below.

Management regime	Short description	Stocking rate	Ex- or en-closing	Inter-cropping	Soil treat-ment	Vegetation removal	Restoring/ planting	Sowing grass	Using F/P/H
I. NATURAL									
Ungrazed rangeland	Grazed by free-roaming natural grazers only. Good rangeland condition; undisturbed mature vegetation.	-	-	N	N	N	-	N	-
Conservation rangeland	All grazing disabled for > 40 years, to optimize vegetation recovery. Good rangeland condition.	-	Ex	N	N	N	Nat	N	-
II. LOW INTENSITY USE									
Low intensity grazed rangeland	Livestock grazing below the carrying capacity. Some palatable grasses persist. Good rangeland condition	L	-	Y	N	N	-	N	-
Restoration rangeland	Actively restored former grazing land.	-	Ex	N	N	Y	Nat	N	-
III. HIGH INTENSITY USE									
High intensity grazed rangeland	Livestock grazing at carrying capacity. Altered vegetation and soils. Poor rangeland condition.	H	En	N	Y	Y	-	Y	F,H
Overgrazed rangeland	Continuous grazing above carrying capacity. Degraded condition; woody encroachment and some bare soils.	O	En	N	Y	Y	-	N	F
Silvo-pasture	Rangelands or sown pastures intercropped with trees to provide shade, fodder or to prevent erosion.	L/H	En/Ex	Y	Y	Y/N	Nat/Ex	Y	P,H
IV. CONVERTED									
Man-made pasture	Original vegetation cleared and replaced by optimal grass for livestock grazing on pre-treated soils.	L/H	En	N	Y	Y	-	Y	F,P,H
Tree plantation	Exotic trees planted on formerly degraded land, with the aim to reduce soil erosion and produce wood.	-	Ex	N	Y	Y	Ex	N	P,H
V. ABANDONED									
Abandoned rangeland	Rangelands or pastures relieved from grazing for <30 years, allowing the vegetation to recover.	-	Ex	N	N	Y	-	N	-
Abandoned degraded rangeland	No longer grazed or used due to irreversible changes in soils (bare) and vegetation (woody encroachment).	-	-	N	N	-	-	N	-

Note: Dashes (-) indicate when information could not be found or indicators do not apply.
Acronyms: Low (L), High (H), Overgrazed (O), Yes (Y), No (N), Natural vegetation (Nat.), Fertilizer (F), Pesticide (P), Herbicide (H).

Possible transitions between management regimes are illustrated in Figure 5.2. These transitions include land-use intensification, restoration or reducing land-use intensity. Each transition requires additional and prolonged management activities (Stringham et al. 2003). All regimes could lead to conversion into *exotic tree plantations* or *man-made pastures*. *Abandoned degraded rangelands* are mostly 'end of line' management regimes, where stepwise restoration is impossible. Figure 5.2 will be used in Section 5.5 to illustrate differences in soil erosion and surface runoff between management regimes. The figure helps to inform decision makers because all management regimes and transitions between them represent clear management choices.

5.4 INDICATORS FOR QUANTIFYING SOIL EROSION AND SURFACE RUNOFF

The frequency of recurring indicators from the reviewed literature for soil erosion and surface runoff is provided in Table S4.1 (Appendix 4). Most studies assessed both erosion and runoff. Only fourteen studies assessed just soil erosion and eleven assessed runoff, respectively. We distinguished three categories because they represent different research aims and, thus, require different indicators. We collected twelve basic indicators for both soil erosion and runoff. Key indicators are described below. Studies that assess both erosion and runoff, use more recurring indicators on a consistent basis. Moreover, studies on just erosion or runoff used indicators that were rarely used by other studies. We note, for example, that annual soil loss was only measured by two studies that focused on erosion only, as compared to eleven that assessed both services. Similarly, annual surface runoff was only measured by three studies, as compared to eight that assessed both services. A possible explanation could be that mono-disciplinary studies generally follow a more detailed research approach.

How all indicators (Table S4.1 in Appendix 4) are related to each other is illustrated in an interaction diagram (Figure 5.3). This diagram depicts information flows rather than matter flows, and connects indicators rather than processes or systemic components. Therefore, the interaction diagram does not explain the dynamic complexity of soil erosion and runoff. 'Soil loss' per area (and per year) approximates soil erosion and 'surface runoff' per area (and per year) approximates surface runoff (e.g. Narain et al. 1997, Cantón et al. 2001, Fu et al. 2011). Most values for 'annual soil loss' and 'annual surface runoff' include all measured rain events of a given year or averaged values of multiple years. Some studies do not specify whether they assessed surface runoff or total runoff (i.e. including sub-surface runoff and drainage). For most studies, we could establish what was measured from the experimental setup, but 'soil loss' and 'surface runoff' that is only measured per unit of area, lacks the temporal dimension. Standardizing and comparing data is therefore difficult. Some studies measured individual rain events, while others focused on seasons, years or longer periods. Finally, hydrological studies often measure runoff as a percentage of total rainfall (i.e. the 'surface runoff coefficient'), which can also be daily, seasonal or annual rainfall. The indicators underpinning soil loss and surface runoff can be grouped into four categories: rainfall, vegetation, and topography and soil (Figure 5.3). The indicators are described below.

Rainfall is positively correlated with soil loss and surface runoff (Kosmas et al. 1997, Marques et al. 2007, Vásquez-Méndez et al. 2010). Although 'annual rainfall' (m^3 ha^{-1} yr^{-1}) can be useful to relate annual soil loss or surface runoff (Le Maitre et al. 1999), most studies provide rainfall data rather than measuring and incorporating soil loss or runoff explicitly. More common indicators are 'total rainfall' (m^3 ha^{-1}) resulting from hydrological experiments, or 'rainfall erosive events' (m^3 ha^{-1}) resulting from erosion experiments. Both indicators approximate the amount of rainfall during a given observation period (Bartley et al. 2006). 'Rainfall intensity' indicates hourly rainfall per area and informs on soil loss and runoff during high and low intensity rainfall events. This indicator is used infrequently because erosion and runoff events are usually assessed over longer periods. Most studies also describe the period during which most rainfall occurs or was measured (i.e. the 'rainfall regime'). This information adds merit to annual rainfall, erosion and runoff, because peaks are identified (Cerda et al. 1998). The amount of rainfall that is not intercepted by vegetation (i.e. 'throughfall'), largely determines the amount of surface runoff (Mills and Fey 2004). Most reviewed studies mentioned throughfall, but 'interceptional loss' (i.e. rainfall minus throughfall) could be calculated. Interception rates are established for different vegetation types (e.g. Dunkerley 2000). 'Evapotranspiration' describes the amount of rainfall that is returned directly to the atmosphere by transpiration (from plants) and evaporation (from soils) (Paço et al. 2009). Most studies, however, establish evapotranspiration indirectly without relating it to surface runoff.

'Vegetation cover' (i.e. % of total land cover) is negatively correlated to both soil loss and surface runoff (Vásquez-Méndez et al. 2010, Fu et al. 2011). Vegetation intercepts raindrops, reduces raindrop impacts and promotes infiltration pathways (Le Maitre et al. 1999). Land management activities and ecosystem and vegetation type determine vegetation cover's structure and density (e.g. Le Maitre et al. 1999, van Luijk et al. 2013). For instance, plant communities in

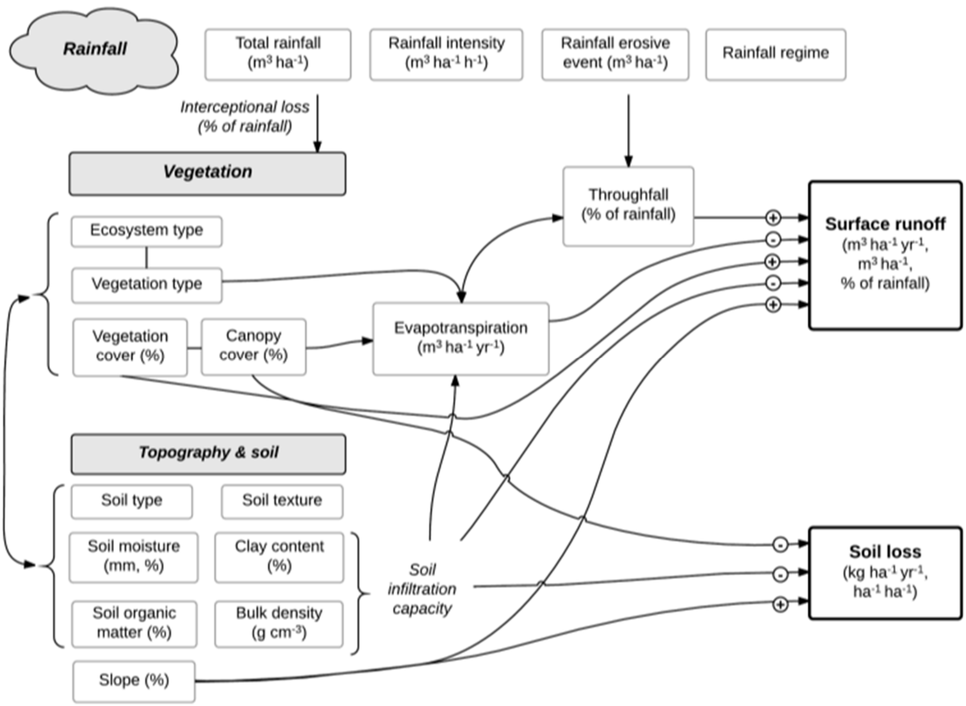

Figure 5.3: Interaction diagram for all the indicators approximating soil erosion and surface runoff. Their units are listed between parentheses.

semi-arid scrublands take up water more efficiently than plants in sub-humid ecosystems (Le Maitre et al. 1999, Mills and Fey 2004). High vegetation covers can be dominated by low grasses, whereas lower vegetation cover can be characterised by a higher canopy cover and, thus, potentially lower runoff and soil loss (Dunkerley and Booth 1999, Silburn 2011).

Below the vegetation, the 'soil infiltration capacity' determines soil loss and surface runoff (Descheemaeker et al. 2006). This capacity is difficult to measure and mostly approximated by interrelated indicators such as 'soil moisture', 'clay content', 'soil organic matter' and ´soil bulk density' (Snyman 1998, Vásquez-Méndez et al. 2010). The individual relations between these factors and soil loss and runoff vary per vegetation and soil type. In general high soil organic matter combined with low soil bulk density increase infiltration capacity (Snyman 1998). The soil-related indicators have limited predictive value in isolation, but, once combined, they usefully approximate the soil infiltration capacity. Finally, slope is a major factor that reduces soil loss and surface runoff (Vásquez-Méndez et al. 2010, Fu et al. 2011). The role of slopes (measured in degrees or percentage) on both processes is more important in sparse vegetation than in dense vegetation (Descheemaeker et al. 2006, Zheng 2006).

5.5 RESULTS

Our analysis provided quantitative mean values for all indicators of soil loss and surface runoff, distributed over all eleven management regimes (Table 5.3). No surface runoff data was available for *abandoned degraded rangelands*. *High intensity grazed rangelands* were by far the most common management regime: 58% of the data entries on soil erosion and 36% on surface runoff related to this management regime. Other prevalent management regimes among soil erosion and surface runoff studies were *ungrazed natural rangelands* (10% each) and *low intensity grazed rangelands* (8% and 11% respectively). Both analyses yielded little data for *overgrazed rangelands* and data for the two *abandoned rangeland* regimes was especially limited for soil erosion.

5.5.1 Soil erosion and surface runoff per management regime

Table 5.3 depicts some clear trends for both soil erosion and surface runoff. Compared to *ungrazed natural rangelands*, both soil loss and surface runoff increase notably with increasing grazing intensity.

The average annual soil loss in *high intensity grazed rangeland, silvo-pasture* and *man-made pasture* differed substantially. Soil loss in *low intensity grazed rangeland* was notably lower than in more intensive management regimes. The mean soil loss in *overgrazed rangeland* was extremely high, but this should be treated with caution because only two data entries were available. The high annual soil loss found for *silvo-pasture* results from including studies on intercropped trees with sown pastures rather than natural rangelands. The limited data for 'natural' *silvo-pastures* suggests annual soil loss values around or below that of *ungrazed natural rangelands* (McIvor et al. 1995). Interestingly, soil loss in both *tree plantations* and *restoration rangelands* is considerably lower than in most other regimes. Both regimes aim to prevent or restore soil erosion. The low soil loss in *restoration rangelands* can, however, mostly be attributed to the predominantly flat surface measurements. Surface runoff values follow a similar pattern to that of soil loss and observations in *silvo-pastures* and *restoration rangeland* apply for surface runoff values as well. Little data was available for soil loss and surface runoff per unit of area (Table 5.3). Higher means for soil loss and

Table 5.3: Mean values ($\overline{x}$) for soil loss and surface runoff per management regime. The standard error (SE) is given after each mean, followed by the number of data entries (n).

		Annual soil loss (kg ha^{-1} yr^{-1})		Soil loss (kg ha^{-1})		Annual surface runoff (m^3 ha^{-1} yr^{-1})		Surface runoff (m^3 ha^{-1})		Surface runoff coefficient (%)	
Management Regime		$\overline{x}$	SE (n)	$\overline{x}$	SE (n)	$\overline{x}$	SE (n)	$\overline{x}$	SE (n)	$\overline{x}$	SE (n)
I.	Natural ungrazed	717	388 (17)	244	88(11)	98	42(13)	73	24(11)	40	5(6)
	Conservation	*no data*		*no data*		508	0 (1)	*no data*		0 (1)	(1)
II.	Low intensity grazed	1370	648(9)	1385	372 (10)	170	43(8)	314	210(4)	29	7(14)
	Restoration	126	28(16)	*no data*		66	15(16)	*no data*		*no data*	
III.	High intensity grazed	4048	1517(22)	500	85(134)	505	113(19)	2227	705(12)	21	2(70)
	Overgrazed	9915	3105(2)	*no data*		810	264(3)	*no data*		22	9(3)
	Silvo-pasture	3348	1029(14)	*no data*		894	209(12)	236	0(1)	5	1(2)
IV.	Man-made pasture	4249	1529(7)	*no data*		919	267(7)	164	0(1)	*no data*	
	Tree plantation	89	211(14)	*no data*		254	56(16)	761	341(2)	*no data*	
V.	Abandoned	2705	1275(2)	100	0(1)	143	84(2)	478	133(21)	16	3(4)
	Abandoned degraded	*no data*		90	8 (7)	*no data*		*no data*		*no data*	

surface runoff in *high intensity grazed* as compared to *low intensity grazed rangelands* seem of limited value since these measurements were based on different lengths of time. The mean runoff coefficient was slightly higher in *natural ungrazed rangelands* compared to other the regimes but did not differ much between more intensive management regimes.

5.5.2 Underlying indicators for soil erosion and surface runoff

We compiled mean values of underlying indicators for soil loss and surface runoff (Table S4.2 and Table S4.3 in Appendix 4). Additionally, we assessed common soil type and texture per management regime. Management regimes with low soil loss and surface runoff were dominated by Udic and Lithic Aridic Haplustalf, as well as Calix Xerochrept soils. Soil texture of these regimes ranged from silty to sandy loam. Management regimes with high soil loss and surface runoff were characterised by many different soil and texture types. This logically follows from the many studies on *high intensity grazed* systems. Around half of the studies reported soil type and texture but other trends were unclear.

Poor data availability limited to observe trends in rainfall-related indicators linked to soil loss. Soil loss in *high intensity grazed rangelands* was generally related to high rainfall intensity (Table S4.2 in Appendix 4), but this could not be reliably compared to other management regimes. Vegetation cover did not differ among management regimes, regardless of differences in soil loss between them. Vegetation cover was even among the highest in *high intensity grazed rangeland.* However, canopy cover was notably higher in regimes with low soil loss values (70-80% vs. 5-40%, Table S4.2). Additional correlation analysis among all data entries yielded a negative but not statistically significant correlation between canopy cover and soil loss (Spearman's ρ = -0.331). Data on the soil-related indicators was not available for all management regimes, but, showed some interesting trends. Soil bulk density was fractionally lower in *high intensity grazed* as compared to *low intensity grazed rangeland* (1.2 vs. 1.3 g cm^{-3}) while clay contents of the same management regi-

Table 5.4: Mean values ($\overline{x}$) of soil loss and surface runoff for each management regime, but limited to slopes between 0 and 10%. The standard error (SE) is given after each mean, followed by the number of data entries (n). Abandoned degraded rangelands were excluded due to limited data.

	Management regime	Annual soil loss ($kg\ ha^{-1}\ yr^{-1}$) $\overline{x}$	SE (n)	Annual surface runoff ($m^3\ ha^{-1}\ yr^{-1}$) $\overline{x}$	SE (n)
I.	Natural ungrazed rangeland	950	550 (2)	*no data*	
	Conservation rangeland	*no data*		*no data*	
II.	Low intensity grazed rangeland	1370	648 (9)	171	43 (8)
	Restoration	*no data*		*no data*	
III.	High intensity grazed rangeland	2338	719 (18)	563	118 (17)
	Overgrazed rangeland	9915	3105 (2)	587	246 (2)
	Silvo-pasture	3348	1029 (14)	894	209 (12)
IV.	Man-made pasture	4249	1529 (7)	919	267 (7)
	Tree plantation	899	210 (14)	255	56 (16)
V.	Abandoned rangeland	2705	1275 (2)	143	84 (2)

mes were around 33% vs. 18%. Soil organic matter contents were generally lower in *high intensity grazed* as compared to *low intensity grazed rangelands*. An additional correlation analysis of all data showed a considerable negative correlation between soil organic matter and soil loss (Spearman's $\rho = -0.757$, sig (2-tailed) < 0.01). Finally, the slopes of various management regimes differed significantly. Soil loss was measured at *ungrazed*, *restoration* and *high intensity grazed rangelands* of which the slopes were steeper than all other management regimes. These steep slopes can explain the high soil loss values because we collected many data entries for these *high intensity grazed rangelands*. However, the slopes were gentler for other regimes with both high and low soil loss, which suggests that this bias is limited. Additional correlation analysis showed a positive correlation between slope and soil loss (Spearman's $\rho = 0.386$, sig (2-tailed) < 0.01).

Similar to soil loss, we could not observe trends in rainfall-related indicators for surface runoff, again due to limited data availability (Table S4.3 in Appendix 4). This suggests that most studies report runoff either without referring to actual rainfall or by linking it immediately to the aridity zone or annual rainfall statistics. Vegetation cover did also not differ conclusively among management regimes, regardless of differences in their runoff. Similar to soil loss, canopy cover was notably higher in regimes with low surface runoff values (70-90% vs. 5-40%). Additional correlation analysis among all data entries yielded a negative but not statistically significant correlation between canopy cover and surface runoff (Spearman's $\rho = -0.362$). Indicators for soil variables also showed similar trends as compared to the soil loss analysis. Limited data on soil bulk density showed again fractionally lower values in *high intensity grazed* as compared to *low intensity grazed rangeland* (1.2 vs. 1.3 $g\ cm^{-3}$). Clay contents of these management regimes were around 31% vs. 18%. Soil moisture contents were fractionally higher in *natural rangelands* as compared to other management regimes. Slopes of all management regimes were generally steeper compared to where soil loss had been measured. Notable slope differences occurred between *high intensity grazed* (12%) and *low intensity grazed rangeland* (5%), but this difference alone is unlikely to alter surface runoff. For instance, runoff in *abandoned rangelands* was measured at a 49% slope on average, but this runoff was only fractionally higher than that of *ungrazed rangelands*. Additional correlation analysis showed a positive correlation between slope and surface runoff (Spearman's ρ = 0.328, sig (2-tailed) < 0.01). Insufficient data on evapotranspiration and throughfall prevented comparisons between the different management regimes.

Because of notable differences in slope between the management regimes for both soil loss and surface runoff, we assessed all data for gentle slopes (less than 10%), which was the most common slope category. Although this reduced the number of data entries that could be analysed, it controlled for any exaggerated slope effects while still showing interesting trends. Table 5.4 shows that trends for annual soil loss and surface runoff follow largely the same pattern as the results for all slope categories (Table 5.3).

5.5.3 Soil erosion prevention and water flow regulation as rangeland ecosystem services

The actual ecosystem services related to both soil loss and runoff (i.e. 'soil erosion prevention' and 'water flow regulation') can be determined by comparing the different indicators' values across various ecosystems with different naturalness and degradation levels (Bartley et al. 2006, Fu et al. 2011). For instance, soil loss of different land-use types is often compared to that of bare soil to determine soil erosion prevention capacity. In our study we consider both soil loss and surface runoff relative to the natural reference (i.e. *natural ungrazed rangeland*) as the potential ecosystem service (sensu Vásquez-Méndez et al. 2010). Based on soil loss and surface runoff of different management regimes relative to *natural ungrazed rangeland* and each other (Table 5.3), we can formulate potential provision of the ecosystem services 'soil erosion prevention' and 'water regulation'. Mean values for annual soil loss in the *man-made pastures*, *high intensity grazed* and *low intensity grazed rangelands* regimes were, respectively six, five and two times higher than *natural ungrazed rangelands*. Surface runoff was, respectively nine, four and two times higher. Moreover, soil loss and surface runoff was reduced in *abandoned* and *restoration rangelands*. Altogether, these results suggest that potential soil erosion prevention and water flow regulation can be provided by reducing grazing intensity and active rangeland restoration. However, these findings should be treated with caution because the differences between management regimes were not tested statistically and study bias may occur. The large standard errors of the means and differences in number of data entries per management regime should be acknowledged (Table 5.3).

Figure 5.4 shows the results of a further analysis performed with only the studies that compared soil loss and surface runoff between one or more management regimes. Changes in soil loss could be derived from eight studies, while only three studies compared surface runoff between management regimes. The results in Figure 5.4 offer preliminary insights in soil erosion prevention and water regulation involved in changing from one regime to another. Soil loss increases notably from, respectively, *natural ungrazed* and *low intensity* to *high intensity* grazed rangelands (Figure 5.4 A). *Exotic tree plantations* reduce soil loss of *man-made pasture* and *silvo-pasture*. Abandonment of *low intensity*, *high intensity* and *overgrazed rangelands* involves stark reductions in soil loss. Similar trends could be observed for surface runoff (Figure 5.4 B).

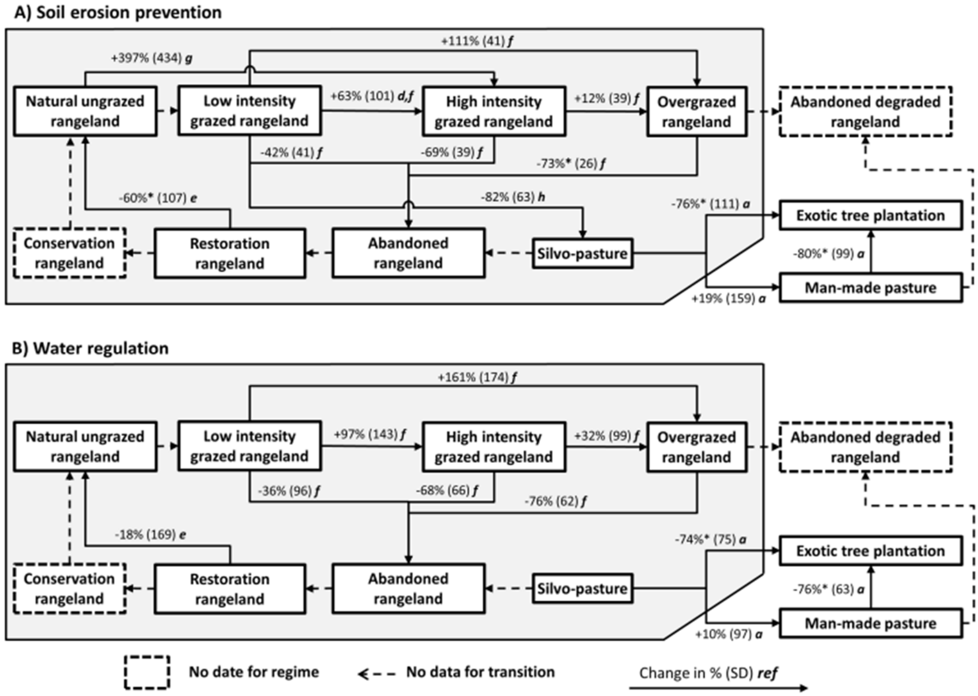

Figure 5.4: Erosion prevention (A) and water flow regulation (B) involved in transitions between management regimes of semi-arid and sub-humid rangelands. Solid arrows indicate change in soil loss (A) and water runoff (B) between two regimes. Asterisks (*) indicate significant differences ($P < 0.05$). Some unquantified transitions were omitted to improve the figures readability. Results are based on a subset of the data as presented in Table 5.3. Data sources are indicated with letters: a - Narain et al. (1997), b - Lechmere-Oertel (2003), c - Snyman and Van Rensburg (1986), d - Snyman (1999), e - Andreu et al. (1998), f - Mwendera and Saleem (1997), g - Helldén (1987), h - de Aguiar et al. (2010).

5.6 DISCUSSION AND CONCLUSION

We assessed the consequences of management decisions in semi-arid to sub-humid rangelands by studying the effects of management regimes on soil erosion and surface runoff. Our results confirm that both soil loss and surface runoff are higher in management regimes with higher livestock grazing intensity. Soil loss and surface was lower in management regimes that aim to reverse land degradation of intensive grazing (i.e. *abandoned* and *restoration rangelands*). Our preliminary assessment of transitions between management regimes suggest that increasing livestock grazing intensity indeed increases soil erosion and surface runoff. Moreover, soil loss and surface runoff are reduced considerably when *man-made pastures* are converted to *exotic tree plantations and* when intensively grazed rangelands are abandoned. Our findings suggest that management can reverse land degradation involved in all management regimes apart from *degraded abandoned rangelands*. Moreover, our research underlines the risks involved in intensifying livestock grazing in semi-arid and sub-humid rangelands.

Our findings are based on a combination of a novel typology of rangeland management regimes, which are firmly grounded in the global rangeland literature, and an extensive set of quantified indicators for soil erosion and surface runoff. The following sections discuss the management regimes typology and our findings on quantified soil erosion and surface runoff per management regime.

5.6.1 A comprehensive typology of management regimes for rangelands

Our management regimes for semi-arid to sub-humid rangelands were based on generic management indicators that reflect livestock grazing intensity, rangeland restoration or conservation. We selected indicators that indicate differences in land-use intensity, refer to key management activities, would be applicable to multiple regions and have clearly distinguishable values that would enable the separation between different management regimes. All indicators were qualitative or binary, such as stocking rate (high, low), soil treatment (yes/no) and planted, natural vs. exotic vegetation. Qualitative indicators reflect the low resolution of our analysis, but the resulting management regimes are, nevertheless, easy to distinguish and comprehensive, and reflect very clear and relevant rangeland management decisions. The selected indicators are based on research in rangeland ecology as well as livestock management, which was conducted in six continents across different temporal and spatial scales.

Developing the typology required several simplifications and this means that certain management indicators could not be considered. Semi-arid and sub-humid rangelands comprise up to ten different biomes and even more vegetation types (Dunkerley 2000, MA 2005a). Assuming that rangeland management will have the same effect on these different vegetation types is a generalization. However, our regime's characteristics reflect the strong consensus that exists on the general effects of no, low and high livestock grazing on vegetation cover (Bellamy and Brown 1994, Fynn and O'Connor 2000, Stringham et al. 2003). Species composition and biodiversity could not be accounted for, as these indicators only partly reflect differences in management and are difficult to generalise. Rangeland productivity and water use efficiency are frequently used rangeland indicators in rangeland ecology. We incorporated them into a more generic indicator called 'rangeland condition', which has a proven negative correlation with livestock grazing intensity (Snyman 1998, Allsopp et al. 2007, Jouven et al. 2010). We did not incorporate specific (semi-)arid livestock management indicators, such as mowing frequency, fire management frequency, irrigation and additional feeding (Perevolotsky and Seligman 1998, Fynn and O'Connor 2000, Todd and Hoffman 2000). These indicators would likely have resulted in only subtle variations of *high intensity use rangelands* and *man-made pastures that* would be more appropriate for regional studies. We also did not incorporate the above discussed management indicators because they have rarely been assessed in relation to soil erosion and surface runoff. We note, however, that further studies into regional rangeland management should develop regionally applicable management regimes.

Despite the vast scientific consensus on ecological impacts of different livestock grazing intensities, we identified many vaguely defined and even subjective categories in the rangeland literature. They ranged from 'proper' to 'somewhat overgrazed' (Smith 1940), 'moderate' (Mwendera and Saleem 1997, Snyman 1998), 'heavy' and 'very heavy' (Dormaar et al. 1994, Mwendera and Saleem 1997). In addition, many studies reported vegetation cover to be affected by several grazing intensities without further specification (e.g. Rothauge et al. 2004, Allsopp et al. 2007). Because livestock grazing is the chief management pressure, we only considered well-defined grazing intensities (i.e. defined relative to the rangeland's carrying capacity and natural productivity (Fynn and O'Connor 2000, Stringham et al. 2003)). Hence, we therefore also ignored highly variable grazing intensities or regime transitions. We note that the grazing duration and its location plays an important role as most rangelands are highly adaptable to different grazing intensities (Perevolotsky and Seligman 1998). We also came across different approaches to restore (overgrazed) rangelands, which could be characterized by the timing and duration of discontinued grazing, and the restoration degree (e.g. Launchbaugh 1955, Dormaar and Willms 1998, Muñoz-Robles et al. 2011). This resulted in the management regimes *conservation* (any grazing disabled, long-term conservation), *restoration* (active restoration, including replanting and removing unwanted vegetation), *silvo-pasture* (planting or leaving trees to reduce erosion), *exotic tree plantation* (same purpose) and *abandoned rangeland* (disabling grazing to let vegetation recover). Several of these management regimes could shift into other regimes. Our typology was expedient in defining these transitions unambiguously.

5.6.2 Soil erosion and surface runoff

Our analysis of soil erosion and surface runoff per management regime is a test of our typology of rangeland management regimes. A challenge for such typologies is to determine whether generalised categories provide results that are precise and reliable enough to adequately mimic regional management effects on soil erosion and surface runoff (Stringham et al. 2003). We based our management regimes on common indicators found in the soil erosion and surface runoff literature. This shows that we retrieved sufficient data for most management regimes. However, an additional literature review was required for *overgrazed, abandoned rangelands and silvo-pasture*. Our comprehensive set of management indicators enabled us to easily link quantified information to a specific management regime via simple cross-tabulation (Table 5.2). Most data was retrieved for *high intensity grazed rangelands*, followed by *ungrazed natural* and *low intensity grazed rangelands*, *silvo-pasture* and *exotic plantations*. Although *conservation, abandoned degraded* and, to a lesser extent, *overgrazed rangelands* were frequently mentioned in the literature, very few quantitative assessments of either soil erosion or surface runoff could be found. The only information for *conservation rangelands* referred to underlying soil-related indicators (Launchbaugh 1955, Dormaar et al. 1994). We were not surprised to find limited information for *abandoned degraded rangelands*, because most studies focused on management regimes in transition to this regime (i.e. *high intensity grazed* or *overgrazed rangelands*). We note that *abandoned degraded rangelands* are frequently found in semi-arid rangelands, where woody encroachment and desertification are major problems (Muñoz-Robles et al. 2011, Puttick et al. 2011, Manjoro et al. 2012). Although many studies claim to study *overgrazed rangelands*, we found them to mostly focus on *high intensity grazed rangelands*. This can be attributed to the frequently used subjective definitions of overgrazing without truly assessing the ecological consequences and new equilibria that can evolve even after heavy grazing.

We established means of annual soil loss and surface runoff per management regime. These results should, however, be treated with caution, because no statistical analysis was conducted. We were able to compile data from multiple studies in different regions for all management regimes except for *conservation, restoration* and overgrazed *rangelands*. Studies on *silvo-pastures* included both natural and man-made *silvo-pastures* and more research is needed to retrieve differences between those two entirely different management systems (McIvor et al. 1995). Our further correlation analysis related underpinning indicators with soil loss and surface runoff. Although few correlations could be established, we found that slope is positively correlated to both soil loss and surface runoff. Moreover, soil organic matter was strongly negatively correlated with soil loss. Soil organic matter and slope are, therefore, useful indicators for quantifying soil erosion and surface runoff. Our preliminary analysis of soil erosion prevention and water flow regulation involved in transitions between management regimes should be considered a first step towards establishing robust relations between rangeland management and these ecosystem services. Future research should focus on compiling information for meta-analyses based on multiple sources per management regime transition.

Other reviews on erosion and surface runoff mainly focused on the impacts of broad land-use types, such as cropland, livestock grazing in general (mainly high) and different forms of agriculture. Although our review did not consider cropland, we were able to retrieve important ecological indicators for both soil erosion and surface runoff from cropland studies (Kosmas et al. 1997, Kisić et al. 2002, Fu et al. 2011). The effects of agricultural management on soil erosion have been better studied compared to livestock or rangeland conservation management. For instance useful management practices that reduce soil erosion include terracing, soil conservation management, mulching and alternative irrigation (Kosmas et al. 1997, Jégo et al. 2008). Although not all of these management practices apply to our typology, especially the ecological indicators could be used to develop our indicator overview and resulting indicator interaction diagram for soil erosion and surface runoff.

Our indicator overview and the interaction diagram offer an extensive overview of the key indicators for soil erosion and surface runoff. We listed recurring indicators that are relative simple to measure between different management regimes. Thus, the indicators can be used to consistently measure soil erosion and surface runoff and, in the longer term, their prevention. We note that studies combining soil erosion and surface runoff, used indicators more consistently than studies focusing on a single process. This is probably because disciplinary studies acknowledge the inherent complexities involved in establishing soil erosion and surface runoff. These studies measure many highly specific indicators, whereas our selected indicators simplify both processes. Runoff studies also position surface runoff as part of the entire hydrological cycle. Erosion studies do this for the soil balance using, for example, the Universal Soil Loss Equation (USLE, c.f. Fu et al. 2011). We did not include this equation's standardized factors, because we only used measured values. Runoff is also only a small part of the hydrological cycle and runoff alone is rarely the best indicator to study water flow regulation (Cerda et al. 1998, Bartley et al. 2006, De Moraes et al. 2006). The merits of reducing runoff are widely acknowledged in the literature and include improving productivity and downstream water quality (Narain et al. 1997, Snyman 1999, Fu et al. 2011).

We did not assess the effects of management regimes on other than soil erosion prevention and water flow regulation services. Despite the regimes' positive effects on soil erosion and surface runoff prevention, exotic tree plantations likely have negative effects on rangelands biodiversity and ecosystem services. Rangelands are biodiverse and provide many different ecosystem services, such as medicinal plants, raw materials, tourism, and carbon sequestration (Mortimore 2009). We noted, however, that none of the erosion or surface runoff studies assessed the consequences to or trade-offs with providing other ecosystem services. Even services that are directly related to livestock grazing (e.g. fodder, milk, meat and wool), affected by soil erosion and/or runoff (e.g. water purification and soil fertility) or rangeland restoration (e.g. tourism, habitat for large grazers and carbon storage), were rarely assessed. An apparent conclusion is that soil erosion and surface runoff have thus far only been studied in high detail by 'traditional' disciplinary soil-science and hydrology, whereas the merits of preventing these processes have been largely neglected.

We combined a typology of management regimes that is firmly grounded in the rangeland literature and based on a comprehensive set of quantitative indicators for soil erosion and surface runoff. Our findings inform decision makers on the consequences of livestock and rangeland conservation management decisions in semi-arid and sub-humid rangelands, and provides a first step in preventing further soil erosion and surface runoff by better managing rangelands.

ACKNOWLEDGEMENTS

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A research framework, three case studies, novel typologies of management regimes, extensive indicator lists and sets of information. This chapter synthesises the thesis' findings and discusses how the findings and methods can be integrated and used to inform decision makers and land managers. This thesis contributes to more informed management decisions regarding nature conservation, land-use intensification, converting nature to support intensive cultivation, land abandonment and restoration.

6 SYNTHESIS, DISCUSSION AND CONCLUSIONS

Humans have transformed and degraded the Earth's land cover at an alarming rate. This worldwide transformation and degradation suggest that land managers and decision makers have limited understanding of what is at stake in terms of environmental, social and economic costs, benefits and values (Barbier et al. 2008). Ecosystem services, which are the contributions of ecosystems to human wellbeing, have become an increasingly popular concept to demonstrate the consequences of land degradation and biodiversity loss (Norgaard 2010). Compiling and analysing empirical evidence on the effects of management on ecosystem services is required, as most management decisions are grounded in poorly verified assumptions (ICSU-UNESCO-UNU 2008, Carpenter et al. 2009). Informing decision makers is crucial, as decision making shapes human activities and behaviour, and thus determines important drivers of ecological degradation and change (Daily et al. 2009, Fürst et al. 2011). Therefore, this thesis aims to quantify the effects of management on ecosystem service provision. My findings should enable scientists to assess ecosystem services in a structured and consistent way, and should help decision makers and land managers to understand the effects of their management choices and activities.

My findings result from a systematic indicator-based analysis, guided by a framework for indicator selection (Figure 6.1) and a novel typology of management regimes. Three research questions guided the analysis: 1) What are key indicators for quantifying ecosystem service provision, 2) How can management effects on ecosystem services be quantified, and 3) How can management regimes be conceptualised to quantify their effects on ecosystem services? Three case studies were conducted to answer the research questions and, thereby, to refine indicator selection and the management regime typologies: in 'Het Groene Woud' (The Netherlands), mangrove systems in Java (Indonesia) and global semi-arid to dry sub-humid rangelands.

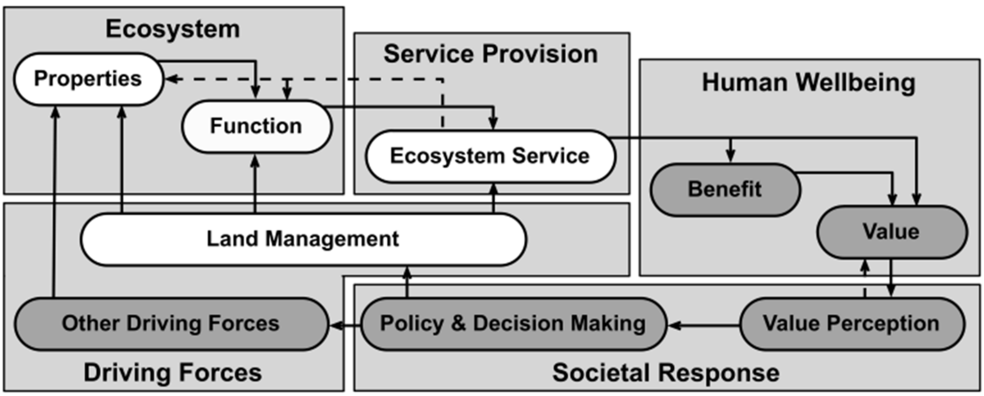

Figure 6.1: Research framework for indicator selection (based on Haines-Young and Potschin 2010; Kienast et al. 2009; De Groot et al. 2010a and Hein 2010). See Chapter 2 for a detailed description.

Table 6.1: Key ecosystem properties and state and performance indicators for ecosystem services analysed in this thesis. Ecosystem services classification based on TEEB (2010b) and De Groot et al. (2010a). Units are given between parentheses, if relevant. Ecosystem services with multiple state and performance indicators were analysed in different case studies. Ecosystem services with an asterisk (*) were not quantified in this thesis.

Ecosystem service	Ecosystem properties	State indicator	Performance indicator
Food: dairy Food: fish and shrimp	Vegetation quality, cow productivity, cow breed, soil quality Nursery service, trophic and physical subsidy, water quality	Number of cows (# ha^{-1}, LSU) Available fish and shrimp stock (kg ha^{-1} yr^{-1})	Dairy production (m^3 ha^{-1} yr^{-1}) Actual fish and shrimp harvest (kg ha^{-1} yr^{-1})
Raw materials: fodder Raw materials: NTFP	Land use type, productivity, landscape elements Species richness, tree age, density and productivity, flood pattern	Area producing fodder (ha), available fodder Available biomass (kg ha^{-1} yr^{-1})	Fodder production (kg ha^{-1} yr^{-1}) Harvested biomass (kg ha^{-1} yr^{-1})
Medicinal resources*	Species richness and diversity, forest size	Available medicinal resources (# or kg yr^{-1})	Harvested medicinal resources (# or kg yr^{-1})
Air quality regulation	Vegetation type, LIA, NPP, PM10 deposition speed, capture capacity	Captured PM10	Change in PM10 concentration (%, ppm)
Climate regulation	NPP, vegetation type, soil organic matter, land cover, soil moisture Soil depth, type and quality, tree diameter, age, productivity and size	Carbon sequestered (kg ha^{-1} yr^{-1}) Carbon storage (ha^{-1})	Change in CO_2 concentration (%, g m^{-3}) C storage relative to reference (kg ha^{-1} yr^{-1})
Moderation of extreme events	Extent, width of forest, species richness, structural diversity, tree age and productivity, water depth	Projected mangrove area (m^2), width of mangrove belt (m)	Wave height or storm surge reduction rate (m^{-1}), wave energy dissipation (%)
Water flow regulation	Soil porosity, evapotranspiration, vegetation type, soil moisture Canopy cover, slope, soil organic matter, bulk density, texture	Water storage capacity (m^3 ha^{-1}) Surface runoff (m^3 ha^{-1} yr^{-1}, % rainfall)	Change in ground water level (m, %) Runoff relative to reference (m^3 ha^{-1} yr^{-1}),
Waste treatment	Plants' nutrient needs, sediment stability, NPP, mangrove area	Potential N and P removal (kg ha^{-1} yr^{-1})	Actual N and P removal (kg ha^{-1} yr^{-1})
Erosion prevention	Canopy cover, slope, soil organic matter, bulk density, texture	Soil loss (kg ha^{-1} yr^{-1})	Soil loss relative to reference (kg ha^{-1} yr^{-1})
Pollination	Land cover, distance with nature, crops' pollinator dependence	Pollinator abundance, pollination rate	Change in crop yield (%)
Biological pest control	Insect species diversity, predator's habitat suitability, vegetation type	Natural predator abundance (# yr^{-1})	Pest insect predation (%)
Nursery service: fish and shrimp	Nutrient trapping, tidal mixing, turbidity, roots height, diversity of spatial and trophic niches, hydrological cycles	Contribution to fish stock (%), fraction of juveniles that mature into adults (%)	Harvest per mangrove area (kg ha^{-1} yr^{-1}), relative contribution to harvest
Maintenance of genetic diversity	Habitat requirement, landscape cohesion, vegetation, land cover, extent protected areas, species dispersal capacity, green elements	Habitat suitability	Number of species or individuals (# yr^{-1})
Aesthetic enjoyment*	Naturalness, land cover type	Number of houses near nature areas (#/ha)	Houses sold at green locations (#)
Nature-based recreation	Land cover type and diversity, walking tracks (m) Ecosystem health, flora and fauna with stated preference	Walking suitability (%) Potential number of recreants (# (ha^{-1}) yr^{-1})	Number of walking people Actual number of recreants (# (ha^{-1}) yr^{-1})
Information for cog-nitive development*	Land cover type with stated interest	Capacity of research facilities and visitor centres (# yr^{-1})	Number of excursions, number of visiting researchers (# yr^{-1})

The three research questions are answered and discussed in Sections 6.1 to 6.3, after which I reflect on an integrative approach for analysing management effects on ecosystem services (Section 6.4). This study's main findings are provided in Section 6.5.

6.1 KEY INDICATORS FOR QUANTIFYING ECOSYSTEM SERVICES

Quantifying ecosystem services involves assessing ecosystem properties that underpin service provision, and state and performance indicators that distinguish between potential and actual service provision, respectively. I developed a framework for systematic indicator selection (Figure 6.1) and tested, applied and further refined this framework in three different case studies. Table 6.1 summarises the key ecosystem properties and indicators for all ecosystem services that were studied in this thesis. The information resulted from extensive reviews into recurring indicators. I analysed sixteen out of the twenty-two ecosystem services of the TEEB-typology (De Groot et al. 2010a). Many ecosystem properties underpin multiple ecosystem services. This suggests potential harmful effects of altering ecosystem properties. Most state and performance indicators are compatible and, therefore, have similar units. However, some regulating, habitat and cultural services have different units that measured potential or actual provision. This mismatch reflects inconsistencies in the literature on defining the actual outcomes (i.e. 'performance') of regulating, habitat and cultural services (Villamagna et al. 2013, and Section 1.3). Differences between indicators also reflect differences in the case studies' data availability, spatial scale and ecological context. Nevertheless, the indicators in Table 6.1 are mostly compatible and quantifiable, and extensively summarise current knowledge on ecosystem service provision.

The selected indicators, as well as the selection process can be evaluated with the criteria that I introduced in Chapter 2. Criteria for the selection process include flexibility and consistency, whereas the indicators themselves need to be comprehensive, temporally and spatially explicit, quantifiable and portable. Moreover, indicators need to be understandable for multiple end-users and sensitive to management effects.

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). I tested the framework for selecting indicators to quantify, map and model ecosystem services in a case study in 'Het Groene Woud' (Chapters 2 and 3). This test required a rigid application of the framework. I was, however, able to advance the framework in the two other case studies, by assessing different (and overlapping) regulating and habitat services more in-depth. During these case studies I was able to select and analyse indicators that are largely consistent with the ones selected in the first case study (Table 6.1). The framework's flexibility is best illustrated by the 'indicator interaction diagram' that I introduce in Chapter 5 (Figure 6.2). The diagram follows the framework's main steps (properties, functions and services), but more precisely and appropriately illustrates the interactions between multiple indicators of these three steps. Chapter 5 and Figure 6.2 also suggest that following the framework's left-to-right direction or 'flow' and thereby arriving at quantifiable indicators is more important than rigidly filling in the framework's three boxes. The approach I introduced in Chapter 2 proved flexible and consistent as it allowed me to study very different case studies and refine it where necessary.

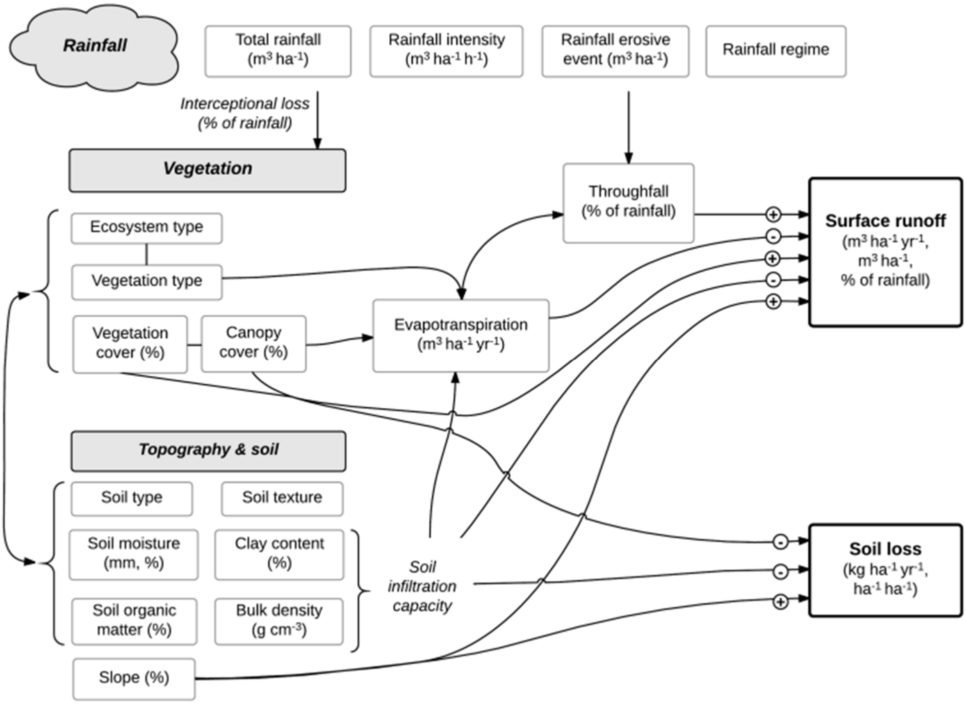

Figure 6.2: Interaction diagram for all the indicators approximating soil erosion and surface runoff. The indicators' units are listed between parentheses. See Section 5.4 for a detailed description.

The framework enabled selecting comprehensive indicators (c.f. Niemi and McDonald 2004), because I could consistently select multiple indicators per ecosystem service. Most state and performance indicators were spatially and temporally explicit, except for the proxies 'recreation suitability' and 'habitat suitability'. Quantifiable indicators ensure that information can be compared easily and objectively (Schomaker 1997, Layke et al. 2012). Although I often combined qualitative and quantitative indicators, especially for regulating services, most services could be quantified by scoring their provision or relating qualitative indicators to reference values. The indicators' portability (Riley 2000) was demonstrated by using similar indicators across different case studies and integrating the indicators' values using multiple data sources.

To effectively communicate findings of ecosystem service assessments, indicators should be understandable for multiple end-users (Niemeijer and de Groot 2008, UNEP-WCMC 2011). Some ecosystem properties in Table 6.1 are possibly too complex to easily communicate, but most state and performance indicators are sufficiently simplified and generic. Finally, Riley (2000) and De Groot et al. (2010b) suggest that indicators should provide information about causal relationships between land management and changes in ecosystem properties. My thesis comprehensively integrates key ecosystem properties underpinning all ecosystem services and explicitly relates these properties to service provision. Because management activities affect ecosystem properties directly, my framework should enable analysing management effects on ecosystem services consistently.

Table 6.2: Management indicators included in this thesis for analysing management effects on ecosystem services. Indicators with an asterisk (*) were not used for quantifying ecosystem services.

Case study 1: Groene Woud (Chapters 2 and 3)	Case study 2: Mangroves in Java, Indonesia (Chapter 4)	Case study 3: Semi-arid to dry sub-humid rangelands (Chapter 5)
Management activities	**Management activities**	**Management activities**
Planting, maintaining or clearing green landscape elements Organic dairy production Mechanised feeding and milking* Protecting areas and elements of cultural and natural importance * Constructing recreation facilities*	Recreational visits Fishing Timber harvesting NTFP harvesting Mangrove replanting	Exclosing or enclosing livestock Intercropping Soil treatment Vegetation removal Sowing grass Restoring natural vegetation Planting exotic trees
Management indicator	**Management indicator**	**Management indicator**
Land-use and land cover type Livestock stocking rate* Pesticide and nutrient use*	Aquaculture pond size Origin, density aquaculture stock Additional feed for aquaculture Fertilizer, pesticide, herbicide use	Livestock stocking rate Fertilizer, pesticide, herbicide use*

6.2 QUANTIFYING MANAGEMENT EFFECTS ON ECOSYSTEM SERVICES

Management involves activities that serve a land-use purpose and directly affect the land cover. My framework embeds management in a decision-making context and illustrates that management activities affect ecosystem properties, which leads to changes in ecosystem service provision (Chapter 2). For each case study, I reviewed relevant management activities (e.g. timber harvesting, clearing unwanted vegetation) and management indicators (e.g. aquaculture inputs, livestock density) that could be used to quantify management effects on relevant ecosystem services (Table 6.2). The management activities and indicators were selected independently from the ecosystem service indicators, to provide a realistic overview of each case study's management context. Only those ecosystem services can be quantified for which ecosystem properties are found with an empirically proven sensitivity to management effects. I note, in Chapter 2, that management could also affect ecosystem functions and even services, but this only relates to highly intensive, mechanised production systems, such as intensive cattle farming.

The indicators used in the first case study on 'Het Groene Woud' (Table 6.2) depend heavily on generic land use and land cover information, because the study was conducted on a landscape level and focused on mapping and modelling. Moreover, some ecosystem properties identified for 'Het Groene Woud' (Chapter 2 and 3), also relate to land cover. The analysis required interpolating national-scale datasets, which were only available in relation to land cover and land-use classes. I was able to look beyond simple land-use and land-cover variables by using more specific management activities and indicators in the other two case studies in Java and the rangelands (Table 6.2). These studies underline that effects from management activities, which were only assumed but not quantified in the first case study, could be assessed at landscape and biome levels.

Quantitative management indicators could only be used for analysing effects of specific land-use purposes, such as dairy farming and aquaculture. Generic indicators are desired, because they better explain general management effects on other land-use purposes than just their supported purpose. I therefore mostly used qualitative, generic and some binary indicators to compare management effects on ecosystem services. Chapters 3 and 4 also suggested several drivers of ecosystem service provision that relate strongly to management but are harder to control. These drivers include accessibility, distance between ecosystem service provider and user, climate and seasonality. I included these drivers while analysing some services, because management related strongly to their influence.

Although many studies acknowledge the importance of land management for ecosystem services provision and biodiversity (e.g. Carpenter et al. 2009, Perrings et al. 2010, Eppink et al. 2012), attempts to characterise and quantify management effects are variable and inconsistent (Chapter 1). I explicitly distinguish between land use, land cover and management, and I propose consistent indicators to quantify management effects on ecosystem services. However, given the generic nature of my study and the application for decision making, I required a flexible approach to consistently account for effects of multiple management activities on ecosystem services embedded in any decision-making context. My typology of management regimes (Chapters 4 and 5) integrates decision making, land-use purposes and management activities.

6.3 CONCEPTUALIZING MANAGEMENT REGIMES AND THEIR EFFECTS ON ECOSYSTEM SERVICES

In this thesis, I define management regimes as bundles of management activities that serve one or more land-use purposes. Regimes reflect differences in land-use intensity and represent clear land-use purposes or management choices. The need for a systematic typology to structure the analysis of land-use intensity, management and ecosystem services is frequently stated in the literature (De Groot et al. 2010b, Erb et al. 2013, Verburg et al. 2013b). Quantifying ecosystem services for different management regimes or states is considered a major research challenge (ICSU-UNESCO-UNU 2008, De Groot et al. 2010b), yet consistent typologies or conceptualisations are currently lacking. My typology distinguishes five broad categories that could be applied to any ecosystem: 'natural', 'low intensity use', 'high intensity use', 'converted' and 'abandoned'. However, quantifying effects of management regimes on ecosystem services requires these broad categories to be further specified, based on distinguishable policy regulations (if applicable) and land-use purposes, management activities and indicators, and ecological characteristics that result from these management activities. I developed management regimes for both a landscape (mangroves) and a regional level (rangelands), based on the same five broad categories of management regimes. These management regimes are shown in Table 6.3.

The mangrove management regimes are strongly embedded in the local policy, management and ecological contexts. These regimes' characteristics (i.e. the selected indicators and criteria) are highly specific and largely quantitative (see Table 6.2 for the management indicators). Because no specific policy regulations or ecological characteristics apply for global rangelands, the management regimes in dry rangelands were intentionally generic, which meant indicators needed to be precise yet generic (Table 6.2).

Management regimes are an important tool for quantifying effects of management on ecosystem services. Precise quantification of ecosystem services per management regime constitutes an important test of a management regime typology (Stringham et al. 2003). When applied on the landscape level, the management indicators and especially ecological characteristics can be quantified to analyse ecosystem services. In Chapter 4, I was able to relate characteristics, such as mangrove tree age, root length and species richness to the provision of seven ecosystem services, by relating them to ecosystem properties underpinning this provision. Such specific indicators fail to

Table 6.3: Management regimes used to analyse management effects on ecosystem services

Management regime	Mangroves in Java, Indonesia	Semi-arid to dry sub-humid rangelands
I. Natural	1. *Protection* of biodiversity, biophysical and ecological functions 2. *Conservation* of biodiversity, local traditions and recreation options	1. *Ungrazed* by livestock 2. *Conservation* of natural vegetation by disabling any grazing
II. Low intensity use	3. *Production* of timber and NTFP 4. *Unprotected*	3. *Low intensity grazed* 4. *Restoration* of grazed rangelands
III. High intensity use	5. *Plantation* to rehabilitate economic and ecological functions 6. *Silvo-fishery* to rehabilitate mangroves and aquaculture	5. *High intensity grazed* 6. *Silvo-pasture* intercropping trees with pasture or rangeland 7. *Overgrazed*
IV. Converted	7. *Eco-certified aquaculture* 8. *Extensive aquaculture* 9. *Semi-intensive aquaculture* 10. *Intensive aquaculture*	8. *Man-made pasture* 9. *Exotic tree plantation* to reverse land degradation and produce wood
V. Abandoned	11. *Abandoned aquaculture*	10. *Abandoned rangelands* 11. *Abandoned degraded rangelands*

account for regional differences and regional studies therefore require more generic indicators for both management activities and ecological characteristics (Chapter 5). The regimes' indicators were used as criteria to relate quantified ecosystem service provision to a specific regime. Both regimes were based on integrating findings from classical rangeland and ecological ecology and other disciplines with ecosystem services research and this was enabled by my indicator-based approach.

My extensive set of indicators enabled analysing ecosystem services for which no or limited quantitative information was available, let alone in relation to management. For example, I integrated quantitative (e.g. species richness, wave attenuation rate) and qualitative indicators (e.g. structural diversity, recreants' preference) to score recreation, nursery and coastal protection services, which had not been quantified before in relation to management. Knowledge on how underpinning ecosystem properties are affected per management regime can yield important information for data-scarce ecosystem services. Moreover, because I distinguish between land use, land cover, management and ecological characteristics, quantified ecosystem service information can be transferred from data-rich to data-scarce regions, based on multiple matching characteristics, rather than just ecosystem type.

My typology assesses management regimes as 'steady states', which means that transitions between regimes require considerable time and/or management effort. The typology presents decision makers with clear choices, but I note that further analysis is needed to establish more quantitative information per management regime as well as differences between different regimes. A meta-analysis as suggested in Chapter 5, could inform decision makers on the consequences of shifting from one regime to another. Moreover, by illustrating possible transitions between management regimes, I present decision makers a means to assess future pathways. My management regimes are a new addition to the literature and inform decision makers on the potential consequences of conservation, changing land-use intensities, conversion and land abandonment. Moreover, I show that management in support of land-use purposes can have unwanted negative or positive effects on untargeted ecosystem services.

6.4 TOWARDS AN INTEGRATIVE APPROACH TO QUANTIFY MANAGEMENT EFFECTS ON ECOSYSTEM SERVICES

In this section, I reflect on how the framework for indicator selection, interaction diagrams, management regime typologies and the study's quantitative information can be integrated in an integrative approach to quantify management effects on ecosystem services.

Relevant ecosystem services first need to be selected, before employing the framework for indicator selection. Selecting ecosystem services to study can be based on the key land uses, problems, opportunities in a given study area (Martín-López et al. 2014). I consulted key stakeholders and reviewed policy documents and grey literature to select ecosystem services for case studies in 'Het Groene Woud' and coastal Java. Note that lacking data did not discourage me from study poorly understood services, such as nursery and coastal protection. Local decision makers in Java appreciated any information I could communicate on those poorly understood services, because these services are considered crucial for the local peoples' livelihoods.

Indicators for quantifying ecosystem services can be selected in line with the stepwise framework, which enables selecting key ecosystem properties and state and performance indicators. Defining all indicators is crucial, even though data scarcity can lead to incomplete information for especially performance indicators. I could not always quantify actual service provision, because empirical evidence lacked. However, by analysing key ecosystem properties underpinning the potential provision of services, such as coastal protection, carbon sequestration and nursery, I could still inform decision makers on 'what matters' for ecosystem service provision. Information on ecosystem properties can be both quantitative and qualitative and can be integrated into 'indexes' that proxy state indicators, such as recreation suitability and potential coastal protection. The interrelation between indicators can be illustrated and communicated through the 'indicator interaction diagram', which follows the same 'flow' as the framework but is more specific. The diagram can also help identify better quantified indicators that underpin poorly quantified indicators. Evaluating the selected indicators should be done throughout the selection process. Key criteria for this selection process are flexibility and consistency and key criteria for indicators are comprehensiveness, sensitivity to management, portability and understandable to multiple end-users. Note that I also identified other factors than ecosystem properties that determine ecosystem service provision, such as distance between landscape element and cropland (pollination), and noise level and crowdedness (recreation). Such factors can be the 'missing link' when explaining service provision and most factors I identified are sensitive to management as well.

Defining management regimes can be done in parallel with selecting ecosystem service indicators. Assessing relevant policy regulations and descriptions is desirable to get a full grasp of the context of the study area. Policy regulations involve stimulating land-use purposes, restricting access or use and provide the general context for which activities are likely to occur. Policy regulations might not apply, due to lacking information or large spatial scale. However, land-use purposes can always be assumed or retrieved from the literature. When all land-use purposes are known, management activities and indicators can be selected that support these land-use purposes. Thus, a landscape or ecosystem can be divided in different management regimes that reflect different land-use intensities and purposes. These regimes can be divided into five broad categories of 'natural', 'low intensity use', 'high intensity use', 'converted' and 'abandoned', but quantifying ecosystem services requires more specific management regimes. The ‘management state’, or ecological characteristics that result from management activities, should ideally be established per management regime. Similar to the policy inventory, spatial scale or limited information might hinder this step. Information on ecological characteristics is desired but not essential, as long as ample information on management activities and indicators thereof exists. Information on land-use purposes, management activities and indicators, and ecological characteristics can be used as criteria for identifying which area corresponds with which management regime.

The final step involves relating the management regimes’ characteristics with quantified information on ecosystem service provision. Information on ecosystem services can be related to management regimes by using the criteria (i.e. cross-tabulation) or by matching ecological characteristics. In case of the latter, quantified ecological characteristics can become ecosystem service indicators. When no quantitative information is available for services, their provision can be assessed in relation to a reference situation (e.g. the ‘natural’ management regime). This is especially the case with key regulation services, such as pollination, coastal protection and nursery. The results (i.e. ecosystem service provision per management regime) are best presented to decision makers in a diagram that shows potential transitions between regimes (Figure 6.3). A table with results would present management regimes as static, linear phases, while a diagram shows possible transitions and illustrates the consequences of management decisions. How the results will be used, depends on decision makers’ preferences and criteria. Therefore, conducting a multi-criteria decision analysis might be desirable (Schwenk et al. 2012). With information on ecosystem service provision per management regime, decision makers can evaluate the consequences of their management choices and decisions.

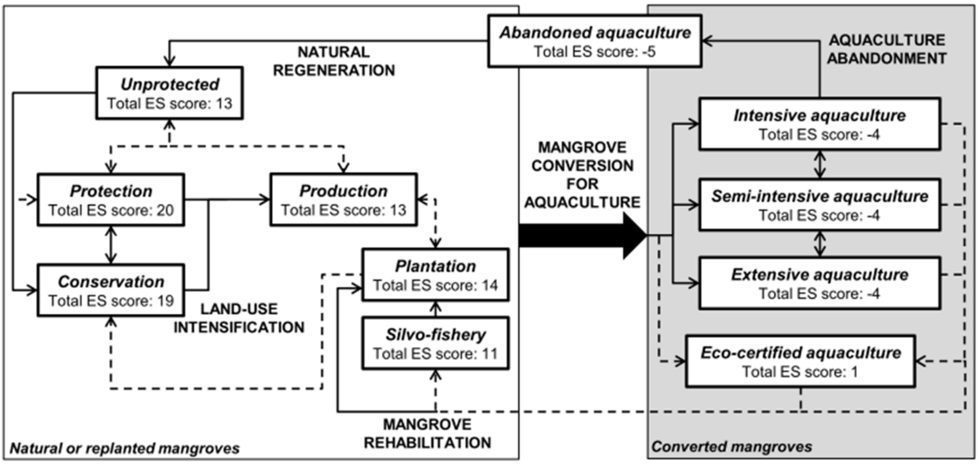

Figure 6.3: Diagram illustrating transitions between management regimes in coastal mangroves in Java, Indonesia. See Chapter 4 for more detailed information.

6.5 CONCLUSIONS

My study clearly shows the different effects of management in support of nature conservation, low and high intensity land use, converting nature into intensive cultivation (including cropping, ranging and aquaculture) and land abandonment on ecosystem services.

Natural management regimes aim to conserve nature and provide critical regulating and habitat services, and recreation opportunities. Providing food and raw materials is also high, but their use should be sustainable to not compromise continued high provision of regulating services and recreation. Balancing natural management regimes' capacity to provide regulating and provisioning services is crucial. Management regimes in support of low intensity or high intensity land use have a crucially different effect on ecosystem services. Intensive land use can refer to both ecosystem service production and nature restoration.

Low intensity use management regimes can provide most ecosystem services, but trade-offs between provisioning and regulating services occur locally. Management to support intensified food and raw materials production (i.e. high intensity use management regimes), generally has adverse effects on recreation opportunities and regulating services, such as carbon sequestration, erosion prevention, water flow regulation and coastal protection. My findings also underline that combining intensive production with active restoration and rehabilitation can partly mitigate these negative effects. Note, however, that most restoration sites are formerly converted and unsustainably used lands on which ecosystem service provision is recovering.

My study explicitly distinguishes converted lands that are now used for intensive food or fibre production. This high production generally occurs at the cost of all other ecosystem services and can even result in 'dis-services', such as carbon emissions, water pollution and high soil erosion. 'Exotic tree plantations' are the only exception I found that positively contribute to soil erosion prevention and water flow regulation. These plantations reduce soil erosion and surface runoff, but the negative effects on local biodiversity and ecosystem functioning are considerable. Finally, abandoned lands are a valuable option to restore nature, but, depending on their underlying actual ecosystem, these lands provide few ecosystem services if left unmanaged.

These findings are based on an integrative approach, which includes a framework for indicator selection, an indicator interaction diagram, a typology of management regimes and several approaches to integrate the results and illustrate transitions between management regimes. The approach integrates a comprehensive set of quantitative and qualitative indicators for quantifying management effects on ecosystem service provision. My approach can be used to inform decision makers on the consequences of management decisions regarding nature conservation, land-use intensification, converting nature to support intensive cultivation and restoring abandoned land. This thesis provides a first important step in preventing further land degradation and loss of ecosystem services by better managing the Earth's land.

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APPENDIX I

Additional information for Chapter 3

Indicators used for the spatial modelling of eight ecosystem services

Milk production

Ecosystem properties: Land-use type, area requirement per cow, percentage of milk cows, milk productivity per cow

Ecosystem function: Number of milk cows = F (land use type, area requirement per cow, percentage of milk cows) = grassland area * area need per cow * percentage of milk cows

Ecosystem service: Milk produced (L yr^{-1}) = F (number of milk cows, milk productivity of cows) = number of milk cows * milk productivity of cows

Fodder production

Ecosystem properties: Land-use type, maize productivity

Ecosystem function: Area of maize cultivation (ha)

Ecosystem service: Maize produced (kg yr^{-1}) = F (area of maize production, maize productivity) = area of maize cultivation * maize yield

Air quality regulation

Ecosystem properties: Land-use type, green elements cover, fine dust capture capacity of vegetation, fine dust emission, fine dust background concentration, relationship between vegetation and atmospheric fine dust concentration, vegetation cover (%)

Ecosystem function: Fine dust captured by vegetation (t ha^{-1}) = F (land use type, green elements cover, fine dust capture capacity of vegetation, fine dust background concentration) = land use & green elements cover * fine dust capture capacity at given background concentration

Ecosystem service: Decrease in atmospheric fine dust concentration (%) = F (% of vegetation cover, relationship between vegetation and atmospheric fine dust concentration, fine dust emission)

Climate regulation

Ecosystem properties: Land-use type, green elements cover, carbon emission factor, carbon equivalent

Ecosystem function: Carbon flux (ton ha^{-1} yr^{-1}) = F (land use type, green elements cover, carbon emission factor of land uses) = land use & green elements * carbon emission factor

Ecosystem service: Changes in atmospheric CO_2 concentration = F (carbon flux, carbon equivalent) = carbon flux * carbon equivalent

Pollination

Ecosystem properties: Land-use type, green elements cover, distance cropland to nature, fruit-set in relation to distance to nature, effective distance, pollinator-dependence of crops

Ecosystem function: Abundance of pollinators (%) = F (land use type, green elements cover, effective distance, fruit-set distance to nature curve)

Ecosystem service: Changes in crop yields (%) = F (abundance of pollinators, pollinator-dependent crops)

Biological control

Ecosystem properties: Land-use type, green elements cover, effective distance crops and nature, location of pest-influenced crops

Ecosystem function: Abundance of natural predators (%) = F (land use type, green elements cover, effective distance crops and nature)

Ecosystem service: Changes in pest predation (%) = F (abundance of natural predators, pest-influenced crops)

Lifecycle maintenance

Ecosystem properties: Land-use type, green elements cover, species dispersal capacity, habitat fragmentation, nature protection areas

Ecosystem function: Habitat suitability (%) = F (land use type, green elements cover, species dispersal capacity, fragmentation, nature protection areas)

Ecosystem service: Species occurrence = F (habitat suitability)

Opportunities for recreation

Ecosystem properties: Land-use type, land-use preference, noise level, proximity to green landscape elements, number of residents, percentage of residents that walk regularly

Ecosystem function: Walking suitability (%) = F (land-use preference, noise level, proximity to green landscape elements)

Ecosystem service: Number of walkers = F (walking suitability, % of residents that walk, number of residents)

APPENDIX II

Table S2.1: Mangrove species used for food, raw materials and medicinal use and speculation of in which management regime they occur, based on required ecological conditions. Sources for uses: Bandaranayake (1998), Iftekhar (2008), Kusmana (2010) and Saenger (2002).

Species	Food	Raw materials use	Medicinal use	Management regime*
Acrostichum Spp.	Fruit			All
Acrostichum aureum	Young plant, raw or cooked			All
Aegiceras cornoculatum			Fish poison (bark, seeds)	N, LIU, HIU, C
Avicennia Spp.	Fruits			N, LIU, HIU, C
Avicennia alba	Seeds (boiled)	Fodder (leaves)	Astringent (bark), contra-ceptive (resin), pox blisters (seeds)	N, LIU, HIU, C
Avicennia marina	Young leaves	Soap (ash)		N, LIU, HIU, C
Avicennia officinalis	Seeds (washed and boiled)			N, LIU, HIU, C
Bruguiera Spp.	Fruits	Plywood, chips, scaffolding, firewood, charcoal		N, LIU, HIU, C
Bruguiera gymnorzhiza	Flavouring fresh fish (bark)	Charcoal, firewood and tannin		N, LIU, HIU, C
Bruguiera sexangula	Young leaves, fruit embryo, root hairs	Incense (roots)	Skin tumours (leaves), eye wash (fruits)	N, LIU, HIU, C
Rhizophora Spp.	Fruits	Chips, scaffolding, charcoal, timber		N, LIU, HIU, C
Rhizophora mucronata		Charcoal, chips	Mosquito repellent (fruit juice, shoots)	N, LIU, HIU, C
Sonneratia Spp.	Fruits			N, LIU, HIU, C
Sonneratia caseolaris		Fodder (leaves), pectin (leaves)	Soften skin	N, LIU, HIU, C
Acanthus ilicifoLIUs			Snake bites, stop bleeding,	N, LIU, HIU
Ceriops tagal		Scaffolding, plywood, fishnet, incense, dye (bark)	General traditional remedies (bark)	N, LIU, HIU
Excoecaria agallocha			Fish poison (sap)	N, LIU, HIU
Heritiera littoralis		Planks, plywood	Fish poison (fruits 'juice)	N, LIU, HIU
Lumnitzera racemosa			Mouth ulcers (leaves)	N, LIU, HIU
Nipa fructicans	Drinks, alcohol (fermented sap), jelly (seeds), salt (leaves)	Roofing (leaves), hats, paper, baskets		N, LIU, HIU
Oncosperma tigillaria	Soft shoots, flowers (flavour rice)	Poles, stilts (houses) scaffolding		N, LIU, HIU
Xylocarpus moluccensis		Planks, decoration (wood)	Treat diarrhoea (bark), hair oil (fruit)	N, LIU

* N – Natural mangrove, LIU – Low intensity use, HIU – High intensity use, C – Converted

APPENDIX III

Additional information for Chapter 4

Ecosystem services provision of different mangrove management regimes

Natural mangroves

Ecosystem services provision for natural mangroves is generally high. Because different management activities take place and younger mangrove trees can be found in *conservation* forests, some differences exist between *protection* and *conservation* (Table S3.1).

Quantitative information on fish and shrimp harvests in Table S3.1 was based on studies by Gilbert and Janssen (1998), Kathiresan and Rajendran (2002), and Rönnbäck et al. (2007). High fish and shrimp harvests are possible due to the nursery service and optimal ecological characteristics. Shrimp harvests are likely to drop with decreasing mangrove age and species richness, resulting in highly variable potential shrimp harvest for *conservation*, due to the wide range of ecological characteristics (Kathiresan and Rajendran 2002). We note that quantified harvests in relation to mangroves are rare, and the numbers are merely indications for local harvests.

Natural mangroves have the largest above-ground biomass of all management regimes, due e.g. to their protection, species diversity and age (Walters 2005b, Rönnbäck et al. 2007). Estimations of above-ground biomass and sustainable harvest were retrieved from studies of *Rhizophora* spp dominated forests with similar age, d.b.h. and species richness as the management regimes (Gong and Ong 1990, Sukardjo and Yamada 1992, Ong 1993, Kauffman et al. 2011). *Protection* forests are likely to have higher biomass than *conservation* forests, due to their age (Table S3.1). Estimations for harvest potential were based on natural productivity of corresponding forests that are used for NTFP harvesting (Gong and Ong 1990, Ong 1993, Bosire et al. 2008).

Data on carbon storage could be retrieved based by matching d.b.h, species richness and mangrove age in the cited studies to the regimes' characteristics (Alongi et al. 2008, Alongi 2012). Studies on Indo-Pacific mangroves found both higher (maximum over 1000 ton C ha^{-1}) and lower outcomes, depending on the maximum measured soil depth (Donato et al. 2011, Kauffman et al. 2011). However, belowground carbon storage data of Indonesian mangrove are scarce. The scores in Table S3.1 are based on carbon storage (i.e. state indicator) and we assume it to be more variable and generally lower in *conservation* mangroves than in *protection* due to the fact that younger mangrove areas store considerably less carbon than more mature areas (Ong 1993).

Natural mangroves are highly suitable for wave attenuation, but storm surge reduction may differ. Because of the species diversity, age, density and length of roots, stems and branches, we assumed that the projected area and structural diversity of all natural mangroves is sufficiently large to attenuate smaller waves (Quartel et al. 2007, Tanaka 2008, Hashim et al. 2013). We furthermore assumed that the width of natural mangrove barriers generally exceeds the 500 m. required for attenuation of waves height by 50 – 99% (McIvor et al. 2012a). Estimations for storm surges have only been made for low intensity surges and with low certainty on the exact contribution of mangroves. McIvor et al. (2012b) reviewed that measured reductions in peak water levels range from of 5 to 50 cm per km of mangrove, which implies that a mangrove belt several km wide is re-

Table S3.1: Ecosystem service provision of *protection* and *conservation* management regimes in natural mangroves. Service provision is scored using circles (●/○). Closed circles (●) indicate high certainty, open circles (○) low.

Ecosystem Service	Protection	Conservation
Food (fish and shrimp)	High potential for fish and shrimp provision: estimations of 1-1.6 ton and 4 ton ha^{-1} of mangrove per year, respectively (○○)	High potential for fish provision, more variable for shrimp: estimations of 1-1.6 ton and 1-4 ton ha^{-1} of mangrove per year, respectively (○○)
Raw materials	Available biomass between 150 and 300 t ha^{-1}, max. sustainable yield about 12-24 t ha^{-1} yr^{-1}. (●●●)	Available biomass between 90 and 250 t ha^{-1}, max. sustainable yield about 10-17 t ha^{-1} yr^{-1}. (●●)
Carbon storage and sequestration	Carbon storage estimations of 430-700 ton C ha^{-1}. Sequestration data lack. (●●●)	Similar carbon storage as *protection*, but with higher variation. No sequestration data known. (●●●)
Coastal protection	Wave height reduced fully, storm surge protection dependent on width of mangrove area. (○○○)	Wave height reduced fully, storm surge protection dependent on width of mangrove area. (○○○)
Water purification (N & P removal)	Capable of removing aquaculture effluent, if sufficiently large area is available (2-21.4ha) (●●●)	Capable of removing aquaculture effluent, if sufficiently large area is available (●●●)
Nursery service	Optimal for both fish and crustaceans (○○○)	Optimal for both fish and crustaceans (○○○)
Nature-based recreation	High potential for recreation around the ecosystem, such as boating, plant and animal watching, and snorkelling and diving (○○○)	High potential for recreation around the ecosystem, such as boating, plant and animal watching, and snorkelling and diving. Infrastructure in place to support recreation (○○○)

quired to significantly reduce storm surge water levels. However, unlike wave attenuation, this reduction is non-linear over distance travelled (Zhang et al. 2012). Because of high species diversity, tree age, density of roots, stems and branches and related factors, we assume that storm surges can be reduced almost entirely by *protection* forests, provided that the greenbelt width is sufficient and the presence of rivers and open areas does not reduce the ability of mangroves to reduce peak water levels (Krauss et al. 2009, Zhang et al. 2012). *Conservation* forests could score somewhat lower due to impacts of recreation and lower tree maturity and species richness. Moreover, *protection* forests are managed for coastal protection and other physical functions and governed locally, i.e. by people who depend direct on coastal protection and are less likely to disturb the ecological integrity (Walters et al. 2005).

Studies of Robertson and Phillips (1995), Gautier (2002) and Primavera et al. (2007) into water purification (N and P) by mangroves focused on mangroves with a species diversity of 3 to 7 and an average age of at least 7 years. Both *protection* and *conservation* forests are capable of removing N and P due to their species diversity and age. Required mangrove areas are between 2.4-9 ha for N removal, and 3-21.4 ha for P removal (Robertson and Phillips 1995). The lower ranges apply to water purification of *low intensity aquaculture* effluents. The high number for P removal applies to intensive aquaculture effluent. We expect no differences between the two management regimes, because of the relatively low age of mangrove trees required for sufficient provision.

Although most studies on nursery service focus on natural mangroves, they do generally not provide ecological characteristics to enable matching with our management regimes. This is often due to the scale of the analysis and the fact that coarse spatial data were used. We assume that both *protection* and *conservation* mangroves are optimal nursery habitats for fish and crustaceans, due to low human disturbance, high age, species and structural diversity, and presence of tall roots found in natural mangroves contribute to this high potential. Moreover, natural mangrove areas are generally embedded in complex, integrated coastal and/or estuarine ecosystems, which implies that hydrological and hydrodynamic cycles are likely to be intact (Baran and Hambrey 1999, Rönnbäck 1999). Studies that quantified the contribution of mangroves to fish catch (see above) indicate high harvests compared to other coastal habitats and with increasing mangrove species diversity (e.g. Kathiresan and Rajendran 2002, Rönnbäck et al. 2003). Although most studies only provide harvest estimations, we consider this an indication of the relative contribution of the nursery service. Furthermore, harvests of shellfish were three times higher in natural mangroves compared to low and high intensity use areas (Kathiresan and Rajendran 2002).

Most nature-based recreation takes place in or is dependent on intact mangroves, but requires additional management. Information on recreation related to mangrove is scarce in the literature. We assume, however, that the occurrence of high biodiversity, opportunity to fish and watch rare plant and animal species combined with the supporting role for beach-recreation (snorkelling and diving) together make natural mangroves highly suitable for nature-based recreation (Knight et al. 1997, Satyanarayana et al. 2012). Recreation, and supporting infrastructure, is promoted in *conservation* forests, whereas recreation is allowed but not promoted in *protection* forests. However, activities such as snorkelling and diving depend highly on nearby mangrove ecosystems, regardless of their management (Mathieu et al. 2003). Moreover, recreational visits to fish and watch plant and animal species around *protection* forests would still be possible, making *protection* also important for nature-based recreation.

Low intensity use mangroves

Ecosystem services provision by low intensity use mangroves is generally lower and more variable when compared to natural mangroves due to timber and NTFP harvesting and the occurrence of less mature mangrove trees (Table S3.2).

Information on fish and shrimp harvests in Table S3.2 was based on studies by Gilbert and Janssen (1998), Kathiresan and Rajendran (2002) and Rönnbäck et al. (2007). Fish harvests are similar to that of *conservation* mangroves but shrimp harvests can be expected to drop. Kathiresan and Rajendran (2002) and Walton et al. (2007) suggest that these low shrimp harvests are caused by limited nursery service and nutrient availability for juvenile shrimp in younger mangroves with fewer species. However, evidence is inconclusive and studies have yielded highly diverse and even contradicting statements (Rönnbäck 1999, Kathiresan and Rajendran 2002). Actual harvests are assumed to be lower around *unprotected* mangroves due to limited accessibility, but estimations of *production* and *unprotected* mangroves could not be found.

Table S3.2: Ecosystem service provision of *production* and *unprotected* management regimes in low intensity use mangroves. Service provision is scored using circles (●/○). Closed circles (●) indicate high certainty, open circles (○) low.

Ecosystem Service	Production	Unprotected
Food: fish and shrimp	Potential for fish provision (0.6-1.5 ton ha^{-1} of mangrove per year estimated), but low shrimp provision (not quantified) (○)	Similar potential as *production* forests, but lower actual harvest due to limited accessibility (○)
Raw materials	Biomass stock between 90 - 200 t ha^{-1}, max. sustainable yield 9-12 t ha^{-1} yr^{-1}. (●●)	Biomass stock between 90 and 150 t ha^{-1}, harvest limited by accessibility. (○○)
Carbon sequestration	Aboveground C around 100 ton C ha^{-1}, but total carbon storage unknown (○○)	Limited management impacts but similar carbon storage as *production* (○○)
Coastal protection	Wave height reduced fully but risk of storm surges due to timber cutting. (○○)	Wave height reduced fully but risk of storm surges. Highly variable outcomes (○○)
Water purification (N & P removal)	Capable of removing aquaculture effluent, if area of 2-21.4ha is available, but lower suitability for P-removal. (●●)	Capable of removing aquaculture effluent, if area of 2-21.4ha is available. (○○○)
Nursery service	High potential for fish but lower for crustaceans (○○)	High potential for fish, lower for crustaceans (○○)
Nature-based recreation	Potential for fishing, hunting and observing traditional agriculture. (○○)	Some potential, but coordinated activities and facilities lacking (○)

Quantitative information on available biomass for timber and NTFP harvest could be retrieved from studies in highly impacted *production* forests in Indonesia and Thailand (reviewed by Sukardjo and Yamada 1992), and Malaysia (Gong and Ong 1990, Ong 1993) with similar age and d.b.h, dominated by *Rhizophora* spp. A study from Kenya (Bosire et al. 2008) provided information on maximum sustainable yields. Based on Gong and Ong (1990), we assume that available biomass in *protection* forests will be lower but still in the same order of magnitude as in *production*, and actual biomass yields are below maximum sustainable yield levels.

Above-ground carbon storage in forests that resemble Javanese *production* forests, in terms of age and management activities, are available from Malaysia (Ong 1993). However, total carbon storage of such systems could not be found, because effects of timber and intensive NTFP harvesting on soil carbon are unknown. Total carbon contents of forests with the age of *production forest* would be around 500 ton C ha^{-1} (Kauffman et al. 2013), but this amount will be lower due to management impacts, although e.g. Ong (1993) reported stable carbon storage of efficiently managed *production* forests. Aboveground biomass would typically be around 100 ton C ha^{-1}, which is considerably lower than similarly aged but not impacted mangrove areas (Ong 1993, Bosire et al. 2008). We provide the same score for *unprotected* as for *production* forests, because carbon storage of both is likely to be in the same order of magnitude and highly variable. Reliable carbon sequestration for low intensity use mangroves could not be found.

Low intensity use mangroves have potential for coastal protection, provided the width of the area is sufficient. We assess the overall potential for coastal protection as lower than that of natural mangroves because species and structural diversity, and root length of low intensity use mangroves are lower. Differences within low intensity use mangroves occur due to impacts of timber and NTFP harvesting in *production* forests as well as the unpredictable situation and high variability within *unprotected* mangroves. Wave attenuation can occur fully in both management regimes, because of the sufficiently high age, large roots and considerable diversity of mangrove species (Mazda et al. 2006, Quartel et al. 2007). Storm surges could have a high impact in *production forests* because of open areas as a result of timber extraction (Krauss et al. 2009). Potential for storm surge protection is difficult to indicate for *unprotected* forests, and highly variable. If undisturbed, their potential for coastal protection could be as high as young *conservation* forests, but this depends strongly on the occurrence of open patches.

We consider the potential of *production* forests for water purification of surrounding aquaculture ponds to be optimal, in view of their age, presence of roots, size, presence of saplings and young trees, and expected structure (Robertson and Phillips 1995, Li et al. 2008). Primavera et al. (2007) and Gautier (2002) showed that *production* forests' productivity could even benefit from filtering excess N, due to frequent timber and NTFP harvesting. However, harvesting could also lead to reduced ability to take up P, because of sediment disturbance (Li et al. 2008). This would be less likely to apply to unprotected mangrove areas, as we assume that timber harvest does not take place. The nursery service potential of low intensity use mangroves is expected to be lower and more variable as compared to natural mangroves, especially for shrimp, due to increased disturbance and decreased mangrove age, and species and structural diversity. Timber and NTFP harvesting in *production* mangroves will likely result in interrupted hydrodynamic cycles and hampered connectivity with other ecosystems (Rönnbäck 1999). However, literature suggests that even mangroves with lower species diversity can provide nursery service, because they are still embedded in coastal and / or estuarine ecosystems (Baran and Hambrey 1999, Rönnbäck 1999). Studies by e.g. Kathiresan and Rajendran (2002), Primavera (1998), and Rönnbäck et al. (1999) suggest that even with lower species diversity and age of mangroves, considerable amounts of fish but few crustaceans could be expected.

Nature-based recreation is allowed in *productio*n forests and is likely to occur because of the naturalness and opportunities to fish, watch birds etc. Compared to natural mangroves, there are few places of natural or spiritual interest in low intensity use mangroves, but especially fishing could be an interesting activity to promote. In addition, production forests might be an interesting location for hunting and observing traditional agriculture. Due to their remoteness and lack of facilities, unprotected mangroves can be assumed to have very low potential for recreation. Both regimes could, however, be important for recreation in nearby coastal areas due to their connectivity with other ecosystems (Mathieu et al. 2003).

High intensity use mangroves

Ecosystem service provision of high intensity use mangroves differs strongly per service and management regime (Table S3.3). Differences occur due to the role mangroves play for *silvo-fishery* ecosystem services and the limited age and species richness of *plantation* forests.

Table S3.3: Ecosystem service provision of *plantation* and *silvo-fishery* management regimes in high intensity use mangroves. Service provision is scored using circles (●/○). Closed circles (●) indicate high certainty, open circles (○) low.

Ecosystem Service	Plantation	Silvo-fishery
Food: fish and shrimp	Fish provision estimated at around 0.6 ton ha^{-1} of mangrove per year. Little shrimp provision (●)	Fish harvest estimated at 0.5 ton and shrimp harvest at 1-3 ton ha^{-1} of pond per yr. (●●)
Raw materials	Biomass stock between 50 and 116 ton ha^{-1}, max. sustainable yield about 6-11 ton ha^{-1}. (●●)	Biomass stock between 17 and 40 ton ha^{-1}, but lower yield due to nursery by mangroves. (○)
Carbon storage and sequestration	Carbon storage up to 93 ton C ha^{-1} and aboveground C sequestration around 10 ton C $ha^{-1}.year^{-1}$. Data for soil carbon unknown (●●)	Assuming 30-40% mangrove cover, a fraction of carbon storage by plantations can be reached. Sequestration unknown. (○)
Coastal protection	Wave height reduced but low storm surge reduction due to low projected area (○○)	Height of small waves reduced, but high impacts from high waves and storm surges. (○)
Water purification (N & P removal)	Capable of removing aquaculture effluent, if area of 2-21.4ha is available (●●●)	Capable of removing N and P from own effluent, but required mangrove area unknown (○○)
Nursery service	Potential nursery service for fish, but variable and largely negligible for crustaceans (○○)	High nursery service within ponds, mainly for shrimp. Poorly quantified (○○)
Nature-based recreation	Potential for recreation, such as fishing, participating in replanting and wildlife spotting (○○)	Potential for recreation, such as fishing, participating in replanting and wildlife spotting (○○)

Fish and shrimp provision by high intensity mangroves is measured differently per management regime; in *plantation* per ha of mangrove and in *silvo-fishery* per ha of pond. Studies by Gilbert and Janssen (1998), Kathiresan and Rajendran (2002) and Rönnbäck et al. (1999) suggest that considerable fish provision but limited shrimp provision can be expected in mangrove *plantations* that match characteristics of our management regimes, in terms of species diversity, age and size. Shrimp populations are likely suffer even more than in *production* forests (Kathiresan and Rajendran 2002, Walton et al. 2007). Estimations of shrimp yields for *silvo-fishery* were based on Gilbert and Janssen (1998), Bengen (2003) and Kusmana et al. (2008). Shrimp yields are attributed to the nursery function of silvo-fisheries, food availability, pollutant removal etc. (Kusmana et al. 2008). Scores as provided in Table S3.3 are relative to maximum aquaculture harvests, although we note that the two regimes are difficult to compare. Fish is only provided if the *silvo-fishery* ponds are loaded with mixed stocks (Gilbert and Janssen 1998).

Estimations of biomass levels are based on Sukardjo and Yamada (1992) and Bosire et al. (2008) but are variable due to their dependence on management, recruitment, location and environmental factors. Maximum sustainable yield levels were based on the same sources as well as Ong (1993). Biomass data from *silvo-fisheries* are not available, but Gilbert and Janssen (1998) propose that *silvo-fisheries* generate up to half the amount of raw materials (mainly timber / construction) compared to similarly sized *plantations*, as a result of high nutrient input from effluents. Assuming mangrove cover of 30-40% (section 4.4.4), we estimate *silvo-fisheries* to have that proportion of the biomass of *plantations*. Actual yields will be much lower than that because of reduced accessibility and the recognised role of the mangroves have for the nursery service.

Aboveground carbon storage and sequestration data are available for *plantations* because they have been monitored since the moment of replanting. Sukardjo and Yamada (1992) monitored 7-year-old mangrove *plantations* with species richness, height and d.b.h. ranges similar to our regime (Table S3.3). Aboveground biomass in *plantations* in Malaysia are somewhat lower, but in the same order of magnitude (Ong 1993). Both estimations should be considered as relatively high, as the intensity of NTFP harvesting was not specified in the above-mentioned examples. Bosire et al. (2008) provided biomass accumulation data for *Rhizophora*-dominated *plantations*. *Silvo-fishery* carbon storage and sequestration is assumed 30 to 40% of that of *plantations*, but reliable estimations of soil carbon storage or sequestration by both regimes are not available.

Coastal protection by mangrove *plantations* is higher than by *silvo-fisheries* due to differences in their projected area; *plantations* have fewer openings because of the absence of ponds and gutters (Zhang et al. 2012). Wave attenuation by *plantations* can be provided due to the presence of roots and high enough trees, but the lack of adult mangroves, and structural and species diversity is considered to be detrimental for storm surge protection (Mazda et al. 2006, Quartel et al. 2007). Literature on coastal protection by *silvo-fisheries* is lacking, but we assume they have some potential for small wave attenuation due to their ecological characteristics. However, impacts of higher waves and storm surges are likely to increase due to ponds and gutters reducing the projected area and dykes increasing the wave and surge height (Krauss et al. 2009, Winterwerp et al. 2013).

Water purification by high intensity use mangroves differs per management regime. Mangrove *plantations* are well-suited because they possess the desired age, size, roots, species diversity as mentioned by Gautier (2002) and Primavera et al. (2007). This rationale is further strengthened by the fact that NTFP is extracted regularly (thus removing nutrients) and replanting takes place (enabling new nutrient uptake) (Primavera et al. 2007). Mangroves in *silvo-fisheries* could purify their own effluent, which will contain less N and P compared to intensive aquaculture. This assumption is based on *silvo-fisheries*' ecological characteristics and efficient water management taking place (Bengen 2003). Both management regimes, however, require sufficient mangrove area for water purification, which complicates estimations for *silvo-fisheries* especially.

Plantations can provide the nursery service, although results vary considerably and are mainly limited to observed number of (fish) species and not actual recruitment (e.g. Primavera 1998, Rönnbäck et al. 1999, Bosire et al. 2008). As described above, fish can still be found around *plantations*, albeit considerably fewer than in natural and low intensity use mangroves, but shrimp prevalence is close to zero (Primavera 1998). Water inlets, protection by dykes and mangroves in *silvo-fisheries* contribute to high nutrient availability, refuge, shelter and clean water (Sofiawan 2000, Rönnbäck 2002). The exact nursery contribution of *silvo-fisheries* has, however, never been quantified due to methodical difficulties and the complexity related to seed input and other management factors (Sofiawan 2000, Bengen 2003). However, *silvo-fisheries* are able to provide large amounts of fish and shrimp without additional feeding, which itself is already evidence of the usefulness of having mangroves planted inside and around the ponds.

High intensity use mangroves have potential for being of educational and recreational interest, although only sporadic observations confirm this. Attractive features that recreants could be drawn to include fishing, taking part in rehabilitation, spotting birds and other wildlife. However, since many mangrove plantations also serve the purpose of coastal barriers, not all areas would be equally suitable for recreation.

Mangroves converted for aquaculture

Only artificial food provision by aquaculture is very high, but actual ecosystem service provision is low or even negative ('disservice') as a result of aquaculture management inputs. Table S3.4 provides a short overview of the ecosystem services and disservices mangroves converted to aquaculture.

Estimations of fish and shrimp harvests for all aquaculture options are based on Gautier (2002), Gilbert and Janssen (1998), Rönnbäck (2002) and Primavera et al. (2007). Increasing harvests in Table S3.4 result of increasingly high inputs of seeds, food and fertilizer, and as such it can be argued that fish and shrimp production by such management would not be called ecosystem service provision. Except for *extensive aquaculture*, such high stocking rates (see section 4.4.4) could not survive without use of fertilizers, pesticides etc. Harvests of *eco-certified aquaculture* are not available and have been based on the assumption that inputs are similar to *intensive aquaculture*.

We assume that only in extensive and eco-certified aquaculture some raw materials will be harvested. Raw material use of mangroves near eco-certified aquaculture is not allowed, so we assume that only some deadwood and leaves will actually be used. However, biomass can be considerable (Sukardjo and Yamada 1992, Ong 1993). Raw materials harvested around extensive ponds will be limited to leaves (fodder, fertilizer) and fuelwood of pruned branches. Only a fraction of the biomass stocks of mangroves in extensive ponds are used. Often, farmers will cut mangroves on regular basis to facilitate pond renovation or additional raw material harvest. No data exists of mangrove use around aquaculture ponds.

Carbon emissions resulting from mangrove conversion into aquaculture will not be described here as we only describe regime in which mangroves have already been converted, and consider effects by management activities that take place within the management regime. A recent study by Kauffman et al. (2013) provides the sole data on carbon storage of soils of abandoned aquaculture ponds. They found significantly lower average carbon storage in abandoned shrimp ponds, namely 95 ton C ha^{-1}, compared to the average of 853 ton C ha^{-1} in all other mangrove areas, indicating the loss of carbon after mangrove conversion to aquaculture. However, we assume that little carbon storage occurs in aquaculture ponds. Ong (1993) even indicate carbon loss from oxidizing sediments can be up to 75 ton C $ha^{-1}.yr^{-1}$ in converted mangrove areas. In addition, shrimp and fish farmers in Java tend to drain their ponds at least twice a year and dig up soil to fortify their dykes and other structures. In combination, these activities will likely lead to further loss of soil carbon (Mcleod et al. 2011). We therefore assume that *extensive*, *semi-intensive*, and *intensive aquaculture* are net emitters of carbon. Carbon emission from sediments is difficult to compare to carbon sequestration of other management regimes, as this is the only instance in which we quantify sequestration or emission in sediment rather than in aboveground biomass. The few mangroves that are found around the ponds are frequently pruned and/or replaced, and do not contribute to carbon

Table S3.4: Ecosystem service provision in mangroves converted for aquaculture. Service provision is scored; circles (●/○) indicate positive, diamonds (◆/◇) negative service provision. Close figures (●/◆) indicate high certainty, open symbols (○/◇) low.

Ecosystem Service	Eco-certified aquaculture	Extensive aquaculture	Semi-intensive aquaculture	Intensive aquaculture
Food: artificial fish and shrimp production *	Shrimp 1-4 t, fish 3-6 ton ha^{-1} of pond yr^{-1}. (○○○)	Shrimp 1 t, fish 1-2 t ha^{-1} of pond yr^{-1}. (●●)	Shrimp 2-6 t, fish 2-3 t ha^{-1} of pond yr^{-1}. (●●)	Shrimp 7-15 t, fish 4-5 t ha^{-1} of pond yr^{-1}. (●●●)
Raw materials	Around 50-90 t ha^{-1} biomass available, but low harvest. (○)	Up to 50 ton ha^{-1} biomass available, but low harvest. (○)	Little biomass available and low harvest. (-)	Little biomass available and low harvest (-)
Carbon storage and sequestration	Carbon storage by replanting nullified by sediment management (◇)	Carbon emission due to drainage and use of sediment. (◇◇)	Carbon emission due to drainage and use of sediment. (◇◇)	Carbon emission due to drainage and use of sediment. (◇◇)
Coastal protection	Wave height increased due to dykes but attenuated by replanted mangroves. Risk of storm surges. (◇)	Wave height increased due to reflection on earthen dykes, little wave attenuation by mangroves. Risk of storm surges (◇◇)	Wave height increased due to reflection on earthen dykes, little wave attenuation by mangroves. Risk of storm surges (◇◇)	Wave height increased due to reflection on earthen dykes, little wave attenuation by mangroves. Risk of storm surges (◇◇)
Water purification (N & P removal)	Emission around 130-200 kg N, 40 kg P ha^{-1}.yr^{-1} in discharge water. Role of mangroves unknown. (◇◇◇)	Emission up to 130 kg N, 40 kg P ha^{-1}.yr^{-1} in discharge water. Limited role of mangroves (◇◇)	Emission ffl 130-180 kg N, 40 kg P ha^{-1}.yr^{-1} in discharge water. No influence of mangroves (◇◇)	Emission up to 200 kg N, 40 kg P ha^{-1}.yr^{-1} in discharge water. No influence of mangroves (◆◆◆)
Nursery service	Potential effect of mangroves nullified by high nutrient inputs (-)	None, due to high stocking rate, pesticide use etc. (-)	None, due to high inputs of nutrient, pesticides etc. (-)	None, due to high inputs of nutrient, pesticides etc. (-)
Nature-based recreation	Some potential for educational activities (○)	None (-)	None (-)	None (-)

* Due to high input and influence of other management factors, the production of shrimp and fish can hardly be defined as an ecosystem service. The harvest numbers are relevant though, because they allow for a comparison with other management regimes.

storage or sequestration. *Eco-certified aquaculture*, on the other hand, could contribute to carbon storage through required mangrove rehabilitation and protection. In-situ rehabilitation and replanting is required and mangroves within the system are of considerable age and size. This could indicate a potential of eco-certification to store carbon. However, a lot would depend on how sediments are treated; if sediment management is similar to that of other aquaculture options very little net gain will be observed.

Aquaculture ponds are surrounded by earthen or concrete dykes, some of which with mangroves planted on them. Although these dykes may offer some protection against wave impacts, they do not buffer against high waves or storm surges. On the contrary, Winterwerp et al. (2013) showed that waves and storm surges reflect on fixed structures thus increasing in height and taking sediment away as well. Only the mangroves of *extensive* and *eco-certified aquaculture* options could reduce some of the height of low waves as a result of replanted or remaining mangrove stretches of sufficient age. However, the roots are extremely small, and the trees are generally uniform in size and pruned, which reduces the already minimal projected area of the mangroves. Finally, although it is argued that *eco-certified aquaculture* could contribute to coastal protection and mangrove rehabilitation in general, we consider this to be an ex-situ measure as it generally involves establishing a greenbelt.

All aquaculture options should be considered emission sources of N and P in effluent water, due to high inputs of feed, fertilizer and fish/shrimp stocks, and lack of mangroves inside ponds. Mangroves planted within *eco-certification* schemes are generally not planted inside or in connection to ponds. In addition, mangrove roots barely touching the water and are therefore unlikely to contribute to water purification. Differences in aquaculture emissions are mainly based on Robertson and Phillips (1995) and confirmed by Gautier (2002), Primavera et al. (2007) and others. Based on matching data for stocking density, pond size, feed input, fertilizer, *intensive aquaculture* (shrimp) ponds will emit 200 kg N and 40 kg of P $ha^{-1}.yr^{-1}$ in effluent water. Feed input is the major source of N input, accounting for up to 90%. Estimations of emissions by other aquaculture are less certain, because data could not always be linked to matching aquaculture indicators such as mentioned above. We interpolated data by Robertson and Phillips (1995), based on stocking density and feed used, but note that artificial and natural feed contribute differently to effluent concentrations. The amount of P in the effluent does not change much per management regime because the majority of P settles in the sediment of the ponds (Robertson and Phillips 1995). Emissions from *eco-certified aquaculture* are also based on that of *intensive aquaculture*, but natural feed will lead to slightly lower emissions. No ecological characteristics could be related to water purification, as the role of mangroves is negligible in most aquaculture options or not yet explored in *eco-certified aquaculture*.

None of the aquaculture options provide nursery service. The fact that *extensive aquaculture* requires no additional feeding can be attributed to the nutritional value of the inflowing water, and not to mangroves. The likelihood for potential nursery decreases with increasing intensity of aquaculture, due to usage of pesticides, fertilizer, nutrients, and drainage of ponds.

Due to the absence of natural features that could be of interest to recreants we assume no recreation service is provided in aquaculture sites. Similar to *silvo-fisheries*, *eco-certified* shrimp ponds have the potential to become recreation sites, because of their role in mangrove rehabilitation (education and ecological interest) and shrimp aquaculture.

Abandoned aquaculture

Abandoned aquaculture sites are characterised by polluted soils, degraded biodiversity and remaining concrete or metal structures (Stevenson 1997). Although some of the abandoned sites are being used for alternative purposes such as housing or storage, we can assume that provision of fish and shrimp, and raw materials is low to non-existent. Nursery service cannot be provide either, as is depend on species diversity, age and biomass of mangrove trees. In addition, nature-based recreation is also unlikely to take place, because of the lack of actual nature, and polluted water and soils. However, carbon sequestration, coastal protection and water purification require some explanation, as the biggest problem related to abandoned ponds lies in their sediments and remaining structures.

A recent study by Kauffman et al. (2013) provides the sole data on carbon storage of soils of abandoned shrimp ponds. The authors found significantly lower average carbon storage in abandoned shrimp ponds compared to the average in all other mangrove areas (95 ton C ha^{-1} vs. 853 ton C ha^{-1}). Much would depend on what happens to the remaining sediment. It is unlikely to be dug up and reused as is the case in active aquaculture ponds, but will still continue to oxidise and leach carbon due to drainage (Ong 1993, Kauffman et al. 2013). Despite the fact that soils of abandoned ponds capture carbon, they must not be considered as contributing to sequestration, because of the lack of vegetation (i.e. production) and decreased drainage conditions. Therefore, abandoned ponds get a negative score, albeit that their emission is unlikely to be higher than that of active aquaculture.

Coastal protection is likely to be worsened by the remaining structures and lack of vegetation of *abandoned aquaculture*. Incoming waves and storm surges are expected to gain in height and level due to remaining (even collapsed) structures (Winterwerp et al. 2013). Finally, sediments of *abandoned aquaculture* ponds contain high amounts of P and N. The majority of the effluents' N and P is stored in sediments, so abandoned ponds can be seen as sources of continuous pollution of excess nutrients (Robertson and Phillips 1995).

APPENDIX IV

Additional information for Chapter 5

Indicators for quantifying soil erosion and surface runoff

Table S4.1: Percentage of data entries that mention indicators of soil erosion (A) and surface runoff (B). The number of reviewed studies that mention the indicator is given between parentheses. We distinguish between studies on just erosion, just surface runoff and on erosion and surface runoff combined.

A

Indicator for erosion	Erosion only (n= 14; 141 entries)	Erosion and surface runoff (n= 17; 205 entries)
Yearly rainfall (m^3 ha^{-1} yr^{-1})	34 (8)	92 (16)
Rainfall regime	55 (10)	73 (12)
Rainfall erosive event (m^3 ha^{-1})	43 (1)	35 (5)
Rainfall intensity (m^3 ha^{-1} h^{-1})	11 (1)	39 (7)
Slope (%)	69 (5)	96 (16)
Soil type	36 (8)	70 (14)
Soil texture	73 (8)	86 (13)
Soil bulk density (g cm^{-3})	60 (6)	47 (7)
Clay content (%)	55 (4)	55 (9)
SOM (%)	14 (3)	32 (8)
Vegetation cover (%)	66 (6)	76 (13)
Canopy cover (%)	4 (5)	16 (5)
Annual soil loss (kg ha^{-1} yr^{-1})	6 (2)	45 (11)
Soil loss (kg ha^{-1})	55 (3)	42 (6)

B

Indicator for surface runoff	Runoff only (n= 11; 73 entries)	Erosion and surface runoff (n= 17; 205 entries)
Yearly rainfall (m^3 ha^{-1} yr^{-1})	96 (10)	92 (16)
Total rainfall (m^3 ha^{-1})	33 (3)	73 (12)
Slope (%)	58 (4)	96 (16)
Soil type	86 (8)	70 (14)
Soil texture	76 (7)	86 (13)
Soil bulk density (g cm^{-3})	13 (3)	47 (7)
Clay content (%)	3 (1)	55 (9)
Vegetation cover (%)	25 (4)	76 (13)
Canopy cover (%)	42 (4)	16 (5)
Soil moisture (mm/%)	81 (7)	-
Throughfall (% of rainfall)	31 (3)	-
Evapotranspiration (m^3 ha^{-1} yr^{-1})	26 (5)	-
Annual surface runoff (m^3 ha^{-1} yr^{-1})	14 (3)	41 (8)
Surface runoff (m^3 ha^{-1})	-	27 (7)
Surface runoff coefficient (% of rainfall)	-	42 (6)

Table S4.2: Mean values ($\bar{x}$) of indicators underpinning soil erosion for each management regime. The standard error (SE) is given after each mean, followed by the number of data entries (n).

		Rainfall erosive event($m^3\ ha^{-1}$)		Rainfall intensity ($m^3\ ha^{-1}\ h^{-1}$)		Slope (%)		Soil bulk density ($g\ cm^{-3}$)		Clay content (%)		SOM (%)		Vegetation cover (%)		Canopy cover (%)	
	Management Regime	$\bar{x}$	SE (n)	$\bar{x}$	SE (n)	$\bar{x}$	SE (n)	$\bar{x}$	SE (n)	$\bar{x}$	SE (n)	$\bar{x}$	SE (n)	$\bar{x}$	SE (n)	$\bar{x}$	SE (n)
I.	Natural ungrazed	353	65(6)	336	60(15)	18	2(27)	1.74	0(1)	8	0(1)	6.7	0.1(5)	61	1(13)	71	3(6)
II.	Low intensity grazed	1303	1702(6)	469	82(10)	5	0.6(15)	1.33	0.05(11)	18	2(15)	3.2	0.4(12)	16	5(17)	32	6(13)
	Restoration	*no data*		135	13(16)	20	0(16)	1.55	0.10(2)	7	0(2)	5.5	0.5(2)	27	3(16)	*no data*	
III.	High intensity grazed	10227	387(116)	737	77(23)	12	0.6(147)	1.21	0.01(119)	33	1(129)	3.5	0.2(15)	69	2(134)	5	0(6)
	Overgrazed	3500	0(2)	*no data*		4	2(2)	*no data*		35	4(2)	2.1	0.4(2)	*no data*		*no data*	
	Silvo-pasture	*no data*		*no data*		5	0.6(14)	1.50	0(2)	8	0(2)	*no data*		62	9(7)	40	7(4)
IV.	Man-made pasture	*no data*		*no data*		4	0.2(7)	*no data*		*no data*		*no data*		56	9(3)	*no data*	
	Tree plantation	*no data*		*no data*		4	0(15)	*no data*		*no data*		*no data*		51	7(6)	88	6(6)
V.	Abandoned	3500	0(2)	*no data*		4	2(2)	*no data*		35	4(2)	2.1	0.4(2)	*no data*		*no data*	
	Abandoned degraded	*no data*		350	0(7)	*no data*		*no data*		*no data*		*no data*		*no data*		*no data*	

Table S4.3: Mean values ($\bar{x}$) of indicators underpinning surface runoff erosion for each management regime. The standard error (SE) is given after each mean, followed by the number of data entries (n).

		Slope (%)		Soil bulk density(g cm^{-3})		Clay content (%)		Vegetation cover (%)		Canopy cover (%)		Soil moisture (%)		Throughfall (%)	
	Management Regime	$\bar{x}$	SE (n)	$\bar{x}$	SE (n)	$\bar{x}$	SE (n)	$\bar{x}$	SE (n)	$\bar{x}$	SE (n)	$\bar{x}$	SE (n)	$\bar{x}$	SE (n)
I.	Natural ungrazed	19	1.7(30)	1.74	0(1)	8	0(1)	52	6(19)	71	3.3(6)	31	3(3)	40	8(3)
	Conservation	27	0(18)	1.10	0.04(18)	*no data*		*no data*		76	0(18)	29	3(20)	55	5(17)
II.	Low intensity grazed	5	4.0(16)	1.33	0.04(13)	18	2(16)	13	4(22)	29	4(20)	24	7(6)	*no data*	
	Restoration	20	0(16)	1.55	0.1(2)	7	0(2)	27	3(16)	*no data*		*no data*		*no data*	
III.	High intensity grazed	12	1.0(92)	1.23	0.01(68)	31	1(76)	61	4(88)	5	0(10)	22	4(8)	51	0(1)
	Overgrazed	24	1.7(19)	*no data*		35	4(2)	*no data*		*no data*		20	2(18)	*no data*	
	Silvo-pasture	5	0.6(14)	1.50	0(2)	7.5	0(2)	62	9(7)	40	7(4)	*no data*		*no data*	
IV.	Man-made pasture	4	0.2(7)	1.24	0(1)	*no data*		45	13(4)	*no data*		7.3	0(1)	*no data*	
	Tree plantation	10	4.2(18)	1.10	0.05(2)	16	1(2)	54	6(8)	88	6(6)	*no data*		*no data*	
V.	Abandoned	49	5.6(25)	1.18	0.03(22)	39	2(23)	70	3(24)	*no data*		24	11(2)	15	0(1)

NEDERLANDSTALIGE SAMENVATTING

Mensen hebben een groot deel van het aardoppervlak veranderd om in de groeiende vraag naar leefruimte en voedsel, drinkwater en andere natuurlijke hulpbronnen te voorzien. Mijn proefschrift onderzoekt de effecten van landbeheer op ecosysteemdiensten. Landbeheer betreft de menselijke activiteiten die landgebruik ondersteunen. Enkele voorbeelden zijn het toepassen van bestrijdingsmiddelen of irrigatie om de voedselproductie te verhogen, de aanleg van infrastructuur om ecotoerisme te bevorderen en het verwijderen van ongewenste plantensoorten om inheemse biodiversiteit te beschermen. Ecosysteemdiensten zijn de bijdragen van ecosystemen aan het menselijk welzijn. Ze zijn uitgegroeid tot een populair concept om de gevolgen aan te tonen van biodiversiteitsverlies en landdegradatie. Echter, de wereldwijde aantasting en transformatie van ecosystemen ten behoeve van monofunctioneel landgebruik suggereren dat beheerders en beleidsmakers beperkt inzicht hebben in de economische en sociale gevolgen van landgebruik. Inzicht in de effecten van beheer op ecosysteemdiensten draagt bij aan beter begrip van de gevolgen van beleidsvorming en landgebruiksbeslissingen. Huidig beheer is grotendeels gebaseerd op slecht geverifieerde veronderstellingen. Derhalve heeft mijn proefschrift tot doel om de effecten van beheer op de levering van ecosysteemdiensten te kwantificeren.

Om dit onderzoeksdoel te bereiken, heb ik een op indicatoren gestoelde methodologie ontwikkeld. Een nieuw ontwikkeld raamwerk voor systematische indicatorselectie staat centraal in de studie (Hoofdstuk 2). Het raamwerk beschouwt ‘beheer’ als de belangrijkste drijvende kracht en onderscheidt verder ecologische eigenschappen, ecosysteemfuncties en -diensten. Ecologische eigenschappen zijn bijvoorbeeld bodemkenmerken, nutriëntenkringlopen en biologische diversiteit. Deze eigenschappen ondersteunen de ecosysteemfunctie, oftewel de capaciteit van het ecosysteem om ecosysteemdiensten te bieden. Deze capaciteit geeft de potentiele levering van ecosysteemdiensten aan en wordt geanalyseerd met zogenaamde 'state-indicatoren'. Voorbeelden hiervan zijn de beschikbare visstand, potentiele koolstofopslag en bezoekerscapaciteit. Of deze capaciteit daadwerkelijk wordt benut, oftewel wordt ‘omgezet’ in een ecosysteemdienst, hangt af van vele factoren, zoals toegankelijkheid van een gebied, de vraag naar ecosysteemdiensten en besluitvorming omtrent beheer. 'Performance-indicatoren' worden gebruikt om de feitelijke levering van ecosysteemdiensten te kwantificeren. Hoofdstuk 2 identificeert criteria om de ecosysteemdienstindicatoren en het selectieproces te evalueren. Het selectieproces moet flexibel en consistent te zijn, terwijl de indicatoren veelomvattend, meetbaar in relatie tot tijd en ruimte, schaalbaar en kwantificeerbaar dienen te zijn. De bruikbaarheid van een indicator hangt tevens af van databeschikbaarheid, de geloofwaardigheid en toepasbaarheid op meerdere gebieden. Het raamwerk werd toegepast in drie case studies, variërend van landschap- tot bioomniveau: in Nationaal Landschap 'Het Groene Woud' in Noord Brabant (Hoofdstuk 3), mangrovesystemen in Java, Indonesië (Hoofdstuk 4), en voor ‘rangelands’ op globale schaal (Hoofdstuk 5).

De doelstelling van de case study in Hoofdstuk 3 is het kwantificeren, karteren en modelleren van het effect van beheer op acht ecosysteemdiensten, waarbij expliciet onderscheid wordt gemaakt tussen potentiële en feitelijke diensten. 'Het Groene Woud' is een typisch Nederlands multifunctioneel landelijk gebied. Lokale beleidsplannen zijn gericht op het combineren van landbouw, natuurbescherming en natuurrecreatie. Het raamwerk werd hier toegepast om te bepalen welke ecologische eigenschappen de levering van ecosysteemdiensten ondersteunen en welke indicatoren geschikt zijn voor het kwantificeren van de ‘state-’ en ‘performance-indicatoren’. De meeste ecosysteemdiensten in ‘Het Groene Woud’ zijn sterk afhankelijk van de omvang en connectiviteit van natuurgebieden. Voorts spelen zogenaamde 'groene landschapselementen' (bijvoorbeeld hagen, bomenrijen en bermen) een cruciale rol voor diensten als bestuiving, biologische plaagbestrijding, fijnstofinvang en bescherming van leefgebied voor dieren. De geïdentificeerde indicatoren helpen om de interactie tussen verschillende ecosysteemdiensten te begrijpen, en de resulterende ruimtelijke modellen helpen om de gebieden te identificeren die het meest belangrijk zijn voor het leveren van meerdere ecosysteemdiensten. Tevens werden twee scenario’s, ‘landbouwintensivering’ en ‘grootschalige natuurbescherming’, onderzocht om de gevolgen te bepalen op zuivelproductie, fijnstofinvang en natuurrecreatie. De scenarioanalyse onderstreept dat groene landschapselementen en natuurgebieden van belang zijn voor recreatie en regulerende ecosysteemdiensten. Intensiverende zuivelproductie gaat ten koste van alle onderzochte ecosysteemdiensten. De vastgestelde ecosysteemdienstindicatoren en relaties daartussen kunnen bijdragen tot het kwantificeren van ecosysteemdiensten in andere landschappen.

In Hoofdstuk 4 ga ik verder in op de beleids- en ecologische aspecten van beheer, welke zijn bestudeerd in de mangrovekustgebieden van Java, Indonesië. Sinds de jaren ’80 is meer dan de helft van de Indonesische mangroven gedegradeerd of omgezet in aquacultuur, landbouw en stedelijk gebied. Java is gekozen als studiegebied omdat het eiland het sterkst is veranderd door ontwikkeling van aquacultuur en omdat overheidsbeslissingen hier veelal voor het eerst worden geïmplementeerd. De consequenties van landgebruiksbeslissingen zijn beoordeeld door de gevolgen van verschillende ‘regimes’ op mangrove ecosysteemdiensten te onderzoeken. Regimes zijn menselijke activiteiten die gezamenlijk het landgebruik bepalen. Ik onderscheid vijf brede categorieën: *natuurlijke* mangroven, *lage intensiteit gebruikte* mangroven en *hoge intensiteit gebruikte* mangroven, mangroven *omgezet ten behoeve van aquacultuur* en, tot slot, *niet langer gebruikte verlaten aquacultuursystemen*. *Natuurlijke* mangroven worden slechts op zeer kleine schaal gebruikt, zonder het ecosysteem nadelig te beïnvloeden. De brede regime-categorieën zijn verder onderverdeeld in elf specifieke regimes, op basis van wettelijke status, beheerindicatoren en ecologische kenmerken. Zeven ecosysteemdiensten zijn geanalyseerd: voedsel (vis en garnalen), hout en andere bosproducten, kustbescherming, koolstofvastlegging, waterzuivering, habitat voor vis en garnalen, en natuurrecreatie. Deze ecosysteemdiensten zijn geselecteerd in samenspraak met lokale beleidsmakers. Vervolgens zijn de belangrijkste ecologische eigenschappen die ten grondslag liggen aan de ecosysteemdiensten bepaald, en ‘state-’ en ‘performance-indicatoren’ geïdentificeerd. De levering van ecosysteemdiensten is geschat en vervolgens gescoord voor elk regime door ecologische en beheerkenmerken van de regimes te relateren aan ecosysteemdienstindicatoren. Belangrijke indicatoren voor de meeste diensten zijn maximale boomleeftijd, soortenrijkdom en structurele diversiteit. *Natuurlijke* mangroven scoorden het hoogst voor de meeste diensten, behalve voor voedsel. De hoge voedselproductie van de *aquacultuur* regimes gaat ten koste van alle andere ecosysteemdiensten. *Aquacultuur* levert zelfs 'negatieve diensten' op, in de vorm van koolstofemissie, watervervuiling en verhoogd risico op overstromingen. Transities tussen regimes zijn tevens geïllustreerd in een diagram dat de gevolgen van landgebruiksbeslissingen laat zien. Zo kan het aanleggen van *plantages* en *silvo-aquacultuur* op locaties van voormalige aquacultuurvijvers de ‘negatieve diensten’ tegengaan, terwijl aanzienlijke hoeveelheden voedsel en hout geleverd kunnen worden. Bevindingen van deze studie zijn door lokale beleidsmakers verwerkt in nieuwe plannen met betrekking tot duurzame aquacultuur en mangroverehabilitatie.

Hoofdstuk 5 onderzoekt de effecten van verschillende regimes op bodemerosie en run-off van oppervlaktewater in door waterschaarste gekenmerkte 'rangelands'. Wereldwijd zijn meer dan één miljard mensen afhankelijk van rangelands voor hun levensonderhoud, omdat ze er hun vee laten grazen en landbouw bedrijven. Elf nieuwe regimes zijn geanalyseerd voor deze rangelands. Elk regime is gebaseerd op tien kwalitatieve indicatoren die verschillende begrazingsintensiteiten en natuurbeschermingsstrategieën weerspiegelen. Belangrijke indicatoren voor het kwantificeren van bodemerosie en run-off zijn de helling, bodemtextuur en –bedekkingsgraad en het organisch stofgehalte. Zowel gemiddeld jaarlijks bodemverlies als run-off nemen toe in de reeks van de regimes *onbegraasde natuurlijke* rangelands, *lage intensiteit begraasde* rangelands, *hoge intensiteit begraasde* rangelands en *aangelegde weilanden*. Bodemverlies bedroeg, respectievelijk, 717 (SF (standaardfout) = 388), 1370 (SF = 648), 4048 (SF = 1.517) en 4249 (SF = 1.529) kg ha^{-1} $jaar^{-1}$. Run-off bedroeg voor dezelfde regimes 98 (SF = 42), 170 (SF = 43), 505 (SF = 113) en 919 (SF = 267) m^3 ha^{-1} $jaar^{-1}$. Verder bleek dat het beëindigen van begrazing alsmede het combineren van begrazing en bosaanplanting bodemverlies en run-off kunnen verminderen. De bevindingen onderstrepen dat bodemerosie en run-off sterk verschillen per regime en dat het behoud en herstel van kwetsbare rangelands deze risico's kan verminderen.

Mijn proefschrift toont aan dat beheer en landgebruik ecosysteemdiensten sterk beïnvloeden. Natuurbescherming in *natuurlijke* regimes behoudt biodiversiteit maar ondersteunt bovendien cruciale regulerende en habitatdiensten, recreatiemogelijkheden en zelfs voedsel en bosproducten. *Lage intensiteit gebruikte* regimes kunnen veel ecosysteemdiensten leveren, maar trade-offs tussen de productie- en regulerende diensten kunnen optreden. Beheer ter ondersteuning van intensieve voedsel- en grondstoffenproductie heeft meestal negatieve effecten op recreatiemogelijkheden en cruciale regulerende diensten. De combinatie van intensieve productie met actief natuurherstel kan deze negatieve effecten gedeeltelijk verzachten. Ik maak expliciet onderscheid tussen *natuurlijke* en *omgezette* regimes, die worden gebruikt voor intensieve voedsel of grondstoffenproductie. Deze intensive productie gaat meestal ten koste van alle andere ecosysteemdiensten en kan zelfs leiden tot 'negatieve diensten'. *Niet langer gebruikte en verlaten* systemen leveren onbeheerd weinig ecosysteemdiensten maar zijn potentieel waardevol voor natuurherstel, afhankelijk van de oorspronkelijke ecosystemen.

Bevindingen in dit proefschrift zijn gebaseerd op een raamwerk voor indicatorenselectie en een combinatie van verschillende informatiebronnen en onderzoeksmethodes, zoals een indicator-interactiediagram, een typologie van regimes en een aanpak om de resultaten te integreren. In Hoofdstuk 6 stel ik voor hoe deze aanpak kan worden geïntegreerd in een 'toolbox'. Dit proefschrift presenteert een zeer volledige set van kwantitatieve en kwalitatieve indicatoren voor het kwantificeren van beheereffecten op de levering van ecosysteemdiensten. De genoemde stappen in 'toolbox' kunnen worden gebruikt om beheerders en beleidsmakers te informeren over de gevolgen van beslissingen met betrekking tot natuurbehoud, landgebruiksintensivering, het omzetten van natuur ten behoeve van intensieve voedselproductie, en het herstel van verlaten, onbeheerd land. Dit proefschrift biedt een handvat voor het voorkomen van verdere aantasting van het aardoppervlak en het verlies van ecosysteemdiensten.

SUMMARY

Humans have altered a large proportion of the Earth's ecosystems to meet growing demands for living space and food, fresh water and other natural resources. The most dominant transformations relate to urban expansion and land conversion to intensive agriculture. My research focuses on management and its effect on ecosystem services. Management involves the human activities that determine land-use purposes and thereby directly affect land cover. Examples include applying pesticides to increase food production, constructing facilitates for ecotourism and clearing unwanted plants to protect endemic biodiversity. Ecosystem services, which are the contributions of ecosystems to human wellbeing, have become an increasingly popular concept to demonstrate how biodiversity loss and land degradation affect an ecosystem's capacity to provide critical services, such as fresh water, food and fuel wood. The worldwide transformation of ecosystems to support mono-functional land use suggest that managers and decision makers have limited understanding of what is at stake in terms of economic and social consequences. Understanding the effects of management on ecosystem services is crucial in assessing the consequences of policies and decisions. Compiling and analysing empirical evidence to support land management decisions is required, as most management tends to be grounded in poorly verified assumptions. Therefore, my thesis aims to quantify the effects of land management on ecosystem services provision.

To achieve this research aim, I developed an indicator-based approach by advancing a research framework for systematic indicator selection (Chapter 2). 'Management' was included in the framework as a driving factor and three clearly distinguishable elements of service provision were defined, namely ecosystem properties, ecosystem functions and ecosystem services. Ecosystem properties include the conditions, structures and processes of ecosystems, such as soil properties, nutrient cycles and biological diversity. These properties underpin the ecosystem's capacity to provide ecosystem services. This capacity is defined as ecosystem function and analysed through so-called 'state indicators', such as fish stock, carbon storage and potential number of recreants. Whether this capacity is converted into an ecosystem service depends on many factors, such as accessibility, demand for services and management choices. 'Performance indicators' are used to quantify actual ecosystem service provision. Criteria were identified to evaluate ecosystem service indicators and their selection process. The selection process needs to be flexible and consistent, while indicators need to be comprehensive, understandable for multiple end-users, temporally and spatially explicit, scalable and quantifiable. An indicator's usefulness also depends on data availability, credibility and portability. The framework was applied in three case studies, ranging from landscape to biome level. Management effects on multiple ecosystem services were quantified and modelled in Dutch rural landscape 'Het Groene Woud' (Chapter 3). Management regimes were developed and ecosystem service provision evaluated in mangrove systems in Java, Indonesia (Chapter 4), and in global semi-arid and sub-humid rangelands (Chapter 5).

'Het Groene Woud' is a typical multifunctional rural landscape in the Netherlands. Local policy and spatial planning strategies emphasise the need for combining agricultural production, nature protection and facilitating nature-based recreation and tourism. The case study aimed to quantify, map and model the effect of management on eight ecosystem services, thereby explicitly distinguishing between potential and actual service provision (Chapter 3). The research framework

was applied to identify which ecosystem properties underpin ecosystem service provision and which indicators are suitable for quantifying state and performance. Most ecosystem services in Het Groene Woud depend strongly on the extent and connectivity of natural areas and so-called 'green landscape elements' (e.g. hedgerows, treelines and berms) were found to play a crucial role for services, such as pollination, biological pest control, air quality regulation and habitat for migratory species. The identified indicators further explained the interaction between different ecosystem services and the resulting high-resolution maps and spatial models helped to identify key areas for providing multiple ecosystem services. Relevant management variables related mainly to land use and land cover. Two scenarios, 'agricultural intensification' and 'large-scale nature protection', served to illustrate management effects on dairy production, air quality regulation (fine dust capture) and nature-based recreation (walking). The scenario analysis underlined that green landscape elements and natural areas are important for providing recreation and regulating services, whereas intensifying livestock grazing would increase milk and fodder provision at the cost of all other ecosystem services. The study's generic relationships between ecosystem service indicators should enable the quantification of ecosystem services in other landscapes.

Management's policy and ecological aspects were further explored in a case study that I conducted in the coastal mangroves of Java, Indonesia (Chapter 4). More than half of Indonesia's mangroves have been degraded or converted to aquaculture, agriculture and urban areas since the 1980s. Java is the most heavily affected by management activities and different land uses because most government decisions are first implemented here. The consequences of management decisions were assessed by studying the effects of different management regimes on mangrove ecosystem services in Java, Indonesia. Management regimes are 'the bundle of human activities that serve land-use purposes' and five broad categories were distinguished: *natural* mangroves, *low intensity* mangroves and *high intensity use* mangroves, mangroves *converted for aquaculture* and *abandoned aquaculture* systems. Eleven specific management regimes were then developed, based on legal status, management indicators and ecological characteristics. Seven ecosystem services were analysed: food (fish and shrimp), raw materials, coastal protection, carbon sequestration, water purification, nursery for fish and shrimp, and nature-based recreation. These ecosystem services were selected in dialogue with local decision makers. Recurring key ecosystem properties underpinning service provision were reviewed and state and performance indicators were identified. Ecosystem service provision was estimated and scored for each management regime, by relating the regimes' ecological characteristics with indicators. Because the regulating and recreation services were better explained by qualitative indicators, qualitative and quantitate information for this assessment were combined. Key indicators for most services were maximum tree age, species richness and structural diversity. *Natural* mangroves scored highest for most services, except for food. The high fish and shrimp production by *aquaculture* regimes occurs at the expense of other ecosystem services. Aquaculture was even found to provide 'disservices' in the form of carbon and water pollutant emission and increased flooding risk. Rehabilitating *aquaculture* systems into *plantations* and *silvo-fisheries* reverses this loss, while still providing shrimp or raw materials. Transitions between management regimes were illustrated in a diagram, to show consequences of management decisions. Our findings have assisted local decision makers to make better informed management decisions regarding sustainable aquaculture and mangrove rehabilitation.

Globally, over one billion people's livelihoods depend on dry rangelands through livestock grazing and agriculture. Livestock grazing and other management activities can erode soils, increase surface runoff and reduce water availability. Therefore, the effects of different management regimes on soil erosion and surface runoff in semi-arid to sub-humid rangelands were assessed (Chapter 5). Eleven management regimes were developed and analysed. These regimes were based on ten qualitative management indicators that reflected different livestock grazing intensities and rangeland conservation strategies. Our review yielded key indicators for quantifying soil erosion and surface runoff. Slope, soil organic matter, soil texture and canopy cover explained both soil erosion surface runoff the best. Canopy cover correlated negatively, while slope correlated positively to soil loss and runoff in all management regimes. The values of all quantitative indicators were compared per management regime. Mean annual soil loss values in the *natural ungrazed*, *low intensity grazed*, *high intensity grazed* rangelands and *man-made pastures* regimes were, respectively, 717 (SE=388), 1370 (SE=648), 4048 (SE=1517) and 4249 (SE=1529) kg ha-1 yr-1. Mean surface runoff values for the same regimes were 98 (SE=42), 170 (SE=43), 505 (SE=113) and 919 (SE=267) m3 ha-1 yr-1, respectively. A preliminary analysis into differences between management regimes showed that livestock grazing abandonment and exotic plantations reduces soil loss and runoff. The findings underline that soil erosion and surface runoff differ per management regime and that conserving and restoring vulnerable semi-arid and sub-humid rangelands reduce these risks.

In conclusion, my study clearly shows the effects of management on ecosystem services. *Natural* management regimes conserve nature and provide critical regulating and habitat services, recreation opportunities, food and raw materials. *Low intensity use* management regimes can provide most ecosystem services, but trade-offs between provisioning and regulating services occur locally. Management to support intensified food and raw materials production (i.e. *high intensity use* management regimes), generally has adverse effects on recreation opportunities and regulating services, such as carbon sequestration, erosion prevention, water flow regulation and coastal protection. Combining intensive production with active restoration and rehabilitation can partly mitigate these negative effects. I explicitly distinguish *converted* lands that are now used for intensive food or fibre production. This high production generally occurs at the cost of all other ecosystem services and can even result in 'dis-services', such as carbon emissions, water pollution and high soil erosion. Finally, *abandoned* lands are a valuable option to restore nature, but, depending on their underlying actual ecosystem, these lands provide few ecosystem services if left unmanaged.

This PhD thesis' findings are based on various approaches and information sources, a framework for indicator selection, an indicator interaction diagram, a typology of management regimes and an approach to integrate the results and illustrate transitions between management regimes. The resulting 'toolbox' integrates a comprehensive set of quantitative and qualitative indicators for quantifying management effects on ecosystem service provision. This toolbox can be used to inform decision makers on the consequences of management decisions regarding nature conservation, land-use intensification, converting nature to support intensive cultivation and restoring abandoned land. This thesis provides an important step in preventing further land degradation and loss of ecosystem services by better managing the Earth's land.

ABOUT THE AUTHOR

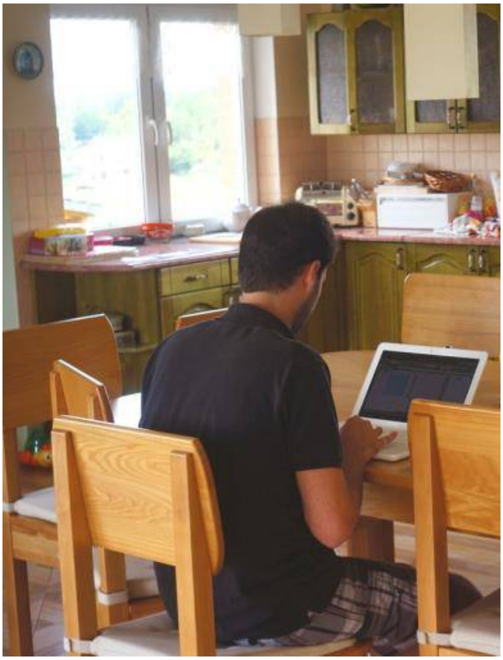
Alexander van Oudenhoven was born in The Hague, The Netherlands, on June 1st 1983. He spent the first years of primary school in Gabon, which is home to some of Africa's most biodiverse rainforests. Almost 80% of the country is forested, and upon seeing this new world from a propeller airplane for the first time, little Alexander described the land as "full of broccoli". Despite this lapse in judgement, he developed a profound love for Gabon's unique natural beauty; surfing hippos, forest elephants, endless herds of buffalo, mysterious primates and an inquisitive local king cobra.

After finishing his secondary education in 2002, Alexander spent half a year in Sri Lanka, to volunteer as an English teacher at several schools and a local training centre. Between 2003 and 2008, he took part in the BSc and MSc programme 'Environmental Sciences' at Wageningen University, specialising in 'Environmental Systems Analysis' in both programmes. For his MSc thesis, Alexander analysed the effects of climate change on the spatial distribution of the oak processionary caterpillar in The Netherlands. This research was done for 'De Natuurkalender' ('Nature's Calendar'), a long-running phenological network. Alexander was initiated into ecosystem services research while doing an internship at the World Resources Institute (Washington, DC), where he investigated indicators used in Sub-Global Millennium Assessments.

Alexander's PhD research dealt with the effects of management on ecosystem services, which is a much referred to yet understudied research topic. Important elements of his research include conceptualising ecosystem service provision and management, finding and compiling indicators for quantifying ecosystem services and developing management regimes. These elements were applied to case studies in Dutch national landscape 'Het Groene Woud', mangroves in Java (Indonesia) and global semi-arid to dry sub-humid rangelands. During his PhD, Alexander was actively involved in the Ecosystem Services Partnership, initiatives by UNEP-WCMC and the CBD, and work commissioned by UNCCD and STAP-GEF. From 2010 onwards, Alexander was managing editor of the *International Journal of Biodiversity Science, Ecosystem Services & Management.*

For several years, Alexander was closely involved in projects of 'Living Lands', a South African not-for-profit organisation that aims to restore living landscapes in a participatory manner. Restoration also featured strongly in the 'Mangrove Capital' project with Wetlands Internationals and partners, which Alexander participated in between 2012 and 2014. Alexander compiled the scientific basis for communicating the multiple benefits that mangrove conservation and restoration can provide in comparison to converting mangroves for aquaculture. His findings and management regime typology were considered by decision makers developing sustainable aquaculture and mangrove rehabilitation. Alexander also worked with PBL to develop an Ecosystem Services Quantification Database (ESQD), which contains extensive indicator sets and values thereof for multiple ecosystem services in relation to land cover, land use and management classifications.

Alexander is passionate about doing applied research in support of landscape restoration and planning, and sustainable use of natural resources. Although quantifying ecosystem services can rarely be an ultimate research goal, it can contribute to managing the Earth's land cover in a more informed way.



Even during vacations, the computer was never far away. Photo: Rafał Wietsma

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- Coordinating SENSE Research Cluster XIII 'Land use, spatial analysis and modelling / ecosystem and landscape services' (2009-2012)
- Managing Editor 'International Journal of Biodiversity Science, Ecosystem Services & Management' (2009-2013)
- Co-organising 4th ESP conference (4 days), Wageningen (2011)
- Supervising four MSc thesis students (2010-2013)
- Lecturing in MSc course 'Integrated ecosystem assessment In regional management' (2010-2012)

Selection of Oral Presentations

- *Quantifying & modelling the effect of restoration on ecosystem services*. Ecosystem Services Partnership (ESP) Conference – Ecosystem Services: Integrating Science and Practice, 4-7 October 2011, Wageningen, The Netherlands
- *Using indicators to quantify ecosystem services - examples from the field and literature*. Ecosystem Services, Human Values & Global Change – Expert Workshop, 24-27 April 2012, Prague, Czech Republic
- *Towards an ecosystem services database: quantifying and structuring ecosystem service provision based on the cascade model*. Joint Workshop on Indication, Integration and Application of Ecosystem Services in Decision Making, 6-8 May 2013, Kiel, Germany
- *Ecosystem services in natural, restored and converted mangrove systems in Java, Indonesia*. NRG BESS Early Career Researcher Conference, 7-8 September 2014, Southampton, United Kingdom

SENSE Coordinator PhD Education

Dr. ing. Monique Gulickx

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