**Propositions**

1. Fully disentangling human and ecosystem contributions is impossible in cultural landscapes where both are necessary to produce human benefits.
   (this thesis)

2. To protect the ecological system for its own sake and for the benefit of humans, both biodiversity and ecosystem services need to be included in ecosystem accounting and conservation planning.
   (this thesis)

3. Maps and monetary values are very powerful communication tools for (mis)guiding public opinion.

4. Value is a multifaceted concept that guarantees critical discussion in interdisciplinary research.

5. A PhD thesis is authored by one person, but developed by multiple minds.

6. Nature is neither fair nor unfair, such phrases belong to humankind.
   (inspired by Carsten Jensen's novel *We, the Drowned,* 2011, Vintage)

7. Investors in fossil fuels are reckless, as besides putting the planet at risk of further climate change, they are putting their own financial stability at risk.

Propositions belonging to the thesis entitled:
"Accounting for ecosystem services and biodiversity in Limburg province, the Netherlands"

Roy Remme
Wageningen, 13 January 2016

# Accounting for ecosystem services and biodiversity in Limburg province, the Netherlands

Roy P. Remme

# Accounting for ecosystem services and biodiversity in Limburg province, the Netherlands

Roy P. Remme

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# Accounting for ecosystem services and biodiversity in Limburg province, the Netherlands

**Roy P. Remme**

**Thesis**
submitted in fulfilment of the requirements for the degree of doctor
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## Table of contents

# Chapter 1 - Introduction

Partly based on:
Schröter, M., Remme R.P., Sumarga, E., Barton, D.N., Hein, L., 2015. Lessons learned for spatial modelling of ecosystem services in support of ecosystem accounting. *Ecosystem Services* 13, 64-69.

## 1.1 Background

### *1.1.1 Ecosystem services*

Humans depend on ecosystem contributions to maintain their quality of life (De Groot et al., 2002; Díaz et al., 2006; Haines-Young and Potschin, 2013). These contributions are referred to as ecosystem services (ESs) (Haines-Young and Potschin, 2010a; UN et al., 2014a). ESs link ecosystem processes and societal wellbeing (Haines-Young and Potschin, 2010a; TEEB, 2010). Ecosystems deliver provisioning services that provide major commodities (e.g. food, water, raw materials and energy sources), regulating services that regulate environmental flows (e.g. cycling of nutrients and filtering air) and cultural services that provide opportunities for relaxation, spiritual interactions or education (e.g. hiking and painting landscapes). Notions of the ES concept have been used since the 1970s, studying ecosystem functions and benefits that influence humans (Gómez-Baggethun et al., 2010; Reyers et al., 2010). The ES concept was mainstreamed in the 1990s, as a result of several landmark studies (e.g. De Groot, 1992; Costanza et al., 1997; Daily, 1997) and was institutionalised over the past fifteen years due to the Millennium Ecosystem Assessment (MA, 2005), The Economics of Ecosystems and Biodiversity (TEEB, 2010) and, more recently, the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) (Inouye, 2014; Díaz et al., 2015).

Multiple definitions and classifications for ESs exist and these are strongly debated in the scientific literature (Costanza, 2008; Fisher et al., 2009). Although multiple authors have called for standardisation of the ES definition (Boyd and Banzhaf, 2007; Nahlik et al., 2012), a diversity of definitions also provides advantages. Different interpretations of the ES concept and the diversity of definitions can spur creativity and provide an opportunity for researchers and decision makers with different backgrounds to use the concept (Schröter et al., 2014a). In this thesis I define ESs as “the contributions of ecosystems to benefits used in economic and other human activity” (UN et al., 2014a), a definition which best suits the purpose of my research. There are three leading ES classifications. The first was introduced by the MA (MA, 2005), the second by TEEB (TEEB, 2010) and the third, the Common International Classification for Ecosystem Services (CICES), by Haines-Young and Potschin (2010b). Although, the classifications have clear differences, they have three categories in common, namely the categories provisioning, regulating and cultural services. These three categories are used in this thesis to classify ESs.

ESs are being studied from many perspectives, ranging from purely ecological to economic research and socio-cultural appraisals, and many different (inter)disciplinary approaches have been developed. The ES concept has connected ecology and economics by bridging methodologies, language and disciplinary differences (Liu et al., 2010). In addition, the concept provides an approach to explicitly or implicitly link humans and nature in research (Reyers et al., 2010). The broadness of the concept and its interdisciplinary character provides opportunities to tackle complex environmental issues (Kremen, 2005; Cowling et al., 2008; Carpenter et al., 2009; Schröter et al., 2014a). To study ESs, many different types of indicators have been used, reflecting ecological, economic and social aspects and values. Ecological indicators are most often aimed at quantifying biophysical amounts of different ESs (e.g. van Oudenhoven et al., 2012), while economic and social indicators usually reflect human appreciation or use of ESs (Liu et al., 2010; Chan et al., 2012; Pert et al., 2015). Both contribute to debates on preparing or making ecosystem-related decisions that depend on ecological, environmental, economic and social knowledge (e.g. De Groot et al., 2012). In this thesis I use both biophysical and economic indicators to study ESs in the context of ecosystem accounting.

### *1.1.2 Ecosystem accounting*

The central topic of this thesis is the operationalization and the application of ecosystem accounting. Ecosystem accounting is a systematic approach to measure and monitor ESs and ecosystem conditions over time (Edens and Hein, 2013; Obst and Vardon, 2014). Ecosystem accounting answers the calls to systematically monitor and assess ESs (Carpenter et al., 2009; Larigauderie et al., 2012), and operationalising the ES concept for decision making and planning (Cowling et al., 2008; Daily et al., 2009). The development of ecosystem accounting stemmed from the strong interest to understand the economic implications of ecosystem degradation and changes in ESs provision (MA, 2005; European Commission, 2011; UK NEA, 2011). Steady progress in conceptualizing ecosystem accounting occurred in recent years, including debates on terminology and definitions (Boyd and Banzhaf, 2007; Edens and Hein, 2013), types of indicators that inform scientists and decision makers (Weber, 2007; Stoneham et al., 2012; Schröter et al., 2014b) and the types of accounts that should be included in an accounting framework (Hein et al., 2015).

Under the auspices of the United Nations the System of Environmental-Economic Accounting - Experimental Ecosystem Accounting (SEEA-EEA) is developed to guide the implementation of ecosystem accounting (UN et al., 2014a). One of the main objectives of the SEEA-EEA is to measure ESs in a way that is aligned with national accounting (as defined in the System for National Accounts (SNA), UN et al., 2009). The SEEA-EEA complements the SEEA Central Framework (SEEA-CF), which serves as an international statistical standard and guideline for environmental-economic accounting world-wide (UN et al., 2014b). The SEEA-EEA is designed to accommodate the integration of ESs into accounting, which is not possible with the compartmental approach of the SEEA-CF (Edens and Hein, 2013; UN et al., 2014a). The SEEA-EEA outlines the components of ecosystems and their services that should be accounted for, and suggesting approaches and methods to create accounts. ESs have a spatial component, which is recognized and strongly integrated in the SEEA-EEA. Accounts should be developed using both biophysical indicators (e.g. kg, $m^3$ or visitor numbers) and monetary indicators, to provide a comprehensive overview of ESs and their different values (UN et al., 2014a). Monetary valuation for accounting requires methods that are aligned with SNA to compare values with accounts for other economic activities (Edens and Hein, 2013). Ecosystem accounting valuation methods are based on exchange values and explicitly exclude consumer surplus (i.e. the difference between a consumer's willingness to pay and what they actually pay). The SEEA-EEA is still under development and the guidelines remain to be empirically tested for different conditions (e.g. countries or regions).

To develop a full set of ecosystem accounts, multiple aspects of interactions between ecosystems and society should be integrated. Four main aspects for accounting are distinguished: ecosystem condition, capacity to provide ESs, ES flow and ES demand. A condition account informs on the state of the ecosystems in an area. The selected condition indicators reflect key ecosystem processes and components that both influence ecosystem functioning (UN et al., 2014a). These include nutrient cycles, soil quality and ecosystem productivity. A capacity account measures the capacity of an ecosystem to generate ESs. Capacity is defined as the long-term potential of an ecosystem to sustainably generate ESs based on current ecosystem conditions and management (Villamagna et al., 2013; Schröter et al., 2014b). Capacity is time- and site-specific, and it can change due to human-induced or natural alterations (Villamagna et al., 2013). An ecosystem can have the capacity to provide ESs, yet these ESs may not actually be used due to specific conditions (e.g. wood in inaccessible forests). Capacity indicators quantify the potential to generate specific ESs. An ES flow account captures the annual flows of ESs from ecosystems to society. ES flow reflects actual use or delivery of each service (Villamagna et al., 2013; Schröter et al., 2014b). ES flow accounts are the core of ecosystem accounting as they represent the actual interactions between ecosystems and society. The flow accounts and the development of relevant indicators are the central focus of this thesis. ES demand reflects the needs of society for certain services, regardless of whether they are actually being provided (Villamagna et al., 2013; Wolff et al., 2015). Demand may exceed ES flows in an area, causing a local shortage of ESs.

Complementary to these four different ecosystem accounts a biodiversity account is also introduced in the SEEA-EEA. This account reflects the aspects of biodiversity that are considered important for ecosystem management and conservation (often based on intrinsic value, i.e. nature is valuable in itself, independent from human appreciation), such as the presence of threatened species or habitat conditions (UN et al., 2014a; Hein et al., 2015). The relationship between biodiversity and ESs is further introduced later in this chapter.

### *1.1.3 Spatial ecosystem service modelling*

To capture the spatial heterogeneity of ecosystems and ES delivery spatial approaches are needed for ecosystem accounting (Edens and Hein, 2013; UN et al., 2014a). Spatial ES models are used to fully cover spatial heterogeneity of ecosystems and ESs in ecosystem accounting (Schröter et al., 2015). Spatial ES modelling has rapidly developed into a popular field of research, with a plurality of methods, area sizes and types of ESs being studied (Martínez-Harms and Balvanera, 2012; Crossman et al., 2013b; Nemec and Raudsepp-Hearne, 2013; Malinga et al., 2015). Many of these models do not use generic ecosystems as the basis of their analysis. Instead, they base their simulations on the heterogeneity of landscapes and environmental gradients to capture the uniqueness of each locality and the specific potential for each ecosystem to deliver ESs. Most ES models are therefore spatially explicit and simulate continuous geographic patterns rather than administrative units.

Building on existing classifications (Eigenbrod et al., 2010; Martínez-Harms and Balvanera, 2012) seven types of static ES modelling methods can be distinguished that are relevant for ecosystem accounting (Schröter et al., 2015), and that I apply in this thesis. These methods include four types of look-up tables (LUT: binary, qualitative, aggregated statics and multiple layer), causal relationships, spatial interpolation, and environmental regression. Binary LUT assess the presence or absence of specific ESs based on land-use or land-cover patterns. Qualitative LUT weigh different land-use or land-cover classes according to their capacity to provide ESs. Aggregated statistics LUT assign ES values based on statistics or other assessments of land-use or land-cover data or boundaries of administrative units. Multiple layer LUT assign ES values to land units based on cross tabulations that are created by overlay of different information layers (e.g. land-use, soils, climate and management strategies). The causal relationship method simulates ESs based on existing process understanding of how environmental variables affect the distribution and abundance of ESs. Spatial interpolation predicts ESs throughout a region based on spatial autocorrelation of measured local data points. Such interpolation is sometimes informed by additional environmental layers. Environmental regression models simulate ESs through the empirical relationship between environmental layers as explanatory variables and measured ES data as response variables (Martínez-Harms and Balvanera, 2012). Although many modelling approaches can be applied to specific localities or ecosystems, ecosystem accounting is generally focussed on larger regions. This is achieved by combining spatially explicit land-use or land-cover data with suitable environmental and socio-economic data, resulting in spatially explicit outcomes. While such spatial modelling methods are most frequently used to map biophysical ES indicators, experience with spatial modelling related to monetary ES indicators has also increased exponentially (Schägner et al., 2013).

### *1.1.4 Ecosystem services, biodiversity and conservation*

Biodiversity is defined as "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (UN, 1992). Biodiversity is important because of its intrinsic value (Justus et al., 2009; Jax et al., 2013). However, biodiversity is also important because it regulates ecosystem properties and ESs that benefit humans (Díaz et al., 2006). Biodiversity thus also has instrumental value (Pearce and Moran, 1994; Jax et al., 2013), i.e. it is directly used or appreciated by humans (Turner et al., 2003). The intrinsic and instrumental values conflict when it comes to nature conservation (e.g. Justus et al., 2009; Jax et al., 2013) but in most areas ecosystems are poorly protected or are managed to provide specific ESs. In these areas, ecosystem accounts can help to improve management by measuring and monitor ESs and their values (Reid et al., 2006).

Biodiversity is rapidly declining globally as a result of human-induced pressures (Vitousek et al., 1997; Myers et al., 2000; Butchart et al., 2010). This also threatens the continued provision of ESs. The ES concept can therefore provide additional arguments and tools for conservation and broaden current conservation and ecosystem management practices (Balvanera et al., 2001; Armsworth et al., 2007; Schröter et al., 2014a). Biodiversity and ESs have a multifaceted relationship. Biodiversity can be regarded as underpinning ESs but also as a final ES itself, related to the human appreciation of the existence of wild plants and animals or genetic diversity (Mace et al., 2012). In this thesis, I primarily focus on biodiversity as a final ES, but also will incorporate the underpinning biodiversity aspects in ES models. Empirical evidence for the links between biodiversity and ESs is not yet established and ESs do not always promote biodiversity conservation (Cardinale et al., 2006; Ridder, 2008; Adams, 2014). Nevertheless, evidence for such links is gradually increasing (Mace et al., 2012; Reyers et al., 2012; Harrison et al., 2014) and some ESs can clearly be considered compatible with biodiversity conservation (Chan et al., 2011).

## 1.2 Challenges in accounting for ecosystem services and biodiversity

The foundations for ecosystem accounting have been laid with a clear relation to national accounting (Obst and Vardon, 2014), a defined purpose and terminology (Boyd and Banzhaf, 2007; Edens and Hein, 2013), an analysis of the necessary components and their relationships (Schröter et al., 2014b; Hein et al., 2015) and a preliminary set of internationally recognised guidelines (SEEA-EEA) (UN et al., 2014a). However, to fully operationalise ecosystem accounting multiple challenges still need to be overcome.

First, the different types of accounts within the ecosystem accounting framework require empirical testing. Ecosystems and their services are recognised to be spatially heterogeneous and therefore ecosystem accounting requires a spatial approach (Edens and Hein, 2013; UN et al., 2014a). Although extensive experience with spatial ESs analysis exists (Maes et al., 2012b; Martínez-Harms and Balvanera, 2012; Crossman et al., 2013b), the applicability of ES mapping and modelling methods for ecosystem accounting has rarely been rigorously tested (Schröter et al., 2014b). A comprehensive account for annual biophysical ES flows is yet to be developed and the feasibility of different modelling methods needs to be assessed. In addition to the lack of experience with spatial modelling the use of monetary ES valuation methods for ecosystem accounting require further research. Monetary valuation is an informative way to convey the importance of ESs to society and many different valuation methods have been developed (Liu et al., 2010; Turner et al., 2010). However, only a subset of these methods can be applied for ecosystem accounting, since valuation should be aligned with SNA (UN et al., 2014a; Obst et al., 2015). Valuation methods need to follow an exchange value approach, excluding consumer surplus from valuation, in order to be aligned with SNA (Edens and Hein, 2013). The applicability and suitability of ecosystem accounting valuation methods also remains to be empirically tested. Currently very few studies have tested the integration of spatial ES models and monetary valuation in the context of ecosystem accounting.

Second, the role of biodiversity in ecosystem accounting requires further research. The multi-layered relationship between biodiversity and ESs (Mace et al., 2012), makes measuring and monitoring of biodiversity necessary to acquire comprehensive ecosystem accounts (UN et al., 2014a; Hein et al., 2015). Biodiversity aspects, such as species diversity, are not captured by accounts focussing on ESs. Biodiversity can be measured in many ways, and over the past decades many types of indicators have been developed that emphasize its various aspects (Noss, 1990; Vačkář et al., 2012; Feest, 2013). For the purpose of ecosystem accounting an open challenge is to assess which set of indicators can be included in a biodiversity account.

Finally, the policy purposes of ecosystem accounting need to be further explored (Schröter et al., 2015). Integrated information from ecosystem accounting on both ESs and biodiversity could be used for decisions on conservation and land management. Moreover, the role of spatial ES analysis in conservation and spatial relations between ESs and biodiversity require further analysis (Cimon-Morin et al., 2013). ESs are increasingly considered alongside biodiversity in spatial conservation assessments (Chan et al., 2011; Egoh et al., 2014; Schröter et al., 2014c). The general premise is that inclusion of ESs in spatial conservation assessments will also benefit biodiversity and increase the cost-effectiveness of conservation. These ideas, however, are not conclusively supported by scientific evidence and require more research, addressing issues such as available cost data (Naidoo et al., 2006) and target setting for conservation goals (Luck et al., 2012).

## 1.3 Objectives and research questions

As outlined above, multiple challenges remain for ecosystem accounting to be operationalised and before it generates adequate and essential information that is applicable for decision making on ecosystem management and conservation. Therefore, this thesis aims foremost to empirically assess how spatial models for ES flows and biodiversity can be applied in the context of ecosystem accounting, management and conservation. Achieving this objective will contribute to the further development of accounting and improve understanding on ecosystem accounting implications for decision making relevant for ecosystem management.

The objective is addressed by studying biophysical ESs flows, monetary valuation for ecosystem accounting, spatial indicators for biodiversity accounting and the use of accounting information on ESs and biodiversity for developing conservation scenarios. The objective is addressed through the following research questions:

RQ1) How can biophysical ES flows be spatially modelled for ecosystem accounting?
RQ2) How can ESs be value in monetary terms for ecosystem accounting?
RQ3) Which species diversity indicators can be applied to develop a comprehensive biodiversity accounting framework?
RQ4) What are the effects of including ESs in planning for an expansion of a biodiversity conservation network?

Given the early phase of development of ecosystem accounting, addressing these research questions will considerably improve the general understanding of the possibilities and challenges of applying a spatial approach to account for ESs and biodiversity, and the broader applicability of ESs and biodiversity data generated for ecosystem accounting. The methodological development of spatial ES models is needed to account for ESs flows at different spatial extents and dimensions, and to capture the spatial variability and heterogeneity in landscapes. Biophysical ES data necessities and model uncertainties need to be assessed to develop accurate biophysical ES flow accounts, and monetary accounts. Monetary valuation methods for ecosystem accounting that are aligned with SNA, require testing to assess which methods are suitable and effective. Modelling both biophysical and monetary flows will provide a comprehensive understanding of spatial patterns of ES delivery. To integrate biodiversity into ecosystem accounting, an assessment of which biodiversity indicators provide relevant information is needed. By addressing the fourth research question, I attend to the policy relevance and wider applicability of ecosystem accounting data. In this thesis, I address such wider applicability by using ecosystem accounting outcomes in the context of conservation. I study whether incorporating ESs into a biodiversity conservation network increases cost-effectiveness through a scenario analysis.

To test the various spatial methods used in this thesis, the province of Limburg in the Netherlands is used as a case study area. The province provides a suitable case study to test ecosystem accounting methods as it is a large administrative unit with varied landscapes, contains a diverse set of ecosystems and is data rich in terms of quantitative environmental and social data. The thesis predominantly focusses on methodological development of ecosystem accounting and spatial approaches for conservation and relevant ecosystem management. Considering the early stage of development in ecosystem accounting and the relatively coarse spatial data that is used in some chapters, the results should not be interpreted as practical guidance for local policy makers but as an illustration of the potential of ecosystem accounting.

## 1.4 Study area

Limburg covers approximately 2,200 $km^2$ and is located in the south-east of the Netherlands (Figure 1.1). Limburg is densely populated with a total population of 1.1 million and a density of over 500 inhabitants per $km^2$(Statistics Netherlands, 2013f). Many centuries of intensive land management have led to a varied cultural landscape (Berendsen, 2005; Jongmans et al., 2013). Most natural ecosystems have been converted and those that remain are highly fragmented (Jongman, 2002). About 50% of the province is used for agricultural purposes, over 20% of the area is built-up, 15% is forest and the remaining 15% is other semi-natural vegetation and water (Hazeu, 2009).

Limburg harbours numerous species of national and even international importance (Statistics Netherlands et al., 2008) and provides habitats that are rare in the Netherlands, such as the calcareous grasslands in southern Limburg (Willems, 2001). Limburg has a yearly average temperature of 10°C and a yearly average precipitation of about 800mm (KNMI, 2011). The Meuse river runs through the province entering in the south at the Belgian border and running to the province’s most northern point. Southern Limburg is nationally renowned for its hilly landscape with limestone and is popular with domestic tourists and cyclists. The province contains three national parks, with distinctly different ecosystems. The Maasduinen runs along the eastern border of the Meuse in northern Limburg and contains one of the longest ridges of Eolian dunes in the Netherlands. The Groote Peel in central-western Limburg is a marshy heathland where peat was formerly extracted. The Meinweg in central-eastern Limburg is a unique natural terraced landscape formed by the Meuse river and is connected to German forests.

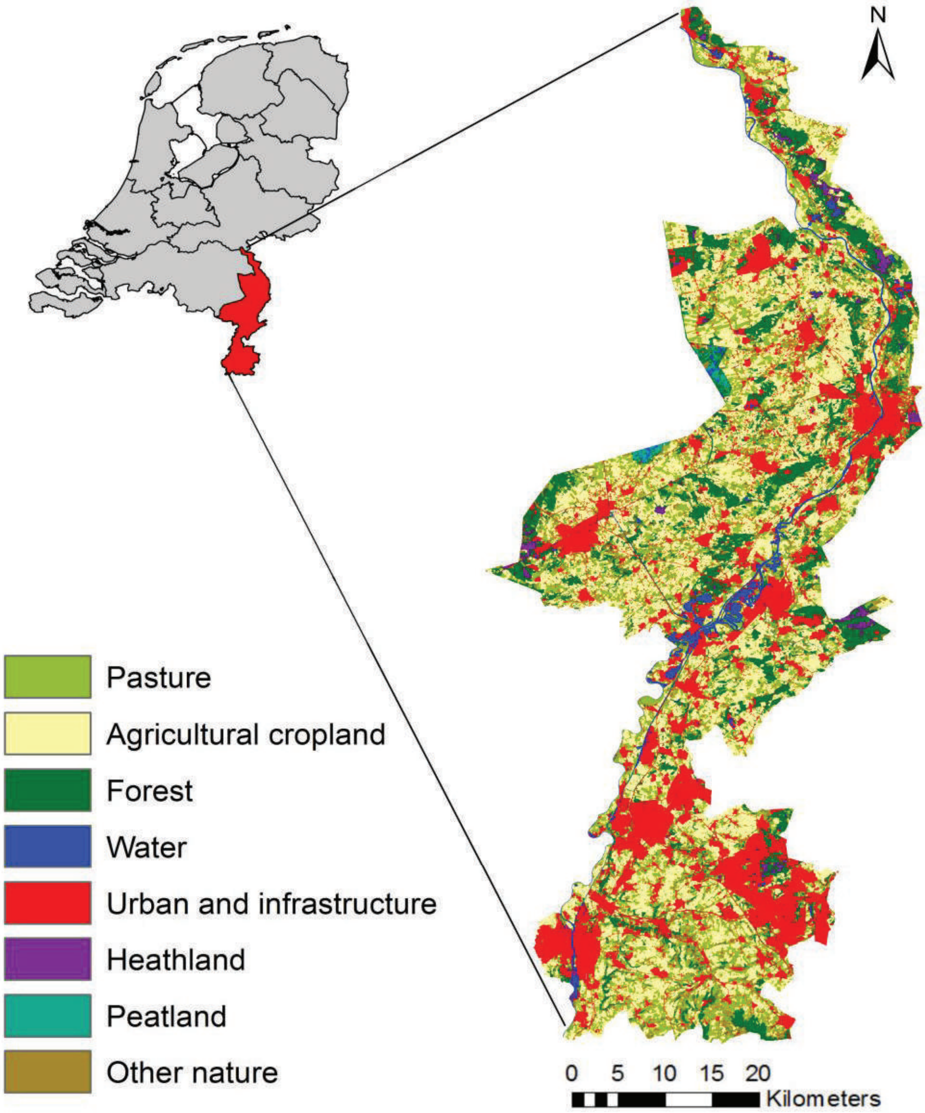


**Figure 1.1:** Location and land-cover of Limburg province, the Netherlands.

## 1.5 Outline of the thesis

In order to address the research questions different components of ecosystem accounts and biodiversity accounts are studied for Limburg and outcomes are consequently applied to assess priority areas for conservation. The introduction (Chapter 1) and the synthesis and conclusion of this thesis (Chapter 6) have been written to connect Chapters 2 to 5 that have been written as independent research papers, developed in cooperation with one or more co-authors. These chapters can be read separately. In Chapter 2 spatial approaches to modelling biophysical flows of ESs are developed. To address the first research question, spatial ES models are developed for Limburg. These models assess biophysical ES flow accounts at three different spatial extents (single pixel, land-cover type and the province as a whole). Subsequently, monetary ecosystem accounts are developed in Chapter 3, building on the developed biophysical ES models. This chapter addresses the second research question. Methods aligned with SNA are applied to value annual flows of seven ESs and develop ecosystem accounting tables. In Chapter 4 spatial indicators that could be used for biodiversity accounts, are studied to address the third research question. Indicators with varying ecological complexity are assessed using different types of information necessary for biodiversity accounting. Chapter 5 addresses the fourth research question by applying information from the developed ecosystem accounts to a biodiversity conservation issue. The heuristic optimisation software Marxan is applied to develop spatial scenarios to expand Limburg's conservation network based on biodiversity and ESs. Chapter 6 provides answers to the research questions, a reflection on the research methods and findings and a final synthesis of key research findings.

# Chapter 2 - Developing spatial biophysical accounting for multiple ecosystem services


## Abstract

Ecosystem accounting is receiving increasing interest as a way to systematically monitor the conditions of ecosystems and the ecosystem services they provide. A critical element of ecosystem accounting is understanding spatially explicit flows of ecosystem services. We developed spatial biophysical models of seven ecosystem services in a cultural landscape (Limburg province, the Netherlands) in a way that is consistent with ecosystem accounting. We included hunting, drinking water extraction, crop production, fodder production, air quality regulation, carbon sequestration and recreational cycling. In addition, we examined how human inputs can be distinguished from ecosystem services, a critical element in ecosystem accounting. Model outcomes were used to develop an ecosystem accounting table in line with the System of Environmental-Economic Accounting – Experimental Ecosystem Accounting (SEEA-EEA) guidelines, in which contributions of land cover types to ecosystem service flows were recorded. Furthermore we developed spatial accounts for single statistical units. This study shows that for the case of Limburg spatial modelling for ecosystem accounting in line with SEEA-EEA is feasible. The paper also analyses and discusses key challenges that need to be addressed to develop a well-functioning system for ecosystem accounting.



Based on:

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## 2.1 Introduction

The importance of protecting ecosystems and the services they provide to sustain human livelihoods is increasingly recognised (MA, 2005; TEEB, 2010; United Nations, 2012). There is high demand from policy makers for sound information on ecosystem services (ESs) (Larigauderie et al., 2012). A crucial step in meeting the information needs of policy makers is measurement and monitoring of the current status and trends in the delivery of ES. While it is widely recognised that ES contribute to human well-being (MA, 2005), and supports economic activities in multiple ways (e.g. Barbier, 2007; Boyd, 2007; TEEB, 2010), they have not yet been systematically monitored in national accounts. National accounts comprise a system for measuring economic activity, and have been developed over the course of the last half century into a comprehensive statistical standard, that is now widely applied across the world (UN et al., 2009). Ecosystem accounting is a promising method to integrate ecosystems and ES into national accounts (Boyd and Banzhaf, 2007; Edens and Hein, 2013). A first guideline for ecosystem accounting was recently developed under auspices of the UN Statistics Commission: the System for Environmental Economic Accounts Experimental Ecosystem Accounting guidelines (SEEA-EEA) (UN et al., 2014a).

Ecosystem accounting measures and monitors the conditions of ecosystems, their capacity to provide services and the ES flows from the ecosystem to society. A key element in the development of methodologies for ecosystem accounting is understanding how ESs can be connected to economic activity, and how flows of ESs can be quantified at large spatial scales, with an accuracy sufficient for accounting purposes (Boyd and Banzhaf, 2007; Mäler et al., 2008; Edens and Hein, 2013). Ecosystem accounting takes a spatial approach towards analysing ecosystems and ESs. The SEEA-EEA guidelines recognise that ecosystems and ESs are spatially heterogeneous, and that this spatial variability needs to be captured in ecosystem accounting (UN et al., 2014a). Developing spatially explicit ecosystem accounts is thus a specific policy application of spatial ES modelling.

Spatial ES modelling is a research field which has progressed rapidly in recent years (e.g. Nelson et al., 2009; Raudsepp-Hearne et al., 2010; Willemen et al., 2010; Burkhard et al., 2012; Maes et al., 2012b; Schröter et al., 2014b; Serna-Chavez et al., 2014). It addresses a wide range of ESs at different spatial scales with a variety of services modelled with different spatial methods (Martínez-Harms and Balvanera, 2012; Crossman et al., 2013b; Nemec and Raudsepp-Hearne, 2013). For ecosystem accounting spatial modelling approaches that use quantified data could be used (e.g. Kareiva et al., 2011; Petz and van Oudenhoven, 2012; Sumarga and

Hein, 2014). ES mapping studies that rely on proxy indicators for ES (Eigenbrod et al., 2010), or on expert judgement (Seppelt et al., 2011; Burkhard et al., 2012) are less suitable for ecosystem accounting. Spatial modelling of ESs for ecosystem accounting calls for a definition of ESs that is aligned with the national accounting framework (UN et al., 2014a), measuring ES flows with quantifiable (spatial) indicators, high resolution, accurate output at large spatial scales (e.g., provinces, nations), and understanding the level of error involved.

The objective of this study is to assess how multiple ESs can be spatially modelled and analysed in a way that is consistent with ecosystem accounting, at a large spatial scale. In particular, we test if and how the spatial approach outlined in the SEEA-EEA for measuring ES flows from ecosystems to society can be applied at the scale of the Dutch province of Limburg. We test which models would be appropriate to model key ESs provided by ecosystems in this province, and discuss what the main challenges and bottlenecks are for further developing ecosystem accounting. We selected Limburg province because it is a data-rich environment, comprising a diversity of landscapes and generating a range of different ESs typical for North Western Europe. We analysed seven ESs: hunting, drinking water extraction, crop production, fodder production, air quality regulation, forest carbon sequestration and recreational cycling.

## 2.2. Conceptual framework and definition of ecosystem services

Current conceptualisations of the ES concept (cf. Haines-Young and Potschin (2010a); further refinements by van Oudenhoven et al. (2012) and van Zanten et al. (2014)) have described the emergence of an ESs as a “cascade” from ecosystem properties to ES values. In accounting, ESs are “the contributions of ecosystems to benefits used in economic and other human activity” (UN et al., 2014a). In this definition it is recognised that human contributions, in the form of labour and manufactured capital, are necessary for humans to benefit from many services (Boyd and Banzhaf, 2007; Haines-Young and Potschin, 2010b; TEEB, 2010; Bateman et al., 2011), and that the processed goods (e.g. milk, processed wood or bread) themselves are not the ES (Schröter et al., 2014a).

Disentangling human and ecosystem contributions in the generation of an ES is not straightforward. In line with van Oudenhoven et al. (2012) and Edens and Hein (2013) we argue that two types of human contributions can be distinguished, namely (historic and current) management of the ecosystem state and the extraction or use of the ES (Figure 2.1).The magnitude of these human contributions varies and depends on the respective ecosystem and ES, but is especially noticeable in cultural landscapes. The current ecosystem state is determined by a combination of ecological properties and human management which often has evolved over the course of centuries. For example, besides ecological properties, the current state of a cropland is determined by current management practices (fertilizer application, irrigation), as well as by the historical management choice to convert a natural ecosystem to cropland. Within an accounting context, these anthropogenic changes to ecosystem properties are possibly recorded in current conventional accounts (fertilizer) or, in the case of historic changes to ecosystems, have possibly been recorded in preceding accounting periods.

For humans to benefit from ESs a flow is necessary from the ecosystem to society. For most regulating ESs this flow can be fully attributed to the ecosystem, i.e. there is no or hardly any human contribution. For example, forests may sequester carbon without human intervention. For most provisioning and cultural ESs, however, a human contribution is necessary for society to benefit. This benefit emerges as a result of the contributions of both the ecosystem and humans, for instance in the form of extraction or other forms of active use (Figure 2.1, Bateman et al., 2011; Böhnke-Henrichs et al., 2013). Hence, in accounting there is a need to conceptually describe the contribution of the ecosystem for specific services. In this paper we propose the following. For provisioning services the benefit is a consumable or marketable good, such as harvested crops or logged timber, while the ES would be the standing crop prior to harvest, or the standing stock of trees that will be logged. For provisioning services a human contribution in the form of labour and manufactured capital is necessary to transform an ES into a benefit ('mobilisation' through investments, cf. Spangenberg et al., 2014). In the case of cultural ESs, a human contribution in the form of an activity is needed besides material input. For example, for the ES cycling recreation a cycling trip (time, bicycle) is required. The ES can be described as the provision of attractive landscapes that make the cycling trip enjoyable, while the benefit is the cycling trip itself. In Table 2.1 we conceptually explain the differences between the ES and the benefit for each ES that was modelled in this study. The notion of ESs as "contributions" has consequences for an ES assessment for cultural landscapes, in particular for the choice of biophysical indicators. In cultural landscapes, where ecosystems and ESs are the result of combined influences of natural processes and human management, contributions of ecosystems are hardly feasible to separate in a meaningful way given current data and knowledge restrictions.

When measuring ESs in biophysical terms, in some cases there is little difference between a suitable indicator for the ecosystem contribution and the benefit, as, for example, between the indicators tons of wheat standing in the field (the ES) and the harvested wheat (the benefit). The wheat example also shows that disentangling the ecosystem contribution is challenging and hardly feasible as human contributions (agricultural knowledge, fertilizer) have already influenced the absolute amount of the ES. Practical empirical endeavours of ecosystem accounting have to face information costs in indicator choice. Much available data on ESs indicates a benefit, which is why a 'realistic' choice of indicators (Figure 2.1) often does not allow for disentanglement. For more intangible ESs such as cycling recreation the contribution of the ecosystem (an attractive landscape) is even more challenging to measure than the benefit (the cycling trips). ES indicators in this study were preferably chosen to reflect the ecosystem contribution, by trying to measure a flow that is most directly related to the ecosystem (Table 2.1) (Schröter et al., 2012; Edens and Hein, 2013). However, in many cases this was not possible because data for ES indicators were not available. For those ESs an indicator which represents the benefit was chosen, as explained in section 2.3.3.

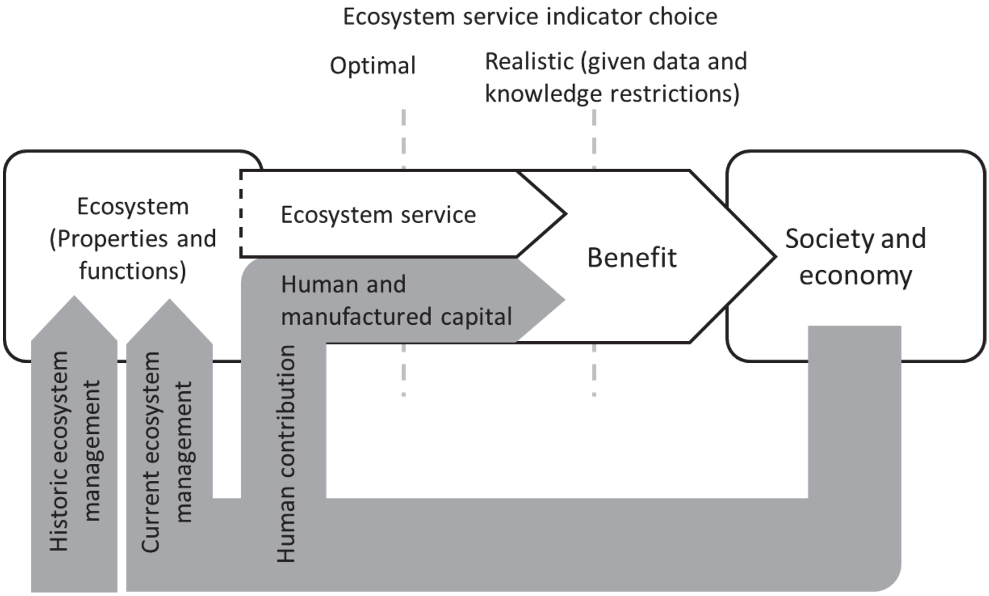


**Figure 2.1.** Framework for conceptualization of human contributions to the emergence of an ecosystem service. Both historic and current management influence ecosystem properties and functions, which in turn has an impact on the ecosystem service. Human and manufactured capital is often needed to realise the benefits that society and economy derives from ecosystems. Indicator choice in empirical ecosystem service assessments often reflects the benefit instead of the contribution of ecosystems to this benefit.

**Table 2.1** The modelled ESs, human management practices in ecosystems and the relation between ecosystem contribution and benefit.

| Ecosystem service name | Examples of human management of ecosystem | Ecosystem service | Benefit as used by humans | Ecosystem service indicator |
|---|---|---|---|---|
| **Hunting** | National parks, ecological corridors | Animals that are shot | Game meat | Game meat |
| **Drinking water extraction** | Groundwater protection zones, extraction zones | Extracted groundwater | Drinking water | Extracted groundwater |
| **Crop production** | Crop choice, fertilizer application, drainage and irrigation | Standing crop (at the time of harvest) | Harvested crop | Harvested crop |
| **Fodder Production** | Fertilizer application, drainage and irrigation | Standing grass (consumed by animals) | Milk, meat | Harvested or grazed fodder |
| **Air quality regulation** | Tree planting | $PM_{10}$ capture | Health benefits | Captured $PM_{10}$ |
| **Carbon sequestration** | Tree planting | Carbon sequestration | Reduced climate change | Carbon sequestered |
| **Recreational cycling** | Cycling paths | Scenic beauty along cycling paths | Cycling trips | Number of cycling trips |

## 2.3. Methods

### *2.3.1 Study area*

Limburg province is situated in the south-eastern part of the Netherlands, covering approximately 2200 km$^2$. It has a varied and fragmented cultural landscape, which has been managed for many centuries (Berendsen, 2005; Jongmans et al., 2013). Similar to many other regions in the Netherlands, most natural ecosystems have been converted and most areas are now highly managed, which has led to landscape fragmentation (Jongman, 2002). Competition for land is high between agriculture, nature and urban land covers (Vogelzang et al., 2010). The province is nationally renowned for the attractive hilly landscape in the southern part of the province. With 1.1 million inhabitants Limburg has a high population density (522 people km$^{-2}$ in 2012) (Statistics Netherlands, 2012).

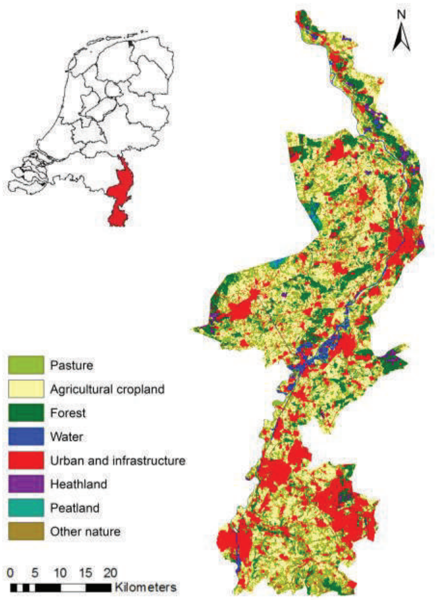


**Figure 2.2** Location and land cover of Limburg province. Source land cover data: Hazeu (2009).

### *2.3.2 Ecosystem accounting units*

We used three types of spatial accounting units that are aligned with those proposed in the SEEA-EEA (UN et al., 2014a). The largest unit type was the ecosystem accounting unit (EAU), which was delineated by administrative boundaries of Limburg province. The second unit type was the land cover/ecosystem unit (LCEU). Eight types of LCEUs, aligned with the main land cover class types, were distinguished for the analysis of ES flows (Figure 2.2). These LCEUs were compiled from the specific land cover classes of the Dutch land cover map LGN6 (Landelijk Grondgebruiksbestand Nederland version 6) (Hazeu, 2009). The category *pastures* includes agricultural grasslands. *Cropland* includes all arable crops, as well as horticulture, nurseries, bulb fields and orchards. *Forest* includes all non-urban forested areas and *water* all open water bodies. The category *urban and infrastructure* includes all urban areas, including green areas, buildings in rural areas, glasshouses, large roads and railways. *Heathland* includes only heath and *peatland* includes only peat. The category *other nature* includes natural grasslands, reed vegetation, swamp vegetation, and drift sands. The smallest unit type was a basic spatial unit (BSU). BSUs are grid cells (25 x 25 m grain) that together make up a LCEU. A BSU is used to assess local variation in ES flows.

### *2.3.3 Modelled ecosystem services*

In this study seven ESs have been modelled, chosen to reflect the diversity of services based on the provisioning, regulating and cultural categories from the frameworks of The Economics of Ecosystems and Biodiversity (TEEB) and the Millennium Ecosystem Assessment (MA) (MA, 2003; TEEB, 2010). The chosen ESs include four provisioning services (crop production, fodder production, drinking water extraction and hunting), two regulating services (air quality regulation and carbon sequestration) and one cultural service (recreational cycling) (Table 2.2). The ESs were chosen based on expert judgement and feedback from provincial policy makers, in combination with the criterion of data availability, which is why the ES list is not exhaustive. However, it does cover important economic aspects (agricultural services), cultural aspects (cycling as a main form of recreation, and to a lesser extent hunting) and human health aspects (air quality regulation and clean drinking water) for the province as well as an ES of international interest (carbon sequestration). With this subset of ES we tested a variety of spatial modelling methods for which we use different types of datasets.

A spatial model was built for every ES. Spatial modelling was done using ESRI ArcGIS 10 and Geospatial Modelling Environment (version 0.7.1.0) software. The ES models were generally developed at a fine resolution using the LGN6 land cover map (Hazeu, 2009). The year 2010 was used as base year for the ES models, unless indicated otherwise in the model descriptions.

**Table 2.2** Modelled ES and information on input data.

| Ecosystem service | Dataset | Spatial | Data type | Source |
|---|---|---|---|---|
| **Hunting** | LGN6 land cover | Yes | Raster (25 m grain) | Hazeu (2009) |
| | Hunting districts | Yes | Polygon | Faunabeheereenheid Limburg (2010) |
| | Roe deer hunted | No | Provincial statistics | Faunabeheereenheid Limburg (2012a) |
| | Wild boar hunted | Yes | Points | Faunabeheereenheid Limburg (2011) |
| | Wild boar hunted in national park | No | Park statistics | Faunabeheereenheid Limburg (2012b) |
| **Drinking water extraction** | LGN6 land cover | Yes | Raster (25 m grain) | Hazeu (2009) |
| | Groundwater protection zones | Yes | Polygon | Provincie Limburg (2010a) |
| | Groundwater extraction 2010 | No | Provincial statistics | Provincie Limburg (2010b) |
| **Crop production** | LGN6 land cover | Yes | Raster (25 m grain) | Hazeu (2009) |
| | Soil map | Yes | Raster (50 m grain) | Alterra (2006a) |
| | Annual crop yield | No | National statistics | LEI and Statistics Netherlands (2011) |
| **Fodder production** | LGN6 land cover | Yes | Raster (25 m grain) | Hazeu (2009) |
| | Soil map | Yes | Raster (50 m grain) | Alterra (2006a) |
| | Groundwater table | Yes | Polygon | Alterra (2006b) |
| | Cattle numbers | Yes | Points | Naeff et al. (2011) |
| | Fodder yield | No | Empirical research | Aarts et al. (2005) |
| **Air quality regulation** | LGN6 land cover | Yes | Raster (25 m grain) | Hazeu (2009) |
| | PM10 ambient concentration 2011 | Yes | Raster (1 km grain) | Velders et al. (2012) |
| **Carbon sequestration** | LGN6 land cover | Yes | Raster (25 m grain) | Hazeu (2009) |
| | Gross primary production | Yes | Raster (1 km grain) | NASA LP DAAC (2012) |
| **Recreational cycling** | LGN6 land cover | Yes | Raster (25 m grain) | Hazeu (2009) |
| | Cycling paths | Yes | Line | Fietsersbond (2012) |
| | Population statistics | Yes | Polygon | Statistics Netherlands and Kadaster (2009) |

### *Hunting*

The ES hunting was modelled for 43 hunting districts in Limburg based on two game species: wild boar (*Sus scrofa*) and European roe deer (*Capreolus capreolus*). For this ES we used consumable meat (kg $km^{-2}$ $yr^{-1}$) from hunted game as an indicator. For modelling hunted wild boar a spatially explicit dataset was used (Faunabeheereenheid Limburg, 2011), except for National Park De Meinweg, for which aggregated annual statistics were available for hunting season 2010-2011 (Faunabeheereenheid Limburg, 2012b). For roe deer, statistics were available per hunting district for 2010 (Faunabeheereenheid Limburg, 2012a). The consumable meat is equal to the dressed carcass weights, which was assumed to be 0.75 of the body weight for wild boar (Grubešić et al., 2011). The mean weight of all wild boar (40 kg) was assumed for wild boar shot inside De Meinweg national park, because data on dressed weight was not available. For European roe deer an estimate of 13 kg was used (Faunabeheereenheid Groningen, 2012). The weight of the game meat was calculated per hunting district and averaged over the area of the district, excluding urban areas, infrastructure and water bodies, which were extracted using the LGN6 land cover map.

### *Drinking water extraction*

Groundwater is an important source of drinking water in Limburg, constituting about 75% of the total drinking water extraction (Waterleiding Maatschappij Limburg, 2013). Although the availability of groundwater for extraction can be attributed to (abiotic) geological processes for a large part, biotic processes are also influential. Vegetation and soil fauna affect soil properties, such as porosity, which influences the infiltration of groundwater. Also, vegetation can have purifying effects on groundwater (e.g. Elowson, 1999). Because of these biotic influences groundwater is considered to be an ES. In this study the extracted groundwater ($m^3$ $ha^{-1}$ $yr^{-1}$) is used as indicator for this ES. Only drinking water extraction from shallow groundwater (unconfined aquifers) was modelled. Groundwater from unconfined aquifers is extracted for the production of drinking water in 10 groundwater protection zones throughout the province, ranging from 394 to 2,386 ha in size. These ground water protection zones are located around the extraction points and were assumed to be the areas to which the ES can be attributed. The groundwater protection zones can be considered as storage areas of drinking water that has infiltrated locally or travelled there from other areas. It was assumed that all areas of the protection zones contributed equally to the storage of drinking water and therefore also to the extracted drinking water, regardless of the assigned land cover type. The extracted volumes of water (Provincie Limburg, 2010b) were divided evenly over the ground water protection zones to calculate the ES. Areas of two groundwater protection zones extended across the border into Germany. The contribution to groundwater extraction from those parts of the protection zones were excluded from the model, because that contribution should be attributed to ecosystems outside Limburg.

### *Crop production*

A third of the area of Limburg is used for crop production. Crop production, especially in intensive agricultural areas, is to a large extent determined by human input such as specific plant breeds, fertilizers, ground water management and insecticides. Nevertheless, the ecosystem makes a valuable contribution in the form of natural processes, such as soil biodiversity and nutrient cycling. Ideally these natural processes should be quantified to determine the ecosystem contribution. However, disentangling these processes from human contributions is difficult, especially since human use has determined the state of the ecosystem for centuries. Due to this complexity we used crop production as an indicator for the ES, noting that this does not accurately reflect the ecosystem contribution.

Spatial modelling of agricultural crops was done based on spatial land cover data (Hazeu, 2009) and national statistics on the annual average agricultural crop yield for 2010 (LEI and Statistics Netherlands, 2011). The land cover dataset contained data on four groups of crops: cereals, potatoes, sugar beets, and other crops. For these groups aggregated statistics were used for the two agricultural regions of the province (north and south); and statistics for potatoes were divided according to agricultural region and according to soil type (clay soils and sandy soils).

### *Fodder production*

In Limburg cattle rearing for dairy and meat is an important economic activity (Statistics Netherlands, 2013a). We distinguish the cattle that produce the meat and dairy as the benefit and the production of fodder by pastures and maize as the contribution of the ecosystem (the ES), consistent with Edens and Hein (2013). In many other studies the livestock production for dairy and meat is modelled as the ES (e.g. Naidoo et al., 2008; Maes et al., 2011; Petz and van Oudenhoven, 2012), however, fodder is more closely connected to the ecosystem than meat and dairy products. Therefore annual production of dry matter (dm) from pastures and maize was taken as the ES flow indicator.

The ES model was developed by adding two components, dm from maize and dm from pastures. For maize national statistics for the average yield in 2010 was used [36 t $ha^{-1}$ $yr^{-1}$ (Statistics Netherlands, 2013c)] for the entire province and a dm content of 30% was assumed. The calculations of fodder from pasture were based on the findings of Aarts et al. (2005), where average fodder yield was measured for four soil categories and four milk production intensity categories of dairy cows (l $ha^{-1}$), creating 16 fodder yield classes. These fodder yield classes were used in our model. The four soil categories that were distinguished are clay, peat, wet sand and dry sand. A soil map with these four categories was created. The four milk production categories were (1) less than 10,000 l $ha^{-1}$, (2) 10,000-14,000 l $ha^{-1}$, (3) 14,000-18,000 l $ha^{-1}$ and (4) more than 18,000 l $ha^{-1}$. Each municipality was classified into one of the four milk production intensity categories, creating a milk production intensity map for Limburg.

The milk production intensity map was created based on milk production figures for dairy cows per municipality. To incorporate non-dairy cows into the model, milk production equivalents were calculated. Calculations were based on the livestock units (LSU), where a dairy cow is 1 LSU. The average LSU for non-dairy cows was calculated based on all non-dairy cattle categories (Naeff et al., 2011), being 0.67 LSU. The total LSU for each municipality was calculated by adding that of dairy cows and non-dairy cows together. The total LSU per municipality was multiplied with the average annual milk production of a dairy cow (8000 l $yr^{-1}$ (LEI and Statistics Netherlands, 2012)) to calculate annual milk production equivalents. Average milk production intensities were calculated per municipality based on the annual milk production equivalents and total area of grassland per municipality (Naeff et al., 2011). Therefore, to calculate the milk production intensity for a municipality the following equation was used:

$$C_m = \frac{(d_m + 0.67 * n_m) * 8000}{A_m} \qquad (1)$$

where $C_m$ is the average milk production intensity in municipality $m$, $d_m$ is the number of dairy cows in $m$, $n_m$ is the number of non-dairy cows in $m$, and $A_m$ is the total ha of pasture in $m$. Using this equation each municipality was categorized into one of the four milk production intensity categories. A fodder production map for pastures with 16 fodder yield classes was created by combining the soil map and the milk production intensity map. For the final fodder production map, fodder production from pastures and from maize were combined.

### *Air quality regulation*

Air pollution has detrimental effects on multiple aspects of human health (Künzli et al., 2000), with a range of pollutants affecting air quality. Particulate matter ($PM_{10}$) is one of the best documented pollutants in the Netherlands (Velders et al., 2012), and has therefore been used as an indicator in this study. $PM_{10}$ is detrimental to human health, also at low concentrations (Künzli et al., 2000; Pelucchi et al., 2009). The capture of $PM_{10}$ by vegetation reduces atmospheric concentrations, and indirectly decreases health risks that result from direct exposure (Beckett et al., 2000). In our model the capture of $PM_{10}$ has been considered as the ES. The contribution of ecosystems to air quality regulation was measured as the vertical capture of ($PM_{10}$) by vegetation. $PM_{10}$ capture by vegetation (μg $m^{-2}$) was calculated according to the following function (Powe and Willis, 2004):

$$PM_{10}\ capture = A * V_d * t * C \qquad (2)$$

where $A$ is area, $V_d$ is vertical deposition velocity for specific land covers, $t$ is the time step (one year), and $C$ is ambient $PM_{10}$ concentration, which has been calculated based on the Dutch national ambient concentration map for 2011. This map depicts average daily ambient $PM_{10}$ concentrations (μg/$m^3$) at 1 $km^2$ resolution (Velders et al., 2012). Values for $V_d$ were adapted from Powe and Willis (2004), and are 0.0080 m/s for needle-leaved forest, 0.0032 m/s for broad-leaved forest, 0.0010 m/s for heath, peatland, grassland, cropland and other nature, and 0 m/s for water and urban and infrastructure land covers.

### *Carbon sequestration*

Carbon sequestration can be largely attributed to the ecosystem. Human management does have influence it indirectly, through for example the type of vegetation planted. However, here the carbon sequestered (tC $ha^{-1}$ $yr^{-1}$) was considered to be the ES. Carbon sequestration was calculated based on a look-up table approach, which assigns quantities of ES flows to land cover units. Using classes from the LGN6 land cover map, eight land cover categories were defined in the analysis. The categorisation is linked to the land cover types in the academic literature used, and therefore differs from the classification applied for the LCEUs. Values for carbon sequestration per land cover type and used references can be found in Table 2.3.

**Table 2.3** Look-up table for carbon sequestration in Limburg.

| Land cover category | Carbon sequestration ($tC\ ha^{-1}\ yr^{-1}$) | Reference | LGN6 land cover classes included |
|---|---|---|---|
| **Grassland** | 0.18 | Janssens et al. (2005) | All types of grassland and heathland |
| **Cropland** | 0 | Kuikman et al. (2003) | All arable and horticultural cropland |
| **Permanent cropland** | 0.29 | Schulp et al. (2008) | Orchards |
| **Forest** | 1.45 | Nabuurs et al. (2008) | All forest types |
| **Peatland** | 0.20 | Janssens et al. (2005) | Peatland and wetland vegetation |
| **Built-up areas** | 0 | Schulp et al. (2008) | Urban areas, buildings in rural areas, infrastructure, glasshouses |
| **Sand** | 0 | Schulp et al. (2008) | Sand dunes and sandbanks |
| **Water bodies** | 0 | Coenen et al. (2012) | All water bodies |

### *Cycling recreation*

Limburg is known throughout the Netherlands for its nature recreation possibilities. Together with hiking, cycling is the most popular nature recreation activity (Goossen, 2009). Annually 10 million recreational cycling trips of at least one hour are made Limburg (Stichting Landelijk Fietsplatform, 2009; NBTC-NIPO Research, 2012b, a; Stichting Landelijk Fietsplatform, 2013). This number excludes cycle racing and mountain biking, for which sufficient data lacked. Modelling these activities requires a different approach and we considered this to be out of scope of our paper. Table 2.4 gives an overview of the percentages of trip lengths of recreational cyclists. In our model trips longer than 50 km (3% of all trips) or with unknown length (6%) were not taken into account. All trips were assumed to take place only inside the province.

**Table 2.4** Percentages of cyclists taking recreational cycling trips of different lengths (NBTC-NIPO Research, 2012a; Stichting Landelijk Fietsplatform, 2013).

| **Length of cycling trips** | **Percentage of trips (%)** |
|---|---|
| 0 - 5 km | 11 |
| 6 - 10 km | 18 |
| 11 - 20 km | 32 |
| 21 - 50 km | 30 |
| Total | 91 |

A database for the national cycle path network (Fietsersbond, 2012) was used to develop an allocation model. The model combines variables for cycling path density, landscape aesthetics and population size to estimate the spatial distribution of recreational cycling in the province. The database contains information on the length of cycling paths, surrounding land cover and the attractiveness of a path, but no quantitative data on use frequency.

Cycling path density was calculated for each hectare by calculating the length of path per ha (m $ha^{-1}$). Information from the database (Fietsersbond, 2012) on the surrounding land cover along paths and a qualitative score for attractiveness of the paths and surroundings, given by users of the cycling paths, were used. In the database attractiveness scores were only assigned to 69% of cycling paths in the province. To estimate attractiveness of all cycling paths a connection between attractiveness scores and land cover type was made. The attractiveness of cycling paths was scored on a five point scale (Fietsersbond, 2012), with scores 4 and 5 being ‘attractive’ and ‘very attractive’. These two categories were used to derive the percentage of cyclists that find certain land covers attractive. Based on the percentage of people that found a certain land cover attractive an attractiveness factor for the five land cover categories was derived from the cycling database (Table 2.5). The least attractive land cover type (built-up without green areas) was given a factor 1. Other land cover types were given an attractiveness score, relative to the least attractive land cover type. The attractiveness factor was given to the corresponding land covers from the LGN6 dataset. For each hectare which contains cycling paths the attractiveness factors of the different land covers were averaged out, to obtain an average attractiveness score per ha. This was multiplied with the cycling path density to give each grid cell a single value which reflects both accessibility and attractiveness (A&A score). These A&A scores were later used for the final allocation of cycling trips throughout the province.

**Table 2.5** Land covers, the percentage of recreational cyclists that find them attractive, and their relative attractiveness factor.

| Land cover | Percentage (%) | Attractiveness factor |
|---|---:|---:|
| Built-up (no green areas) | 7.0 | 1.0 |
| Built-up (many green areas) | 25.9 | 3.7 |
| Agricultural land | 52.0 | 7.4 |
| Nature (non-forest) | 74.6 | 10.6 |
| Forest | 87.1 | 12.4 |

Another factor determining the allocation of cycling trips was the spatial distribution of the population. For this spatial population statistics for 193 districts were used (Statistics Netherlands and Kadaster, 2009). Recreational cycling trips were spatially modelled according to these districts. The 10 million cycling trips were distributed equally over all inhabitants, resulting in approximately 9 cycling trips per person per year. Measured from the centre of each district, rings with a radius of 2.5, 5, 10 and 25 km were created and cycling trips were allocated according to the number of trips passing through these rings. For example, all modelled trips (91%) passed through or stayed within the 2.5 km ring and 80% passed through or stayed within the 5 km ring. Within each ring recreational cycling trips were allocated according to the A&A score, as a fraction of the total score within each ring. For example, if a ring contained a total A&A score of 1000 and a single BSU had an A&A score of 10, 1% of all cycling trips within this ring would be allocated to that specific BSU.

### *2.3.4 Accounting for ecosystem services*

The model outcomes were used to set up basic ecosystem accounting tables, for the three types of accounting units (EAU, LCEU and BSU). Biophysical accounts were created for nine individual BSUs, as an example for detailed ecosystem accounts that can monitor spatial variability of ESs at high resolution. For each example BSU a separate account was created, in which land cover was determined and quantities of the seven modelled ESs were calculated. Furthermore, at a provincial level the model outcomes were used to account for the quantities of ESs provided by each LCEU using an overlay analysis in ArcGIS, as well as for the province as a whole (EAU account). For this accounting table, the total annual flows, means and standard deviations (SD) were calculated for each ES.

## 2.4. Results

### *2.4.1 Spatial ecosystem service models*

Figure 2.3 shows the spatial distribution of the annual flows of the modelled ESs in Limburg province. The spatial models show substantial spatial variation of the different ES flows across the study area. The differences in the spatial resolutions of the models can be explained based on four different types of models that were used.

For the first model type administrative boundaries were used to allocate statistical data. We quantified hunting and drinking water extraction using this approach. Hunting districts were used to delineate the service (Figure 2.3a), resulting in a limited resolution and therefore limited spatial variability. Similarly, drinking water extraction is limited to the groundwater protection zones, covering a small part of the province (Figure 2.3b). Second, three ESs were derived from land cover types using look-up table approaches: crop production, fodder production and carbon sequestration (Figure 2.3c, 2.3d and 2.3f). The third model type couples environmental conditions to land cover types. This model was used for quantifying air quality regulation (Figure 2.3e). The result of this approach is that the model output roughly follows the spatial distribution of land cover types, while the distribution of ES quantities relies additionally on environmental input (in this case ambient $PM_{10}$ concentration). The final model type can be considered as a socio-ecological model, where the resolution and spatial distribution of ES quantities depend on both social data and land cover data. This model was used to quantify recreational cycling (Figure 2.3g).

Hunting is highest towards the eastern borders of the province, in districts with relatively large forest areas which serve as a habitat. Drinking water provision is highest in the small extraction area in the southeast of the province. Crop production shows large spatial variation depending on the type of crop produced. It is highest in the southern part of the province due to the fertile loess soils found there. Fodder production has large spatial variation throughout the province. Air quality regulation is highest in areas with large forests and lowest in urban areas. Carbon sequestration is mostly concentrated in forest areas, because this land cover type has a substantially higher sequestration rate than all other land cover types. Cycling recreation is highest in the more densely populated southern part of the province. Quantities in urban areas are generally lower than in other land covers in densely populated areas. Highest values for cycling recreation are found in non-urban land covers directly adjacent to large cities.

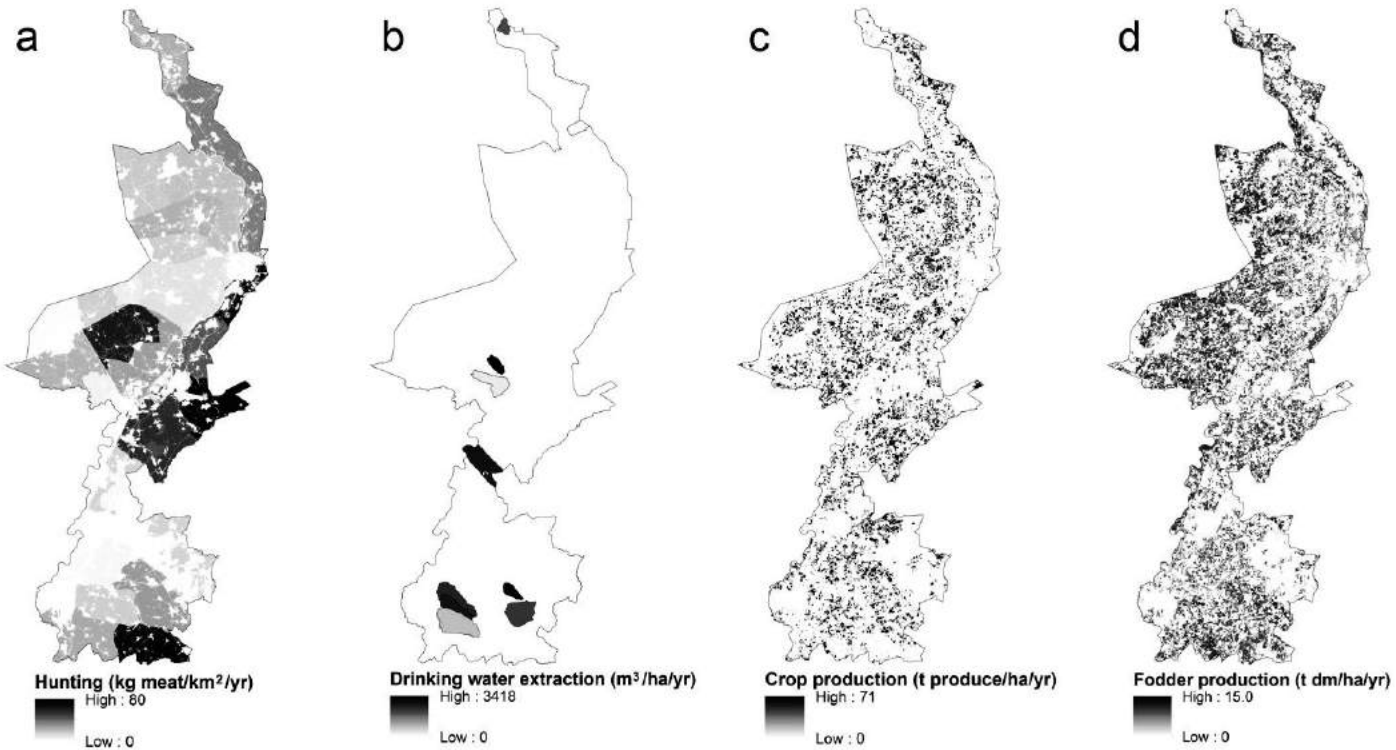


**Figure 2.3** Model output of ecosystem service flows for 2010 in Limburg. For data sources see Table 2.2.

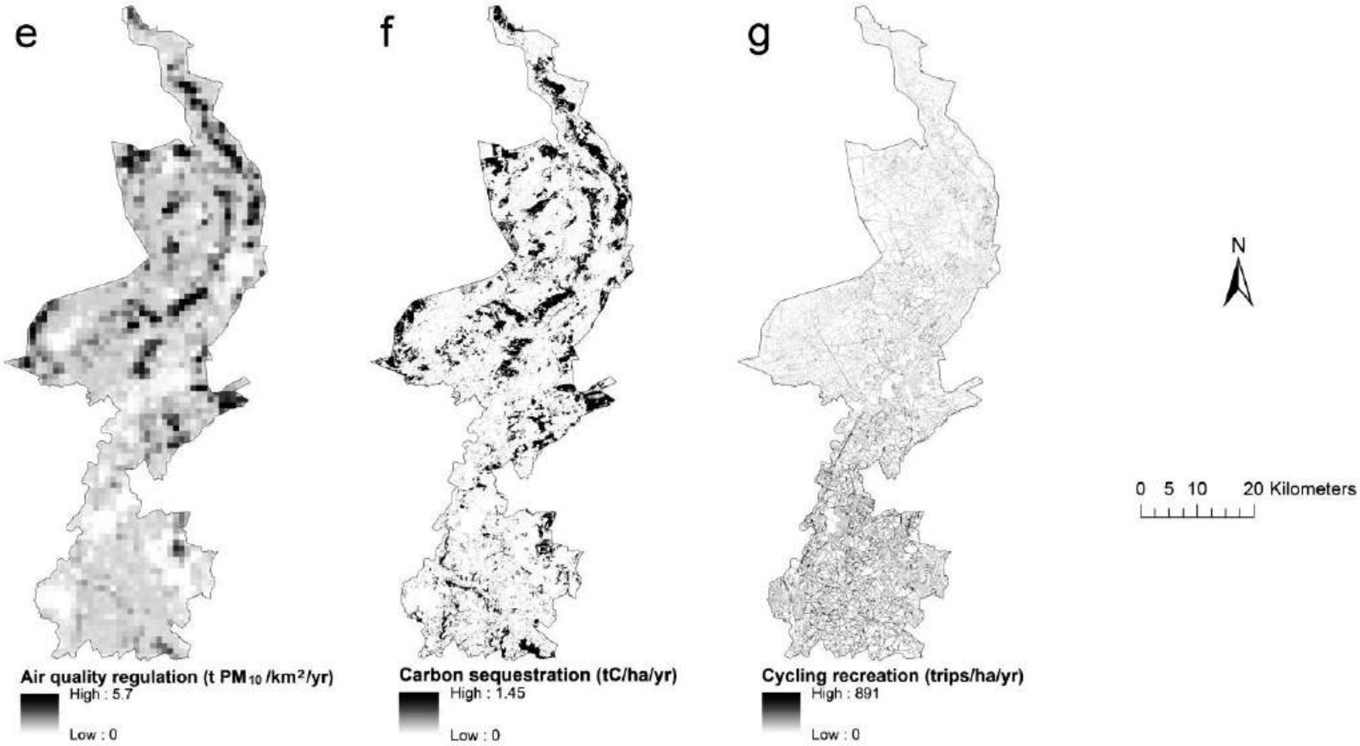


Figure 2.3 (continued)

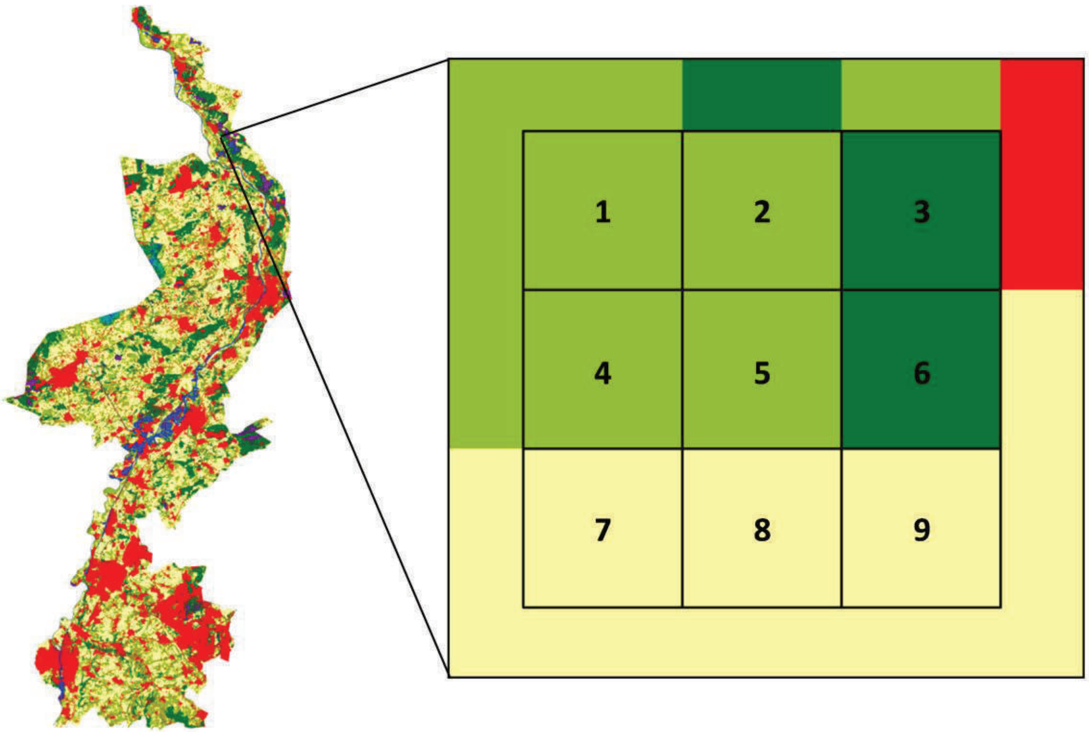


**Figure 2.4** The nine selected BSUs in northern Limburg. The numbers 1 through 9 correspond with the BSU numbers in Table 2.6.

**Table 2.6** An ecosystem accounting table for the nine example BSUs (25 m grain) presented in Figure 2.4.

| Ecosystem service | Unit | Basic spatial unit number | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 |
| **Hunting** | *kg/yr* | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 |
| **Drinking water extraction** | *m3/yr* | 58 | 58 | 58 | 58 | 58 | 58 | 58 | 58 | 58 |
| **Crop production** | *kg/yr* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Fodder production** | *kg/yr* | 935 | 935 | 0 | 935 | 935 | 0 | 681 | 0 | 0 |
| **Air quality regulation** | *kg/yr* | 0.6 | 2.3 | 2.3 | 0.6 | 2.3 | 2.3 | 0.6 | 2.3 | 2.3 |
| **Carbon sequestration** | *kg/yr* | 11 | 11 | 91 | 11 | 11 | 91 | 0 | 0 | 0 |
| **Recreational cycling** | *trips/yr* | 0 | 4 | 4 | 0 | 4 | 4 | 4 | 1 | 1 |
| **Land cover** | - | Grass land | Grass land | Forest | Grass land | Grass land | Forest | Crop land | Crop land | Crop land |

**Table 2.7** Total and mean annual ecosystem service flow per LCEU. Standard deviations (SD) are indicated for mean values over all BSUs. Missing values indicate that ecosystem services was not modelled in that LCEU.

| LCEU | Ecosystem service | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Hunting | | Drinking water extraction | | Crop production | | Fodder production | |
| | Total *kg meat* | Mean (SD) *kg meat $km^{-2}$ $yr^{-1}$* | Total *$10^3$ $m^3$ water* | Mean (SD) *$m^3$ water $ha^{-1}$ $yr^{-1}$* | Total *$10^6$ kg produce* | Mean (SD) *kg produce $ha^{-1}$ $yr^{-1}$* | Total *$10^6$ kg dm* | Mean (SD) *kg dm $ha^{-1}$ $yr^{-1}$* |
| **Pasture** | 9,100 | 21 *(17)* | 7,026 | 2,389 *(694)* | - | - | 533 | 12,019 *(1,571)* |
| **Cropland** | 14,732 | 20 *(17)* | 11,227 | 2,329 *(744)* | 1,868 | 41,804 *(24,339)* | 251 | 10,894 *(0)* |
| **Forest** | 8,100 | 23 *(20)* | 3,117 | 2,119 *(811)* | - | - | - | - |
| **Water** | - | - | 478 | 1,369 *(621)* | - | - | - | - |
| **Urban** | - | - | 4,071 | 2,298 *(788)* | - | - | - | - |
| **Heath** | 678 | 32 *(25)* | 214 | 1,262 *(614)* | - | - | - | - |
| **Peat** | 70 | 13 *(3)* | - | - | - | - | - | - |
| **Other nature** | 1,513 | 25 *(20)* | 862 | 2,248 *(750)* | - | - | - | - |
| **Provincial total** | 34,193 | | 26,995 | | 1,868 | | 784 | |

Table 2.7 (continued)

| LCEU | Ecosystem service | | | | | |
|---|---|---|---|---|---|---|
| | Air quality regulation | | Carbon sequestration | | Recreational cycling | |
| | Total *$10^3$ kg $PM_{10}$* | Mean (SD) *kg $PM_{10}$ $km^{-2}$ $yr^{-1}$* | Total *$10^3$ kg carbon* | Mean (SD) *$10^3$ kg C $ha^{-1}$ $yr^{-1}$* | Total *$10^3$ trips* | Mean (SD) *trips $ha^{-1}$ $yr^{-1}$* |
| **Pasture** | 404 | 909 *(528)* | 8,019 | 0.18 *(0)* | 1,863 | 103 *(77)* |
| **Cropland** | 717 | 956 *(535)* | 273 | 0.00 *(0.03)* | 2,611 | 98 *(72)* |
| **Forest** | 700 | 2,001 *(1,228)* | 50,664 | 1.45 *(0)* | 1,565 | 128 *(95)* |
| **Water** | 40 | 613 *(558)* | 0 | 0.00 *(0)* | 139 | 109 *(91)* |
| **Urban** | 272 | 535 *(546)* | 875 | 0.02 *(0.05)* | 2,690 | 70 *(56)* |
| **Heath** | 45 | 2,056 *(1,116)* | 393 | 0.18 *(0.03)* | 30 | 84 *(61)* |
| **Peat** | 7 | 968 *(347)* | 149 | 0.18 *(0)* | 3 | 84 *(43)* |
| **Other nature** | 69 | 1,153 *(705)* | 1,056 | 0.20 *(0)* | 220 | 127 *(95)* |
| **Provincial total** | 2,254 | | 61,429 | | 9,122 | |

### *2.4.2 Ecosystem accounting for Basic Spatial Units*

Nine adjacent BSUs with a variety of land covers were selected as examples for setting up detailed spatial ecosystem accounts (Figure 2.4). The separate accounts for each BSU are shown in Table 2.6. This analysis shows that at a very local scale there can be considerable variations in number of ES available and also the quantity in which they are available. It shows that even between adjacent BSUs from the same LCEU, ES flows differ. This can be explained by the spatial variation in input variables of the different ES models, such as soil type, groundwater tables, landscape attractiveness and ambient $PM_{10}$ concentration. Spatial ecosystem accounts could be created for all BSUs within the province in order to monitor changes in ES flows and land cover over time.

### *2.4.3 Ecosystem accounting for Land Cover/Ecosystem Units*

LCEUs that have the largest contribution to the total annual flow of an ES do not necessarily have the highest mean annual flow (Table 2.7). While the total annual ES flow is generally lowest in the more natural LCEUs with a smaller extent (*heath*, *peat* and *other nature*), the mean ES flow from these LCEUs is highest for multiple ESs. *Heath* has the highest mean annual flow for hunting and air quality regulation. *Other nature* has one of the highest mean annual flows for cycling recreation. *Forest* has high mean as well as total values for the regulating and cultural services. For drinking water provision the less natural LCEUs have the highest mean annual flows (*pasture, cropland* and *urban and infrastructure* respectively). Mean annual flows for crop production and fodder production can only be calculated for *cropland* and *pasture* and *cropland* respectively, because other LCEUs do not contribute to these ESs. SDs were relatively high for most modelled ES. The presented SD reflects the spatial variation of BSUs. The SD is low for ESs that use aggregated statistics as input data.

## 2.5. Discussion

### *2.5.1 Ecosystem services in cultural landscapes*

The definition of ESs as stated in the SEEA-EEA makes a clear distinction between ESs and benefits, recognizing that, apart from ecosystem contributions, human contributions are often involved in deriving benefits from ecosystems (UN et al., 2014a). We argue that in strongly modified cultural landscapes such as Limburg and many landscapes of Europe it is challenging and not realistic to completely disentangle all human and ecosystem contributions, given current data and knowledge limitations. Especially management of the ecosystem can hardly be separated from ecological properties and functions. Nearly all ecosystems in Limburg are anthropogenically influenced; agricultural lands have been created out of forested areas and have themselves been modified to enhance production (e.g. by installing drainage systems). Forests have been modified for timber harvesting, and the populations of large mammals are managed. Since an ecosystem is often modified by people, ESs cannot be related to natural processes only (as suggested in Boyd and Banzhaf, 2007), and ecosystem accounting needs to be further developed on the premise that ecosystems in cultural landscapes are the resultant of targeted, as well as unintentional human modifications of once natural systems. In less intensively managed systems the ecosystem contribution and human contribution are more straightforward to disentangle. For example, in Telemark county, Norway, sheep are released to graze in natural areas (Schröter et al., 2014b). This system requires little human involvement, and therefore for fodder production there are very few processes that need to be disentangled. The benefit can be fully attributed to the ecosystem, as opposed to the highly managed systems in Limburg.

ES are measured at the last point in space and time where ecological processes play a significant role (Schröter et al., 2012). This would mean that extraction of matter (in the case of provisioning services) constitutes a boundary at which one can account. For crop production, for example, the last point where ecological processes play a significant role is in the field, prior to harvesting. At the moment the crops are harvested, they enter a production chain that is part of a socio-economic system, and ecosystem processes do not contribute significantly anymore. The interpretation of such an “accounting boundary” should result in a measurable indicator, and is internally consistent with the way other provisioning services are included in ecosystem accounting.

Defining the last point where ecological processes play a significant role is not always easy, and caution is needed. As discussed by Edens and Hein (2013), human influence in livestock rearing is very high. Therefore, we argue that the last significant contribution of ecological processes occurs in the production of fodder. Another example where the boundary is vague, is hunting. We defined the last ecological contribution as the game at the moment it is shot. Since the human influence on the foraging and health of game is much smaller compared to livestock, we argue that in this case the live game can be seen as the last significant contribution of ecological processes. Nevertheless, these examples show that the last significant ecological contribution can be subject to debate. ESs and their indicators need to be well defined if they are to be incorporated in ecosystem accounting.

### *2.5.2 Challenges of and uncertainties in spatial ecosystem service modelling for accounting*

The specific requirements we outlined for spatial ES models in the context of ecosystem accounting included a specific definition of ESs for accounting, using quantifiable spatial indicators, high resolution models, accurate output at large spatial scales and an understanding of the level of uncertainty involved. More generally, also an accurate understanding of the ecological conditions and the use systems are necessary. These requirements were largely met by the developed models. The 25 m grain we used for the BSUs proved feasible for accounting at the scale of Limburg. Also, multiple ESs flows have been modelled at a resolution that is representative for the variation in land cover. However, the uncertainty of the developed models deserves more attention. We developed an understanding of the uncertainties underlying the models, but were unable to validate our models, due to lack of suitable data. The lack of validated models is a recurring issue in many ES assessments (Seppelt et al., 2011). On the other hand, there are examples that show that validation techniques for ES models are available (e.g. Schulp et al., 2014b; Sumarga and Hein, 2014). Recurring uncertainties in ES mapping studies are generated by combining different types of spatial and non-spatial data, data aggregation and scaling, and the chosen indicators (Martínez-Harms and Balvanera, 2012; Crossman et al., 2013b; Nemec and Raudsepp-Hearne, 2013). These issues also caused uncertainties in our spatial models, which we will discuss in this section.

The accuracy of the developed models varied, depending on the available data. Many types of input data were used in the models. Spatial ES models require data with a degree of spatial explicitness, aggregated at a lower level than the unit of analysis, i.e. the study area. A consequence is that this limits the data choices (Nemec and Raudsepp-Hearne, 2013). However, much of the data related to ESs is not spatially explicit, and models are often built using a combination of spatial and non-spatial information (e.g. Chen et al., 2011; Petz and van Oudenhoven, 2012), ranging from look-up tables, to statistical datasets, satellite data or field measurements. Combining different data types, with different degrees of spatial explicitness and spatial variation, increases errors in the models, which cannot easily be quantified. It should be noted that the LGN6 land cover map that was used in the development and analysis of multiple models has inherent uncertainties and inaccuracies (Hazeu et al., 2010; Schulp and Alkemade, 2011), which affected model outcomes. For example, grassland statistics (Naeff et al., 2011) indicate the total area of grassland to be 25% smaller than in LGN6. These differences are the result of inaccuracies of remote sensing based maps (Schulp and Alkemade, 2011), as well as different interpretations of what constitutes a grassland. Therefore, besides accurate spatial models, accurate and specific definitions of land cover classes are also essential in ecosystem accounting.

Static look-up table approaches such as applied in the carbon sequestration model might not always be suitable for ecosystem accounting. Such methods do not provide information on spatial variability within land cover classes and can therefore only inform on changes between land cover classes over time. Moreover, different studies indicate different values of carbon sequestration for land cover types. For example, for maize Dutch studies show both relatively high sequestration (Hanegraaf et al., 2009; Reijneveld et al., 2009) and emissions (Lesschen et al., 2012), as well as a carbon neutral value (Kuikman et al., 2003), which we chose to use in this case.

The use of coarse resolution grid-based data involves loss of spatial variability, and aggregation to a coarser resolution (upscaling) further decreases heterogeneity (Schulp and Alkemade, 2011). For the air quality regulation model an upscaling strategy was used, and high resolution data was adjusted to the coarse resolution ambient $PM_{10}$ concentration map (Velders et al., 2012). This led to spatial uncertainty, as urban areas that theoretically should not capture $PM_{10}$ (Powe and Willis, 2004), did receive positive values because of adjacent land cover types that do capture $PM_{10}$. Another type of uncertainty is associated with indicators that are related to movements of beneficiaries, such as cyclists in the case of recreational cycling. Such movements are difficult to capture in (static) maps, which cannot record the precise movements of stakeholders over time. In addition, indicators for cultural services often reflect perceptions of stakeholders. Changes in an ecosystem will affect behaviour of stakeholders and thereby ES flows based on these perceptions, with associated uncertainties in terms of linking ecosystem properties to services (Daniels et al. 2012).

### *2.5.3 Further development of biophysical ecosystem accounts*

A main goal of the ecosystem accounting is “the organisation of information sets for the analysis of ecosystems at a level suitable for the development, monitoring and evaluation of public policy” (UN et al., 2014a). Spatially explicit ecosystem accounts provide multiple advantages for reaching this goal and the need for a spatial approach has been mentioned in SEEA-EEA as important for ecosystem accounting (UN et al., 2014a). However, it seems that for accounting the full potential of spatially explicit analyses of ecosystems is currently underestimated; the SEEA-EEA remains very general on the contributions of spatial models to ecosystem accounting.

We have explored some possibilities that spatial accounting at multiple scales provides. First, it allows for a broad overview of ES flows at a large spatial scale that can be used in reporting systems. Although this study was done at provincial level, we believe that the methods that we followed would in principle also be appropriate at the national scale. Second, it provides the possibility to compare flows from different classes, as was demonstrated by the analysis of LCEUs. Third, it provides an understanding of the underlying spatial variation of the ES flows, relevant for local applications. Further refinement of spatial units for ecosystem accounting could lead to a more comprehensive information system. The smallest spatial units (i.e. the BSUs) could be filled with information on both ecosystem flows and conditions (e.g. land cover or soil type), as well as socio-economic characteristics (e.g. population density and economic activities, land management). Such an approach would make it possible to monitor ecosystems from more perspectives than land cover type, as was done in this study. Moreover, it would make a spatial ecosystem accounting system ideal for monitoring the capacity of ecosystems to provide ESs as well as the resulting flows. The BSU level can provide important accounting information that is relevant for assessing effects of local policy, and monitoring trade-offs between ESs and ES bundles.

A spatial ecosystem accounting approach is useful in support for various policy applications, varying from local to regional and national contexts. Ecosystem accounting can provide information for reporting systems at larger scale (e.g. provincial or national), through aggregated statistics such as total and mean ES flows. Besides providing conventional aggregated statistics, ecosystem accounts also enable spatially monitoring of where changes and degradation in ecosystems occur at high. Such a high resolution monitoring system could provide policy makers at a local and regional scale with information on effects of implemented policies on ESs, if monitoring happens on a regular basis. Also, for land use planners it provides information on the ESs that could be lost or enhanced under future management plans.

In the context of ES research closer collaboration among scientists from different disciplines and decision makers is needed (Crossman et al., 2013a). Ecosystem accounting requires collaboration between policy-makers, land managers, economists and ecologists, but also for example, the academic spatial modelling community. Closer involvement of the spatial modelling community in the development of ecosystem accounts can lead to more accurate models. Using novel spatial methodologies that incorporate both information on capacity of ecosystems to provide services as well as ES flows (e.g. Schröter et al., 2014b) and that more strongly incorporate the spatial distribution of and use by beneficiaries (e.g. Bagstad et al., 2013) could improve the information potential of ecosystem accounting. Further work is needed in order to distinguish ES flows and the capacity of ecosystems to provide services. Both are essential elements in ecosystem accounting, with the capacity representing 'ecosystem assets' under current ecosystem management. Mapping both ES flows and capacities of ecosystems to sustain these flows would also be an important method to analyse the sustainability of ecosystem use: areas where flow exceeds capacity during indicate unsustainable ecosystem use (Schröter et al. 2014).

## 2.6. Conclusion

This study has shown that spatial modelling of selected ESs for ecosystem accounting in line with SEEA-EEA is feasible for the case of Limburg province, the Netherlands, for which a lot of data is available. We outlined specific requirements for spatial modelling for the purpose of ecosystem accounting, namely a clear definition of ESs, quantifiable spatial indicators, high resolution models, high accuracy output at large spatial scales and an understanding of uncertainties. We empirically tested seven spatial models of ES flows, that largely met these requirements. In addition, the contributions of ecosystems and the contribution of humans to benefits were conceptually assessed, which were often difficult to disentangle. In the context of ecosystem accounting, in particular in cultural landscapes, it should be acknowledged that ecosystems are not fully natural systems, and are a result of ecological processes and historical human alterations that are often challenging to disentangle. The developed models for seven provisioning, regulating and cultural services were used to set up ecosystem accounting tables for the spatially detailed BSU level and for LCEUs. The models showed various uncertainties that need to be dealt with if a spatial approach to ecosystem accounting is to be operationalized. In a spatial accounting context a detailed system with BSUs that contain information on ecosystem conditions, ES flows and socio-economic characteristics would be more informative for monitoring spatial changes than highly aggregated statistics. Such a detailed system could be also be relevant in the context of spatial planning and strategic environmental assessments. For further development of spatial ES models for ecosystem accounting, a primary focus should be to increase data availability and accessibility, and developing models for ESs that have been rarely modelled, in particular cultural services.

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# Chapter 3 - Monetary accounting of ecosystem services: a test case for Limburg province, the Netherlands


## Abstract

Ecosystem accounting aims to provide a better understanding of ecosystem contributions to the economy in a spatially explicit way. Ecosystem accounting monitors ecosystem services and measures their monetary value using exchange values consistent with the System of National Accounts (SNA). We pilot monetary ecosystem accounting in a case study in Limburg province, the Netherlands. Seven ecosystem services are modelled and valued: crop production, fodder production, drinking water production, air quality regulation, carbon sequestration, nature tourism and hunting. We develop monetary ecosystem accounts that specify values generated by ecosystem services per hectare, per municipality and per land cover type. We analyse the relative importance of public and private ecosystem services. We found that the SNA-aligned monetary value of modelled ecosystem services for Limburg was around €112 million in 2010, with an average value of €508 per hectare. Ecosystem services with the highest values were crop production, nature tourism and fodder production. Due to exclusion of consumer surplus in SNA valuation, calculated values are considerably lower than those typically found in welfare-based valuation approaches. We demonstrate the feasibility of valuing ecosystem services in a national accounting framework.



Based on:
Remme, R.P., Edens, B., Schröter, M., Hein., L., 2015. Monetary accounting of ecosystem services: a test case for Limburg province, the Netherlands. *Ecological Economics* 112, 116-128.

## 3.1 Introduction

There is an increasing interest in environmental accounting as an approach to better understand economic implications of environmental change (Bartelmus, 2013; Obst and Vardon, 2014; UN et al., 2014b). A consortium led by the United Nations has recently released the third version of the System of Environmental-Economic Accounting (SEEA-2012), of which the Central Framework (SEEA-CF) serves as an international statistical standard and guideline for environmental-economic accounting (UN et al., 2014b). The compartmental approach of the SEEA-CF does not yet allow for integration of ecosystem services (ESs) into accounting (Edens and Hein, 2013). Therefore, a separate set of guidelines for ecosystem accounting were developed, the SEEA Experimental Ecosystem Accounting guidelines (SEEA-EEA) (UN et al., 2014a). A key objective of ecosystem accounting is to measure ESs in a way that is aligned with national accounts (as defined in the System for National Accounts (SNA), UN et al., 2009) (Edens and Hein, 2013; UN et al., 2014a). There has been steady progress in conceptualizing ecosystem accounting in recent years, yet, considerable challenges remain (e.g. Boyd and Banzhaf, 2007; UK NEA, 2011; Weber, 2011; Stoneham et al., 2012; Edens and Hein, 2013; Schröter et al., 2014b).

The SEEA-EEA emphasizes the importance of a spatial approach for ecosystem accounting, for both biophysical quantification and monetary valuation of ESs (UN et al., 2014a). The added value of using a spatial approach is threefold. First, it offers the opportunity to monitor local changes in addition to aggregated information collected in the SNA (Edens and Hein, 2013). Monitoring spatial changes can provide information for planning processes, such as land-use planning, for example by assessing whether specific ecosystems are degrading (Sumarga and Hein, 2014; Schröter et al., 2015). Second, it can help to shed light on spatial interrelationships between ES and dependence of ESs on socio-environmental conditions (Schröter et al., 2014b). Third, spatial modelling can offer wall-to-wall coverage of ESs in the absence of complete datasets (Stoneham et al., 2012).

The SEEA-EEA distinguishes between biophysical and monetary ecosystem accounting (UN et al., 2014a). While some empirical experience has been developed with biophysical ecosystem accounting (Remme et al., 2014; Schröter et al., 2014b; Schröter et al., 2015), only few studies apply monetary ecosystem accounting aligned with SNA principles for multiple ESs in a spatially explicit way (e.g. Campos et al., 2014). Monetary valuation can be a valuable complement to biophysical ES assessments (Troy and Wilson, 2006; Schröter et al., 2014a) and, for instance, be used to quantify and sum ESs using monetary estimates as a value measure and commensurable unit of account (Daily et al., 2009). In addition, monetary valuation can help to develop better informed land-use decisions (Goldstein et al., 2012).

The objective of this study is to test and apply a number of valuation approaches for ecosystem accounting building upon SEEA-EEA. Specifically, we assess how SNA valuation principles can be applied to a set of ESs and how resulting values can be represented in accounts for Limburg province, the Netherlands. Valuation is carried out for seven ESs, namely crop production, fodder production, drinking water production, air quality regulation, carbon sequestration, nature tourism and hunting. All monetary valuation approaches were coupled to spatial biophysical models developed for Limburg province (Remme et al., 2014), with exception of nature tourism and hunting. For these two ESs, new biophysical approaches were developed (section 2.2).

Although we do not aim to study specific policy applications of ecosystem accounting, we do elaborate on an example of how monetary accounting information can provide policy-relevant insights. We mapped public and private ES value, to raise awareness on the distribution of value to different types of beneficiaries across Limburg. We classified ESs as public or private according to the degree of rivalry and excludability (cf. Costanza, 2008; Kemkes et al., 2010). An ES is considered rival if use of the ES by one person prevents another person from using it. A service is excludable if people can be prevented from using it (Kemkes et al., 2010).

## 3.2 Methodology

### *3.2.1 Case study description*

Limburg province is located in the south-east of the Netherlands and covers approximately 2,200 km$^2$ (Figure 3.1). Limburg is densely populated (522 inhabitants per km$^{-2}$ in 2010), with a total population of 1.1 million people (Statistics Netherlands, 2013f). Over half of the inhabitants live in the southern one-third of the province. The southern part of the province is also nationally renowned for its hilly landscape and is popular with domestic tourists. The province has a varied cultural landscape, which has been managed for many centuries (Berendsen, 2005; Jongmans et al., 2013). Most natural ecosystems have been converted, resulting in a highly fragmented landscape (Jongman, 2002). There is high competition for land between agriculture, nature and urban land-uses (Vogelzang et al., 2010).

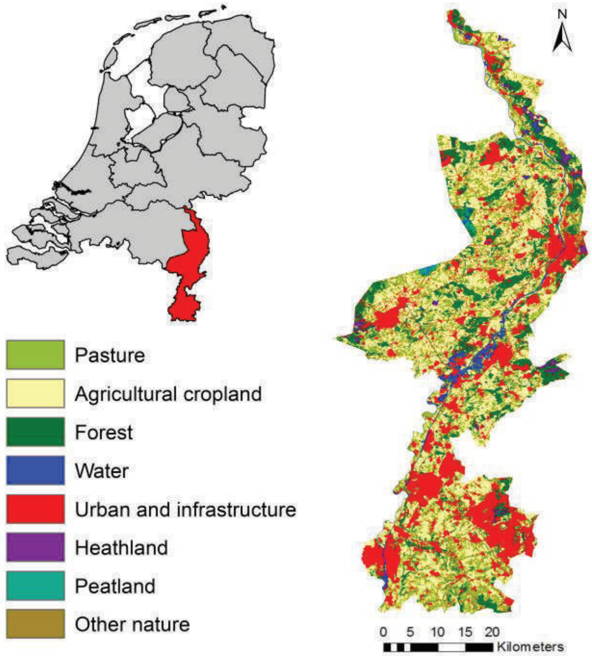


**Figure 3.1** Location and land cover of Limburg province, the Netherlands.

### *3.2.2 Biophysical spatial ecosystem service models*

Quantitative biophysical data of each modelled ES was used as input for valuation models. For the ES crop production, fodder production, drinking water production, air quality regulation and carbon sequestration, spatial biophysical models were used that are described in detail in Remme et al. (2014). All ESs were modelled for the year 2010. Most biophysical models were developed based on the Dutch 25 x 25 m land cover dataset LGN6 (Hazeu, 2009), with the exception of drinking water production and nature tourism. The latter models were developed using administrative boundaries (see Remme et al. (2014) and Appendix I).

For crop production biophysical production statistics were collected for five crop groups (cereals, potato, sugar beets, and other crops) (LEI and Statistics Netherlands, 2011). The production statistics were assigned to agricultural crop classes from the LGN6 national land cover map (Hazeu, 2009). Fodder production was modelled for 17 fodder production classes (16 grass classes, and silage maize) based on production statistics, soil types and livestock density (Aarts et al., 2005). Groundwater extraction for drinking water production was quantified and mapped for eleven groundwater protection zones ($m^3$ $ha^{-1}$ $yr^{-1}$). Air quality regulation was modelled based on particulate matter ($PM_{10}$) capture by vegetation, using ambient $PM_{10}$ concentrations per $km^2$ (Velders et al., 2012) and different vegetation types from the land cover map (Hazeu, 2009). Carbon sequestration was modelled by assigning carbon sequestration values from scientific literature to specific land cover types.

For the ES nature tourism we developed a biophysical spatial allocation model to represent spatial distribution of tourists visiting nature areas in Limburg. This allocation model calculates the number of tourists visiting nature areas within a 15 km radius around their accommodations, based on the accommodation capacity and distribution, as well as visitor statistics for three regions of Limburg (ZKA Consultants & Planners, 2011; Statistics Netherlands, 2013d). See Appendix I for a model description, underlying assumptions and data (Tables AI.1 and AI.2).

For hunting, the total area of five land cover types was used as biophysical indicator (contiguous forest (>40 ha), forest patches (<40 ha), cropland and natural grassland, pastures, and urban areas and infrastructure). These land cover types were used because the Royal Dutch Hunters Association collects data about prices of hunting rights on them (van Hout, personal communication). The LGN6 map was reclassified to match these five land cover types, and the areas of each land cover type were calculated.

### *3.2.3 Methodological foundation: ecosystem service valuation methods in the context of ecosystem accounting*

The main difference of ecosystem accounting valuation compared to welfare-based ES valuation methods (e.g. Liu et al., 2010; Turner et al., 2010) is that ecosystem accounting applies an exchange value approach (Edens and Hein, 2013; UN et al., 2014a). The exchange value approach focuses on valuing transactions as “amounts of money that willing purchasers pay to acquire goods, services or assets from willing sellers” (UN et al., 2009). A key characteristic of the approach is that consumer surplus is excluded from calculations (Edens and Hein, 2013). Use of exchange values is consistent with SNA valuation principles and allows integrating and comparing outcomes with information from national accounts, which is one of the main purposes of ecosystem accounting (UN et al., 2014a). Note that a welfare-based valuation approach may be more appropriate for other policy questions, such as cost-benefit analyses of projects or policies aimed at internalising environmental externalities (i.e. including side-effects of economic activities in their price) (Bateman et al., 2013). SEEA-EEA lists ES valuation methodologies that can be used in an ecosystem accounting context. The two most important methods are the resource rent method and replacement cost method, which are explained below. Some revealed preference valuation methods, such as the avoided damage cost method, travel cost method or hedonic pricing method, can potentially also be used within ecosystem accounting, if the method retrieves exchange values of ESs (UN et al., 2014a). We applied the avoided damage costs (section 2.3.3).

The SEEA-EEA defines ESs as “the contributions of ecosystems to benefits used in economic and other human activity” (UN et al., 2014a). Some of the benefits to which ecosystems contribute are already captured within the SNA (called “SNA benefits”). In such cases, ecosystem accounting makes the contribution of the ecosystem to the final product explicit, for example, by separately identifying the provisioning service of agricultural land (i.e. the contribution of the ecosystem) used in crop production. Ecosystem accounting also recognizes various benefits that ecosystems provide that are not captured in the SNA (called “non-SNA benefits”) as their provision is not considered as output of a productive activity in SNA terms (e.g. air quality regulation).

***Resource rent method***

According to the resource rent method, ES value can be estimated as the residual of the total revenue, after all costs for capital and labour have been subtracted (SEEA-CF, paragraph 5.118, UN et al., 2014b). Resource rent is calculated as follows:

$$RR = TR - (IC + LC + FC) \qquad (1)$$

where $RR$ is resource rent, $TR$ is total revenue or output of sales of a specific economic activity, $IC$ are intermediate costs, $LC$ are labour costs, $FC$ are user costs of fixed capital. Total revenue consists of the sales value expressed in basic prices, i.e. prices before subsidies on products are subtracted, and taxes on products and Value Added Tax are added (UN et al., 2009). Intermediate costs consist of operating costs, i.e. only current expenses excluding capital expenses or investments[1]. User costs of fixed capital consist of depreciation (consumption of fixed capital) and a return on fixed capital (the costs of capital). For the return on fixed capital an interest rate of 3.4% was applied, which consists of the interbank lending rate in 2010 and a risk premium (Veldhuizen et al., 2009). Resource rent represents the return on natural assets used in production (UN et al., 2009). The resource rent method has been applied for crop production, fodder production and nature tourism.

### *Replacement cost method*

The replacement cost method is a cost-based approach to value ESs that cannot be valued based on their market price (Liu et al., 2010; Turner et al., 2010). The method requires the existence of a substitute for the ES (Shabman and Batie, 1978; UN et al., 2014a). Three conditions need to be met to use the replacement cost method: (i) the substitute provides functions equal in quality and quantity, (ii) the substitute is the least cost alternative, and (iii) users can be expected to invest in the replacement if the ES is no longer available (Shabman and Batie, 1978; Bockstael et al., 2000; NRC, 2004). The ES can then be valued as the difference between the costs to acquire the ES and the costs of the most viable alternative (Gupta and Foster, 1975; Thibodeau and Ostro, 1981). Although the replacement cost method is not recommended for welfare-based valuations (NRC, 2004), it is suitable for exchange value-based valuation (UN et al., 2014a).

### *Avoided damage costs*

The avoided damage cost method is also a cost-based method. It estimates the value of an ES based on the costs that would have been incurred if the ES was absent (Liu et al., 2010; TEEB, 2010). The method can be used in situations where no suitable substitute exists for the ES (NRC, 2004). This is the case for the regulating services carbon sequestration and air quality regulation in this study. The applicability of the avoided damage cost method for ecosystem accounting is further discussed in section 4.1.1.

[1] The operating costs include taxes (minus subsidies) on production (see SEEA-CF paragraph 5.119, UN et al. (2014)). This type of information is however not readily available and could not be obtained for this study.

### *3.2.4 Monetary ecosystem service models for Limburg province*

All data were collected for the year 2010, unless stated otherwise, and all values presented are annual. Monetary values from other years were converted to 2010 euro values based on the consumer price index (Statistics Netherlands, 2013g). An ES value map was produced for each service. Ecosystem accounting tables were set up based on the model outcomes. Monetary values of the ESs were assessed for eight land cover types: *cropland, pasture, water, urban and infrastructure, forest, heath, peatland* and *other nature* (building on Remme et al., 2014).

#### *Provisioning services*

*Crop production*

The ES crop production was valued through the resource rent of agricultural companies engaged in crop production in the Netherlands, using data from the Dutch agricultural economics database BINternet (LEI, 2013d). The resource rent was calculated for four aggregated crop groups used in biophysical accounting (see Remme et al., 2014): *cereals, potatoes, sugar beets*, and *other crops*. For these calculations six arable crop groups from the BINternet were used (LEI, 2013e): wheat and barley for *cereals*; seed potatoes, starch potatoes and potatoes were aggregated for *potatoes*; and sugar beets was used for *sugar beets*. For *other crops*, data for 'open field vegetables' were used (predominantly consisting of cabbage and lettuce, but also other vegetables) (LEI, 2013a). Resource rent calculations were done separately and consistently for arable crops and *other crops*. We describe the method for arable crops, which was repeated for *other crops.*

Available data on revenues[2] and costs per hectare for six arable crops (wheat, barley, seed potatoes, starch potatoes, and potatoes and sugar beets) (LEI, 2013e) was used as input for resource rent calculations. The available intermediate costs items for these crops included costs for planting and energy costs (LEI, 2013e). Other intermediate costs items, such as fuel and maintenance of machines, financing costs, and external labour also needed to be deducted in order to calculate resource rent, and were taken from the profit and loss account for arable farms (LEI, 2013b). These costs were distributed across all six arable crop types after weighing them per hectare per crop based upon the number of hectares for an

[2] Revenues may differ from basic prices due to the existence of net taxes on products. In case of agricultural crops this difference is insignificant (Statistics Netherlands, unpublished data).

average farm[3]. Labour costs were deducted for each of the crop types (LEI, 2013b, a).The user costs of fixed capital were estimated using information about depreciation from the profit and loss account (LEI, 2013b, a), and information about the stock of fixed capital from the balance sheet of crop farms (LEI, 2013c). The user costs of fixed capital were distributed across the crop types after normalizing the costs per hectare based upon the relative share in total revenues of the crop types. Herewith we obtained the resource rent per hectare per crop. The resource rent per hectare was expressed as resource rent per ton crop produced using information about crop yields per hectare and aggregated (based on relative number of hectares per crop) to the four crop groups used in biophysical quantification: *cereals, potatoes, sugar beets*, and *other crops* (Table 3.1). For arable crops BINternet data was used (LEI, 2013e), for "other crops" information about crop yield per hectare was obtained from Statistics Netherlands (2013b).

**Table 3.1** Revenue, costs and resource rent for the four modelled crop groups, calculated based on data from LEI (2013d).

| | Cereal | Potatoes | Sugar beets | Other crops |
|---|---|---|---|---|
| **Total revenue (€/ton)** | 231 | 172 | 42 | 344 |
| **Intermediate costs (€/ton)** | 128 | 104 | 20 | 188 |
| **Labour costs (€/ton)** | 14 | 4 | 5 | 73 |
| **User costs of fixed capital (€/ton)** | 56 | 42 | 10 | 46 |
| **Resource rent (€/ton)** | 33 | 22 | 7 | 37 |

*Fodder production*

Fodder production was calculated based on grass and maize produced for on-farm use, both through harvesting and grazing (Remme et al., 2014). In the Dutch livestock sector cattle are fed harvested and stored fodder for a large part of the year, while in summer months harvested fodder is combined with grazing. Additional fodder purchased from other sources and not produced by the local ecosystem was excluded from the calculation. The used monetary cost data for

[3] It was not possible to make an estimate for (net) taxes on production per type of crop. However, based on regional accounts for Limburg province (Statistics Netherlands 2013b) we know that taxes on production for the whole agriculture sector in Limburg (ISIC Section A Agriculture, forestry and fishing) are slightly smaller than subsidies on production (resulting in an upward adjustment of the gross operating surplus of 4 percent in 2010). The absence of information on (net) taxes on production is therefore expected not to have a large effect on our results.

fodder production reflects the combination of grazing and harvesting of fodder (Alfa Accountants en Adviseurs, 2011). The value of fodder production was calculated as resource rent generated by fodder production. Revenue was based on the average purchaser price (excl. VAT) for a ton of hay, straw and maize in 2010 (LEI, 2013d). The contribution to revenue of these three fodder products was weighted according to the production on an average Dutch dairy farm (Alfa Accountants en Adviseurs, 2011). The purchaser price of 1 ton of fodder dry matter (dm) was approximately €121 in 2010. Transport and retail margins were estimated to be 10% of the purchaser price (Statistics Netherlands, unpublished) and were deducted to obtain basic price of €109/ton dm. Intermediate costs, labour costs and user costs of fixed capital involved in the production of fodder were based on fodder production costs of an average Dutch dairy farm (Alfa Accountants en Adviseurs, 2011). These costs combined were €96/ton dm. The obtained resource rent was multiplied with biophysical fodder production per location.

*Groundwater extraction for drinking water production*

Water extracted from shallow groundwater by the provincial drinking water company (WML) to produce drinking water was valued as the ES. Groundwater contributes to about three quarters of Limburg's drinking water (Vewin, 2013). Other drinking water is extracted through riverbank filtration, which was excluded from our calculations. Water companies in the Netherlands operate in a strongly regulated environment. This makes the resource rent method unsuitable for valuing this ES (Edens and Graveland, 2014). Instead, the replacement cost method was used. The least-cost substitute that can reasonably be expected to replace groundwater is surface water (in the form of water from the Meuse river). We therefore valued the ES as the difference between drinking water production costs for groundwater and for surface water. This cost difference was calculated as average production costs for Dutch surface water-based drinking water companies (at least 85% of production from surface water) minus average costs for Dutch groundwater-based drinking water companies (at least 85% of production from groundwater). Production costs and percentage of groundwater used by drinking water companies were obtained from Vewin (2013). Production costs included operating costs, costs of capital and depreciation and excluded taxes. The cost difference was €0.40/m$^3$. This value (€/m$^3$) was multiplied with the quantity of extracted groundwater (Remme et al., 2014) to obtain the ES value.

### *Regulating services*

*Air quality regulation*

To value the ES air quality regulation an avoided damage costs approach was used, with $PM_{10}$ capture by forests as biophysical indicator. The monetary value was spatially modelled using with data on ambient $PM_{10}$ concentration (Velders et al., 2012), forest cover (Hazeu, 2009) and population size (Statistics Netherlands and Kadaster, 2009) per km$^2$. Based on results from a British study of the West Midlands and Glasgow areas (McDonald et al., 2007) the relation between the percentage of forest cover and the decrease of the $PM_{10}$ concentration in the lower atmosphere can be expressed as:

$$C_P = 0.15 * F_P \qquad (2)$$

where $C_P$ is the reduction in $PM_{10}$ concentration (expressed as percentage) due to air filtration by forests and $F_P$ is the percentage of forest per km$^2$. The results from McDonald et al. (2007) were used because no such studies have been carried out in the Netherlands, and the case study areas are relatively similar (densely populated, mainly urban and agricultural land, and hilly terrain). The percentage of forest cover was calculated for each km$^2$ grid cell based on the LGN6 map (Hazeu, 2009). Using equation 2 the concentration difference between the current situation and a situation in which forests would have been absent was calculated, to calculate the total contribution of existing forests to the ES air quality regulation.

The avoided increase in $PM_{10}$ concentration was valued based on avoided air pollution-related health costs. We used health impact categories identified in a study by Preiss et al. (2008) on monetary costs of air pollution for health in the European Union. We included categories that were based on direct costs, while excluding categories that include components of consumer surplus (e.g. years of life lost and increased mortality risk). Damage costs for a person due to an increase in $PM_{10}$ concentration of 1 μg/m$^3$ were estimated using the various health impact categories (Table 3.2). The calculations estimate damage costs for an average person, using corrections for differences between adults and children from Preiss et al. (2008). The estimated damage value for an increase in concentration of 1 μg/m$^3$ is about €8 per person. Total avoided damage costs were calculated spatially by multiplying population size per km$^2$ (Statistics Netherlands and Kadaster, 2009), with the avoided $PM_{10}$ concentration and damage costs per μg per person, to obtain a monetary value map for air quality regulation by forests. The use of a 1 km$^2$ resolution was in line with several studies in the UK and the Netherlands (Powe and Willis, 2004; Oosterbaan et al., 2006), as well as the resolution of the input data on $PM_{10}$ concentration used for the biophysical model (Velders et al., 2012). In view of the uncertainty related to our assumption, we carried out a sensitivity analysis for a different spatial resolution of this model (section 4.1.2).

**Table 3.2:** Health impact categories resulting from $PM_{10}$ concentration change, their physical impact on a person and the monetary value of the treatment costs. Physical impacts and treatment costs are adapted from Preiss et al. (2008), unless stated otherwise.

| **Health impact categories** | **Physical impact per person per μg $PM_{10}$ (1/(μg/m³))** | **Treatment costs per case for 2010 (€)** | **Costs per person per μg $PM_{10}$ (€/person/μg/m³)** |
|---|---|---|---|
| Work loss days | $1.39 * 10^{-2}$ | 362 | 5.03 |
| New case chronic bronchitis | $1.86 * 10^{-5}$ | 22748[a] | 0.42 |
| Respiratory hospital admission | $7.03 * 10^{-6}$ | 2453 | 0.02 |
| Cardiac hospital admission | $4.34 * 10^{-6}$ | 2453 | 0.01 |
| Medication/bronchilator use child | $4.03 * 10^{-4}$ | 1.23 | 0.0005 |
| Medication/bronchilator use adult | $3.27 * 10^{-3}$ | 1.23 | 0.004 |
| Lower respiratory symptoms adult | $3.24 * 10^{-2}$ | 47 | 1.51 |
| Lower respiratory symptoms child | $2.08 * 10^{-2}$ | 47 | 0.97 |
| Total avoided costs per person per avoided $PM_{10}$ concentration increase | | | 8 |

[a] adapted from RIVM (2012).

*Carbon sequestration*

Carbon (C) sequestration does not require capital or labour inputs, therefore monetary values for avoided carbon emissions reflect the value of the ES. Carbon sequestration was valued using the social cost of carbon (SCC). The SCC is calculated based on damage costs of climate change. The SCC is based on the estimated economic damages of a marginal increase in $CO_2$ emissions, usually measured in metric tons per year (United States Government, 2013). We used the SCC as calculated by the United States Government (2013), which gives SCC values for three different market discount rates (2.5%, 3% and 5%). We converted the prices from US dollar to euro using average exchange values for 2010. Subsequently, we converted the prices from €/ton $CO_2$ to €/ton C. Carbon prices were calculated in 2010 euros, for the three discount rates. The SCC was assumed to be between €32/t C (5% discount rate) and €150/t C (2.5% discount rate). Obtained values are conservative estimates due to incomplete information on future impacts of climate change (IPCC, 2007). The SCC was multiplied with the biophysical quantities from the carbon sequestration model in Remme et al. (2014) to calculate the value of sequestered carbon in Limburg. For further calculation we use the highest discount rate applied by the United States Government (2013) (i.e. 5%) as a lower-bound value estimate of this ES. The selected discount rate differs from the rate of return applied in the resource rent approach, as the discount rate is applied for a different purpose compared to the rate of return on fixed capital. The discount rate includes aspects such as human health and non-market sectors and is used to analyse the SCC (United States Government, 2010), whereas the rate of return relates to financial capital.

### *Cultural services*

*Nature tourism*

The ES nature tourism was valued as resource rent generated by nature-based tourism. The total revenue for the tourism sector in Limburg was approximately €1.4 billion in 2010 (ISIC Section I Accommodation and food serving, Statistics Netherlands, 2013e), of which 23% can be accounted to business trips (ZKA Consultants & Planners, 2011). Revenues and costs for business trips were excluded from calculations because they are only marginally related to nature tourism opportunities provided in Limburg. Approximately 23% of all activities that were undertaken by tourists in Limburg were related to nature tourism (ZKA Consultants & Planners, 2011). Therefore, we assume that 23% of the remaining €1.1 billion total revenue can be allocated to nature-based tourism. Costs for nature tourism were calculated based on ISIC Section I Accommodation and food serving, Statistics Netherlands (2013e). Total revenue of nature-based tourism was € 247 million. Intermediate costs were €127 million, labour costs were €68 million and user costs of fixed capital were €14 million. The resulting resource rent for nature tourism was spatially allocated to nature areas across Limburg according to tourist visits and their expenditures, as described below.

Approximately 1 million tourists visited nature areas in Limburg in 2010. North Limburg and South Limburg each attract approximately 420,000 nature tourists, nearly three times more than Central Limburg. Average tourist expenditures differ between North, Central, and South Limburg, with expenditures being highest in the south (ZKA Consultants & Planners, 2011). Average resource rent per tourist was calculated separately for the three regions based on differences in average expenditure and the number of tourists visiting the area. Resource rent was spatially allocated to nature areas based on the number of tourists visiting nature areas within a 15 km radius around each accommodation. The 15 km radius was proposed by de Vries and Goossen (2002) for nature-based recreation in the Netherlands. Nature areas were defined as all areas that fall under a form of nature protection policy. All nature areas included in this study were freely accessible for tourists. Resource rent allocated to each specific accommodation was spread equally across all nature areas within the predefined radius of that accommodation.

*Hunting*

Hunting can be considered to be both a provisioning service (game meat) and a cultural service (recreational activity). In the Netherlands, hunting is primarily considered as a recreational activity (Bade et al., 2010). Therefore, we value the recreational service provided by hunting. Costs that are made by hunters to obtain the hunting rights for an area were used as indicator to value the ES, which is a way of estimating the resource rent (referred to as the access price method in the SEEA-CF, UN et al., 2014b). Hunters must obtain the hunting rights for a contiguous area of at least 40 ha in size to be allowed to hunt. The price paid for hunting rights depends on the contractual agreement between the hunter and the landowner. Values collected by the Royal Dutch Hunters Association (van Hout, personal communication) for Limburg were used. These values were assigned to the reclassified land cover map to obtain the ES value.

### *3.2.5 Value maps and private versus public services*

Based on the monetary value maps for each ES an aggregated value map was constructed, both at a hectare resolution and for each municipality. The modelled ESs were also mapped separately for services with a public and private character, using rivalry and excludability as criteria (Costanza, 2008; Kemkes et al., 2010). Crop production, fodder production and hunting were classified as private ESs, because they are all both rival and excludable. Carbon sequestration and air quality regulation, nature tourism and drinking water production were classified as public ESs. Carbon sequestration and air quality regulation are pure public goods, because they are both non-rival and non-excludable. Although the physical structures that contribute to and facilitate the use of the ES nature tourism are excludable (e.g. private hotels and restaurants), the ES itself is non-excludable, i.e. all tourists can visit nature areas. Therefore nature tourism was classified as a (congestible) public ES (Kemkes et al., 2010). It should be noted that this ES is congestible instead of non-rival, because crowding in nature areas can cause the quality of the experience to decrease, but we do not further consider this difference. Extracted water is sold by water companies and is therefore both rival and excludable. However, the ecosystem contribution, which is the filtration and storage of extractable drinking water, depends on a wide range of ecological processes that we consider to be non-rival and non-excludable. As we valued the ES and not the final good, we considered the ES drinking water extraction to be a public service.

## 3.3 Results

### *3.3.1 Ecosystem service valuation and maps*

***Ecosystem service valuation***

For the ES crop production, the resource rent was estimated to be €46 million (Table 3). The specific resource rents per crop group were: €7/ton for *sugar beets*, €22/ton for *potatoes*, €33/ton for *cereals* and €37/ton for *other crops*. The total resource rent from the *other crops* group constituted 62% of the total resource rent for the ES crop production. For fodder production, subtracting the total costs from the price per ton fodder gives a resource rent of €13/ton dm. Given the total fodder production in Limburg of 784 million kg dm (Remme et al., 2014), the value of the ES fodder production was approximately €10 million (Table 3.3). For drinking water production, the difference in costs between groundwater extraction and surface water extraction, was around €0.40/m$^3$, leading to a value of the ES drinking water production of around €11 million. The estimated value of $PM_{10}$ regulation by forests was approximately €2 million for Limburg. For carbon sequestration, the SCC-based value is €2 million with a 5% discount rate. Resource rent from nature tourism was about €39 million in 2010 (Table 3.3). The value of the ES hunting was estimated to be around €3 million (Table 3.4).

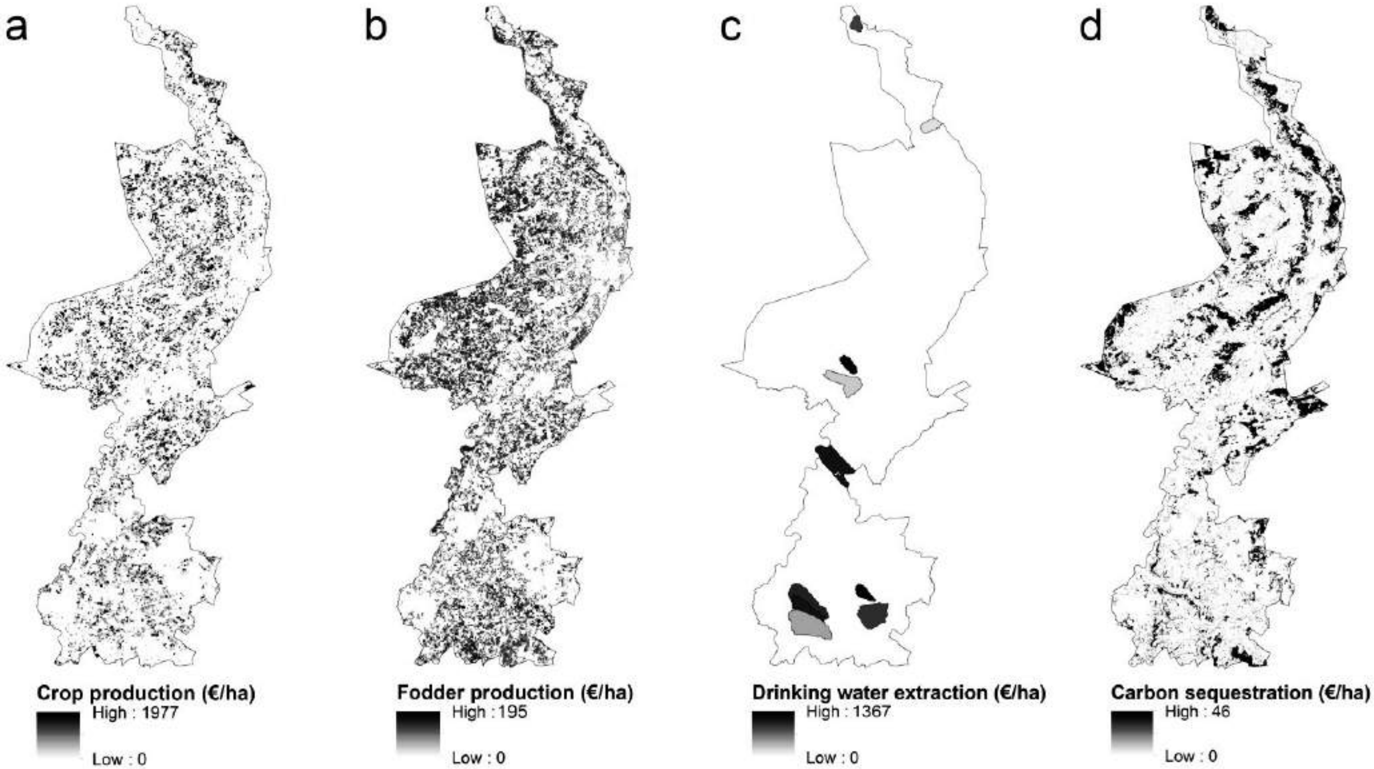


**Figure 3.2** Monetary value maps of the modelled ecosystem services: (a) crop production, (b) fodder production, (c) drinking water extraction, (d) carbon sequestration, (e) air quality regulation, (f) nature tourism, (g) hunting. In all maps white indicates the lowest values and black the highest values. All values are in €/ha.

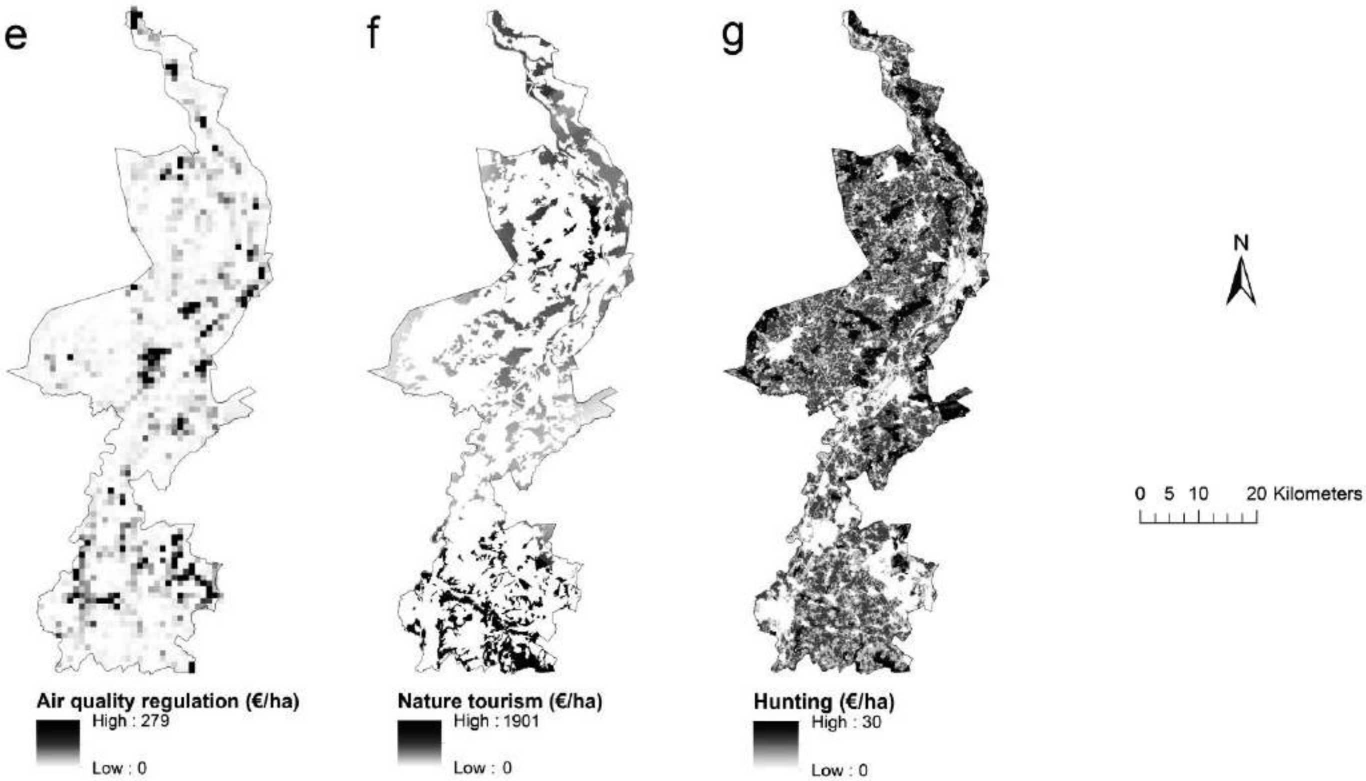


Figure 3.2 (continued)

*Ecosystem service value maps*

A monetary value map was produced for each modelled ESs (Figure 3.2). In Limburg, crop production and fodder production are spatially mutually exclusive, because these ESs are located on distinct land cover types. Monetary values of these two ESs show a large spatial variation. Drinking water production only covers a small spatial extent, spread across a large diversity of land cover types (seven). Carbon sequestration and hunting are highest in large forested areas, because the highest values for these ESs are found in that particular land cover type. For hunting, Figure 3.2 shows the results for the median value column in Table 3.4. Values for air quality regulation are highest in areas with a relatively large percentage of forest combined with a relatively high population density. Values are low in areas that have either a large population density and a low percentage of forest, or vice versa. Values for nature tourism are highest in the south, because this region receives a relatively large amount of tourists and resource rent per tourist is highest there.

**Table 3.3** ES valued with the resource rent method. Total revenue, costs and resource rent for crop production, fodder production and nature tourism.

| | Crop production (in million €) | Fodder production (in million €) | Nature tourism (in million €) |
|---|---|---|---|
| **Total revenue** | 386 | 86 | 247 |
| Intermediate costs | 214 | 17 | 127 |
| Labour costs | 61 | 27 | 68 |
| User costs of fixed capital | 65 | 32 | 14 |
| **Resource rent** | 46 | 10 | 39 |

**Table 3.4** Hunting value per land cover type for the lowest, average and highest indicated values per ha.

| Land cover type | Area (x 1000 ha) | Value range per ha (€/ha)* | Range provincial value (x1,000 €) | Median provincial value (x1,000 €) |
|---|---|---|---|---|
| Contiguous forest | 22 | 20 – 40 | 433 – 866 | 650 |
| Forest patches | 84 | 10 – 15 | 1,257 – 1,676 | 1,466 |
| Cropland, natural grassland | 12 | 15 – 20 | 121 – 181 | 151 |
| Pastures | 49 | 5 – 10 | 247 – 494 | 371 |
| Urban areas and infrastructure | 55 | 0 | 0 | 0 |
| **Total** | 222 | | 2,058 – 3,217 | 2,637 |

*Source: van Hout (personal communication).

Aggregated value maps of the ESs are presented in Figure 3.3. Figure 3.3a shows a relatively high spatial variation in monetary value per hectare, with a concentration of the highest values in southern Limburg. The high values are primarily driven by nature tourism, as well as crop production, fodder production and drinking water production. The values are lowest in large urban areas, where ESs flows are generally low. The high value in the south of the province can be explained by overlap between multiple ESs with high value per hectare (primarily nature tourism, crop production, fodder production and drinking water production). The map of average value per ha for each municipality (Figure 3.3b) shows a similar spatial distribution, with the highest values in the south of the province. The municipalities with the highest average values per ha are nature tourism hotspots and contain important drinking water extraction areas. Municipalities with large cities generally have a lower ES value per ha than more rural municipalities.

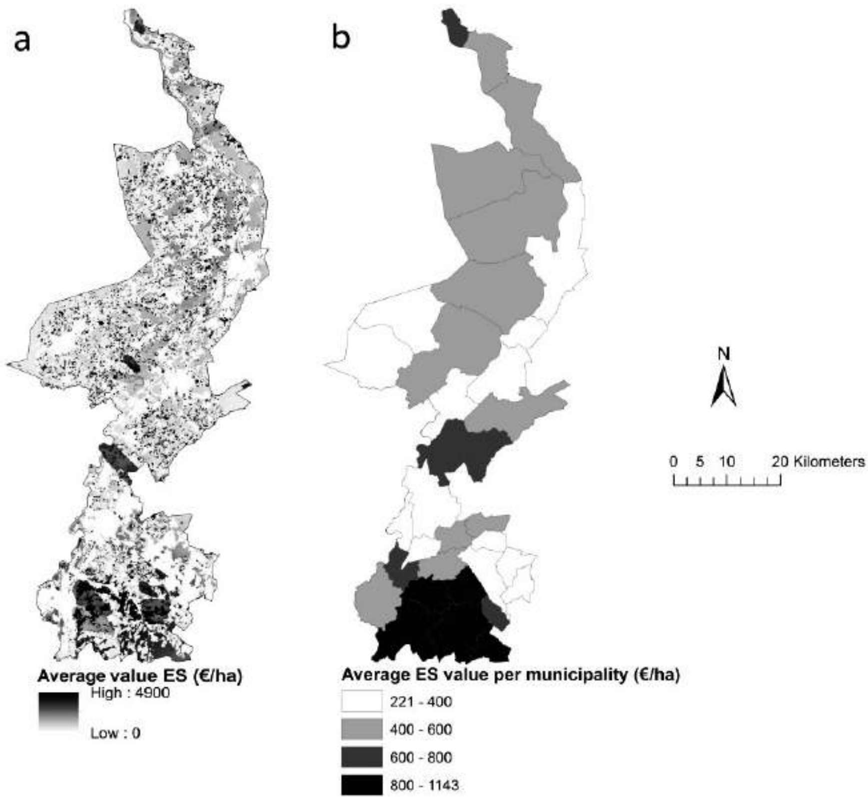


**Figure 3.3** Aggregated value maps (€/ha) for ecosystem services represented (a) per hectare and (b) per municipality.

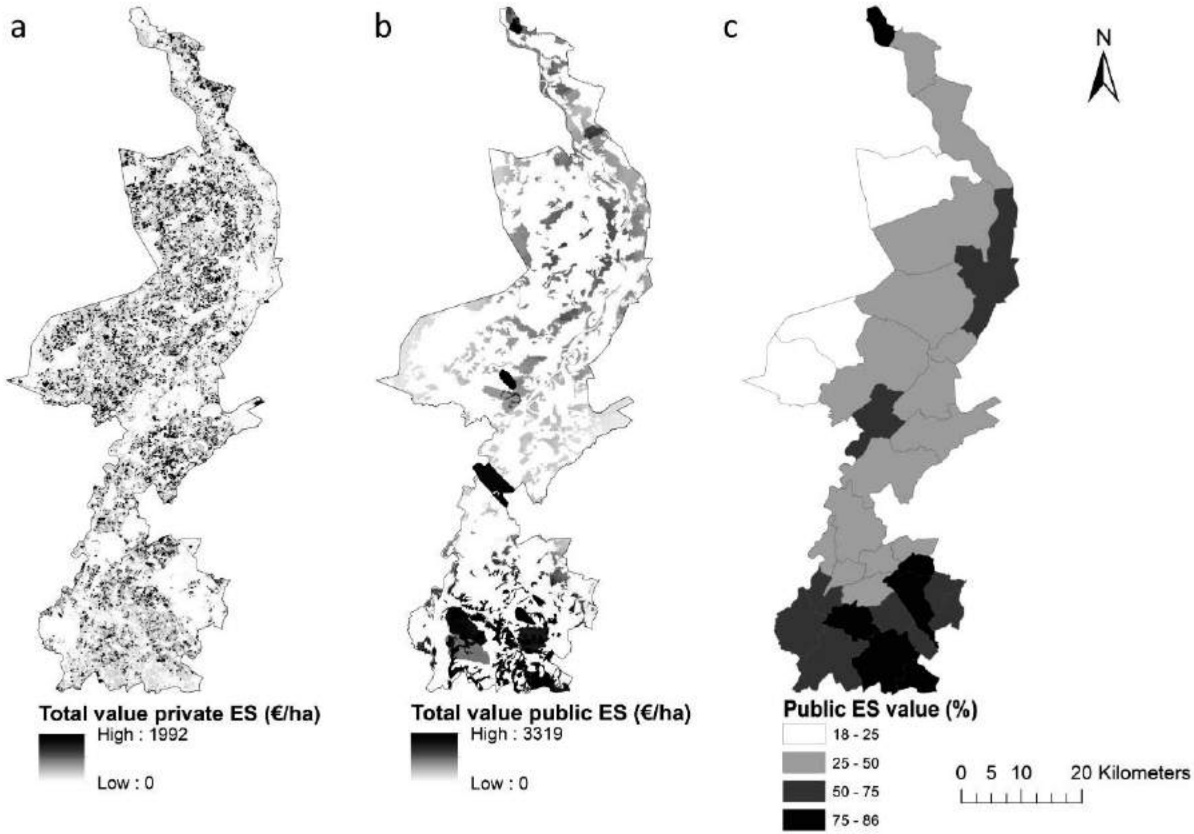


**Figure 3.4** Aggregated monetary value maps for (a) the private ecosystem services and (b) the public ecosystem services; and (c) the percentage of value generated by public ecosystem services per municipality.

ESs with a public and private character have a different spatial distribution of monetary value. Value of private ESs (Figure 3.4a) is predominantly found on agricultural land, whereas the value of the modelled public ESs (Figure 3.4b) is largely found in areas under some form of protection (e.g. nature areas or drinking water protection zones). The value of public ESs is concentrated in the south of the province, mainly because of groundwater extraction and the high number of nature tourists, whereas the spatial value distribution of private ESs is highly scattered throughout the province. Private ESs attribute 52% of the calculated ES value, while public ESs attribute 48% (Table 3.4), but this relative share of public and private ESs is not evenly distributed across the province (Figure 3.4c). In the central municipalities the contribution of private ESs is generally higher than that of public ESs, while in the southern municipalities the contribution of public ESs is generally higher. The southern municipalities are also the municipalities with the highest average ES value per ha. In 18 out of the 33 municipalities the contribution of private ESs to the aggregated value is higher than that of public ESs.

### *3.3.2 Ecosystem accounting tables*

#### *Aggregated value of modelled ecosystem services*

The total, SNA-aligned monetary value of the modelled ESs for Limburg was estimated to be about €112 million in 2010 (Table 3.5). The average value per hectare was €508 (SD ±655). Crop production and nature tourism constitute the two most important ESs in monetary terms. Together, these two ESs contribute about 75% of the monetary value of the modelled ESs. The two regulating services have the smallest calculated monetary value. For the ESs crop production, fodder production, drinking water extraction and nature tourism, the value of the service only constitutes a small portion of the gross revenue (10% to 16%).

**Table 3.5** Total annual biophysical flow and calculated monetary value of ecosystem services, and gross revenue for services considered in the SNA.

| Ecosystem service | Biophysical quantity | Gross revenue (million €)[a] | Monetary value of ecosystem service (million €) |
|---|---|---|---|
| **Crop production** | 1.9 $*10^9$ kg produce[#] | 386 | 45.9 |
| **Fodder production** | 0.8 $*10^9$ kg dm fodder[#] | 86 | 10.2 |
| **Drinking water extraction** | 28 $*10^6$ $m^3$ water[#] | 104[b] | 10.8 |
| **Air quality regulation** | 2.3 $*10^6$ kg PM10[#] | - | 2.0 |
| **Carbon sequestration** | 61 $*10^6$ kg C[#] | - | 2.0 |
| **Nature tourism** | 1.0 $*10^6$ tourists | 248[c] | 38.7 |
| **Hunting** | 1.7 $*10^3$ $km^2$ hunting ground | - | 2.6 |
| **Total** | | | 112 |

[#] For calculations see Remme et al. (2014)
[a] For ES that are part of the SNA only.
[b] Waterleiding Maatschappij Limburg (2010)
[c] For nature tourism only. Derived from Statistics Netherlands (2013e) and ZKA Consultants & Planners (2011).

### *Accounting per ecosystem/land cover*

*Cropland* accounts for approximately 55% of the annual value of the modelled ESs (Table 3.6), mainly because of the high value calculated for the ES crop production. Land cover types with a higher degree of naturalness (*forests, heath, peatland, water* and *other nature*) together are responsible for about 25% of the aggregated value, of which the largest part can be attributed to *forests*. *Cropland* has the highest average value per hectare, resulting mainly from the high value per hectare of the ES crop production. *Other nature* has a similarly high average value per hectare, mainly due to the ES nature tourism. *Forests* also have an average value per hectare which is higher than the provincial average, for a large part due to the ES nature tourism. The land cover *urban and infrastructure* has a very low average ES value per hectare. Public ESs are strongly dominant in all land covers except *cropland* and *pasture*. *Cropland* is the only land cover type in which private ESs strongly determine the monetary value. It should be noted that standard deviations, reflecting the distribution of values per grid cell, are high for all land covers. In some cases they are higher than the average value per hectare.

**Table 3.6** Accounting table for value of modelled ecosystem services.

| Land cover type | Cover (%) | Total ES value (million €) | Average value (€/ha) | Standard deviation (€/ha) | Minimum value (€/ha) | Maximum value (€/ha) | Value public ES (%) | Value private ES (%) |
|---|---|---|---|---|---|---|---|---|
| Cropland | 33.9 | 61.9 | 823 | 815 | 14 | 4,900 | 18 | 82 |
| Pasture | 20.2 | 18.6 | 412 | 507 | 10 | 3,361 | 61 | 39 |
| Forest | 15.3 | 19.9 | 587 | 473 | 56 | 3,226 | 96 | 4 |
| Urban and infrastructure | 23.6 | 4.8 | 90 | 277 | 0 | 2,900 | 99 | 1 |
| Other nature | 2.7 | 4.9 | 814 | 687 | 15 | 3,186 | 94 | 6 |
| Water | 3.0 | 1.6 | 239 | 313 | 0 | 2,906 | 100 | 0 |
| Heath | 1.0 | 0.9 | 426 | 288 | 20 | 1,923 | 96 | 4 |
| Peatland | 0.3 | 0.3 | 457 | 135 | 21 | 653 | 97 | 3 |
| Total province | | 112 | 508 | 655 | 0 | 4,900 | 48 | 52 |

## 3.4 Discussion

### *3.4.1 Uncertainty and sensitivity in valuation approaches*

***Model uncertainties***

Transparency on uncertainties in monetary valuation is essential in ES research (Liu et al., 2010), especially since ES valuation studies have drawn wide attention in science and media. Assessing the uncertainty of specific models remains an aspect of ES research that requires more attention (Seppelt et al., 2011). Therefore, we assessed the main uncertainties of both the biophysical and monetary aspects of our models (Table 3.7). Very few biophysical ES models have been validated (Martínez-Harms and Balvanera, 2012), and uncertainty in many current ES maps is high (Schulp et al., 2014a). In our biophysical models uncertainties are commonly related to a lack of local data on the ESs (Table 3.7). Data availability was insufficient to validate spatial variation in the biophysical models (Remme et al., 2014). Monetary valuation models are affected not only by insufficient availability of input data (Schägner et al., 2013), but also by uncertainties in the biophysical models. Better understanding uncertainties underlying biophysical ES models in future ES modelling studies will help increase the reliability of monetary information for decision-making. Obst et al. (2013) signal that availability of high quality data is an important precondition for ecosystem accounting and call for investments to achieve this.

**Table 3.7** Main uncertainties in the spatial models and valuation approaches per ecosystem service. For more extensive discussion on biophysical uncertainties, see Remme et al. (2014).

| Ecosystem service | Main biophysical uncertainty | Main valuation uncertainty |
|---|---|---|
| **Crop production** | -Production figures based on regional statistics, little local variation | -Resource rent estimate based on Dutch averages, instead of data specific for Limburg (or even better, micro-data)<br>-Information missing about (net) taxes on production<br>-Part of the resource rent will reflect mixed income |

Table 3.7 (continued)

| Ecosystem service | Main biophysical uncertainty | Main valuation uncertainty |
|---|---|---|
| **Fodder production** | -Lack of local quantitative data on fodder production | -Transport and retail margins were estimated for fodder, due to lack of data-An average mix between fodder types assumed due to lacking local data<br>-A single quality of fodder was assumed, due to lacking data fodder quality |
| **Drinking water extraction** | -Spatial variation within groundwater protection zones could not be modelled | -Average values at company level used, i.e. no local variation in differences on costs for surface and groundwater production |
| **Air quality regulation** | -Little empirical data on relation between vegetation and $PM_{10}$ concentration<br>-Analysis done at a 1 km² resolution, coarse compared to other ES | -Little national data on costs of treatments resulting from air pollution<br>-Valuation only carried out for forests, data for other land cover types was not available |
| **Carbon sequestration** | -Look-up table approach is very static, no variation within land cover types | -Choice of discount rate and social costs of carbon |
| **Nature tourism** | -Assumed distance travelled from accommodation (max. 15 km) | -Assumed time and expenses of tourists allocated to nature<br>- Assumed attractiveness evenly, while areas are more aesthetically pleasing than others and will make people travelling longer |
| **Hunting** | -Valuation model not connected to local species populations | -Monetary values for land cover types only indicative<br>-Hunting rights only a partial indicator of hunting as a recreational activity |

The monetary valuation methods have additional uncertainties, mostly related to the aggregation level of data (Table 3.7). For most models not all required monetary data was available at local or regional scale, and we had to resort to national averages, for instance for fixed asset values. Although general limitations of specific ES valuation methods have been widely documented (e.g. Chee, 2004; NRC, 2004; Liu et al., 2010; Turner et al., 2010; Bateman et al., 2011), the valuation methods we applied had some specific additional limitations. A disadvantage of the resource rent method is that the residual may not exclusively consist of the return on natural capital. A well-known issue for instance is the existence of mixed income in agriculture (UN et al., 2014b), i.e. compensation of self-employment by the farmer or other members of the household that will form part of the operating surplus. Methods to separate mixed income from resource rent have been tested (Campos et al., 2009), but we lacked the data for this calculation. Since we did not distinguish between resource rent and mixed income, we could overestimate resource rent. In addition, while capital gains from ES are sometimes included in calculations of resource rent (e.g. Cavendish, 2002), we excluded this in order to be consistent with SNA principles. The residual resource rent may also include, next to the contribution of the ecosystem, return on other types of (intangible) capital (e.g. social, institutional or knowledge). The resource rent of crops is therefore an upper bound of the ES value. As for the avoided damage cost method, it is as yet unclear if this method is indeed fully aligned with the SNA valuation principles (UN et al., 2009). In particular, it is not a given that society would indeed choose not to (partially) mitigate damage costs, would these costs occur as a consequence of ecosystem degradation. However, in the case of Limburg, there is no alternative that is more aligned with the SNA to value carbon sequestration and air quality regulation.

***Sensitivity of results***

The sensitivity of outcomes was tested for some ESs by adjusting the model resolution and by changing input values. To test the spatial sensitivity of the air quality regulation results, the ES was also modelled at 2x2 km resolution, using the same procedure as for the 1 km$^2$ model. Both the avoided change in $PM_{10}$ concentration and population size were recalculated for the coarser resolution model. The model resulted in an ES value of €2.7 million for Limburg, 30% higher than the 1 km$^2$ model. There was a fair spatial correlation between the models (Spearman's rho = 0.68). Both models slightly overestimated the population size of Limburg, due to rounding errors in the upscaling procedure. However, the overestimation was larger in the 2x2 km model (about 10% compared to 6% for the

1 km$^2$ model), contributing to the differences in outcome between the models. The sensitivity analysis shows that model resolution can have a strong influence on monetary ES value. This influence was also demonstrated by Konarska et al. (2002), by comparing the ES value of land cover datasets with different scales. The study, however, found an opposite relation compared to our case, with higher values for higher resolution land cover data. The dependence of valuation results on spatial resolution requires further attention (Tianhong et al., 2010).

As presented for carbon sequestration and hunting, different input values are important for determining ES value. For carbon sequestration the value was calculated based on SCC with three different discount rates. The value for this ES ranged between €2-8 million depending on the chosen discount rate. This shows that, in this particular case, applying a different discount rate might change the estimated value of carbon sequestration by a factor 4. For hunting, a range of input values for hunting rights per land cover type was provided by the Royal Dutch Hunters Association (Table 3.4), resulting in a total ES value of between €2.1-€3.2 million. Given the modest contribution of carbon sequestration and hunting compared to crop production and nature tourism, the effect of these uncertainties on the overall monetary value estimate is small.

### *3.4.2 Limitations of using exchange value and not welfare*

Not all ESs can currently be accounted for in ecosystem accounting (Bartelmus, 2013), especially services that mainly or only generate a consumer surplus (e.g. artistic and education services, cf. Chan et al., 2012). To give an example for Limburg, we have been able to account for the value of nature tourism (a SNA benefit), but were unable to model the recreational value of nature for local residents (a non-SNA benefit). A potential way forward may consist of using methods that estimate a demand curve for a specific service that is subsequently intersected by a modelled supply curve, as in the simulated exchange value approach (Campos and Caparrós, 2006; Oviedo et al., 2010). Another alternative would be to base the demand calculations on empirical observations (e.g. the number of visits to a nature area). These methods have not been widely tested and further research is needed to explore how they can be used for valuing cultural and regulating services in an ecosystem accounting context. Alternatively, including approaches into ecosystem accounting that are more lenient towards the use of valuation methods that include consumer surplus could be explored (e.g. Banzhaf and Boyd, 2012), more closely related to a welfare-based approach. Examples of accounting frameworks that provide a welfare-based approach are Inclusive and

Comprehensive Wealth Accounting (e.g. Arrow et al., 2003; Mäler et al., 2008; Duraiappah and Muñoz, 2012). However, such approaches would make ecosystem accounting inconsistent with SNA and are therefore not a viable option from the perspective of the SEEA-EEA (UN et al., 2014a).

We illustrate the difference in monetary values between the exchange value approach and welfare-based approach for air quality regulation. We calculated a provincial value of €2 million, resulting in a value of approximately €900/ton $PM_{10}$ avoided. When compared to air quality regulation studies reviewed in Gómez-Baggethun and Barton (2013), our results (in €/ton $PM_{10}$ avoided) are between a factor 2 to 20 lower. Likewise, for the Dutch national park Hoge Veluwe, Hein (2011) valued one ton of $PM_{10}$ captured at over €10,600, more than a factor 10 higher than our result. If all welfare-related health damage categories from Preiss et al. (2008) are added to our ecosystem accounting result (see Appendix I Table AI.3 for values), the air quality regulation value would be about €4900/ton $PM_{10}$ avoided and the provincial value of this service would be nearly €11 million. This result is within range of the studies included in Gómez-Baggethun and Barton (2013).

### *3.4.3 Implications for policy-making*

The primary functions of ecosystem accounting are to monitor changes in ecosystems and the services they provide, and to make the contributions of ecosystems to economic activities visible (UN et al., 2014a). Hence, ecosystem accounting has not been developed based on specific policy goals, but rather as an information system which is useful for different policy contexts, including policy evaluation (Obst and Vardon, 2014). It has the potential to support a variety of policy purposes, including recognizing, demonstrating, monitoring and capturing value (Schröter et al., 2015). Bartelmus (2013) argues that the current SEEA revision does not sufficiently address capabilities and limitations of ecosystem accounts to inform and monitor sustainability policies. This is mainly due to a missing track-record in terms of informing and evaluating policy. Further work is required to test the potential and limitations of ecosystem accounting as a sustainability and policy evaluation tool (Obst and Vardon, 2014), as briefly discussed below.

At the provincial or national scale, monetary ecosystem accounting can increase our understanding of the contributions of ecosystems to economic activities, and can help to raise awareness about services that are not covered by national accounts, such as regulating services. Assessments based on ecosystem accounting information can serve as early warning systems that signal degradation or loss of ES value, comparable to other integrated assessments (e.g. MA and TEEB, cf. Bateman et al., 2011), in order to trigger policies that target specific ES or ecosystems. In addition, aggregated ecosystem accounting information can provide a foundation for evaluating existing policies that focus on land-use change or nature conservation. Comparing ecosystem accounting results with national or regional accounts could be possible, but should be done with caution. For example, the €112 million euro ES value seems insignificant compared to Limburg's value added of over €31 billion in 2010 (Statistics Netherlands, 2013e), but it is important to keep in mind that we have not valued all ESs in this study. Furthermore, exchange values of ESs do not fully reflect their importance for society. For instance, drinking water is crucial to sustain human lives and fertile soils are essential to generate agricultural revenue. We value the subset of ESs according to an ecosystem accounting approach, which is just one of several possible ways to value ESs and should by no means be understood as the total value of nature.

At local scale, spatial monetary accounts can contribute to analysing and informing land-use policies or understanding trade-offs between ESs. Optimizing spatial patterns of land-use types and management of ecosystem flows remains challenging (De Groot et al., 2010). Spatially explicit ecosystem accounting information can contribute to informing such policy processes. For example, the analysis of public ES value (Figure 3.4) can raise awareness on which areas are of high value to the general public, and how public and private ES values are distributed across the province and municipalities. Such information could provide a starting point for dialogue between policy-makers and other stakeholders to develop local land-use plans. Local land-use policies are unlikely to be developed on ecosystem accounting information alone, since other values, such as community values (Raymond et al., 2009; Plieninger et al., 2013) are also of crucial importance here. Spatially explicit monetary accounting can also raise awareness on ES trade-offs that occur as a result of changes in the landscape. For example, the effects of a conversion of forest into another land-use can be displayed through changes in the ES value of the area.

## 3.5 Conclusions

Our study shows the feasibility of valuing ecosystem services in a national accounting framework for Limburg province. As the exchange value approach was applied, the results of our study are aligned with UN accounting standards (SNA). The average value per hectare for seven ESs in Limburg was calculated to be €508 in 2010. Crop production, nature tourism and fodder production made the highest contribution to the total ES value. Private ESs provide a higher contribution to the aggregated provincial value than public ESs. We demonstrate that the value of some services, such as air quality regulation, is considerably lower than the value in a welfare-based valuation approach. This difference in value is related to the relative contribution of consumer surplus to the overall economic value. Combined with biophysical accounts for ESs, monetary accounting can provide information on ES flows at local and provincial scales. Our study illustrates some of the remaining challenges in ecosystem accounting, such as a lack of monetary data on ESs at local scale, causing uncertainty in finer scale distribution of ES value. Furthermore, modelling choices, such as the spatial resolution of a model and the selected discount rate, considerably affect the model ES value. In its current state, ecosystem accounting is a suitable system for elucidating the contributions of ecosystems to economic activities recorded in the national accounts, as well as for capturing exchange values of some ESs that are not included in these accounts. However, capturing the value of many regulating and cultural services with exchange value methods remains a challenge. Further research and testing is necessary to assess how to integrate them into an ecosystem accounting framework. Our study shows how ecosystem accounting provides spatially explicit information on the contribution of ecosystems to economic activities, and that valuation approaches for ecosystem services aligned with accounting can be applied at the scale of a province.

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# Chapter 4 - Exploring spatial indicators for biodiversity accounting


## Abstract

In the context of the System for Environmental-Economic Accounting (SEEA), biodiversity accounting is being developed as a tool to monitor and increase the understanding of human impacts on biodiversity. Biodiversity accounting aims to structurally measure and monitor changes in multiple biodiversity components. Indicators relevant for ecosystem functioning and indicators that reflect human appreciation of ecosystems can be included in biodiversity accounting. In this paper we focus on the latter. We assess various indicators for species diversity for Limburg province, the Netherlands in terms of their applicability in the SEEA framework. In particular, we analyse a range of indicators reflecting species richness, the presence of rare and threatened species and species abundance using six different criteria. We show that for Limburg province spatial variation between the occurrence of different species groups is large, which implies that in the development of biodiversity accounts multiple species groups should be considered. Species richness is useful as an indicator to identify areas of particular importance for biodiversity conservation, with Limburg province showing a strong spatial correlation between species richness for all species and species richness of threatened species. Rarity indicators and species abundance indicators showed weak spatial correlation with species richness, providing complementary information on species distribution. All indicators had different strengths and weaknesses, implying that in the development of biodiversity accounts multiple species groups and multiple indicators need to be combined. However, the specific combination that provides the most comprehensive information while restricting the amount of indicators is likely to differ between different areas.



Based on:
Remme, RP, Hein L, van Swaay CAM. Exploring spatial indicators for biodiversity accounting. *Ecological indicators* (submitted).

## 4.1 Introduction

Biodiversity and ecosystems are under increasing threat (Vitousek et al., 1997; MA, 2003; Hooper et al., 2005; Díaz et al., 2006). Numerous international agreements and strategies to halt biodiversity loss have been developed over the past decades, with the Convention on Biological Diversity (CBD), the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) and the European Union Biodiversity Strategy (European Commission, 2011) as prominent examples. Nevertheless, biodiversity is still in decline (Butchart et al., 2010; Pereira et al., 2012) and more efforts are needed to halt further losses. In order to monitor trends in biodiversity and support the design of response options, biodiversity and ecosystem accounting systems are being developed (Edens and Hein, 2013; UN et al., 2014a; Hein et al., 2015).

Approaches to include ecosystems and biodiversity, and ecosystem services in accounting systems are currently being designed under auspices of the UN Statistics Division as the System for Environmental-Economic Accounting (SEEA) (UN et al., 2014b; UN et al., 2014a). The SEEA Central Framework has been adopted by the UN Statistics Commission as a statistical standard (UN et al., 2014b), but does not capture ecosystem services and biodiversity. Therefore, the SEEA Experimental Ecosystem Accounting framework (SEEA-EEA) was developed, which offers a more comprehensive scope for including biodiversity and ecosystem services (UN et al., 2014a). Various ecosystem accounting aspects introduced in the SEEA-EEA have been tested, including accounting for ecosystem capacity, biophysical and monetary ecosystem service flows (Schröter et al., 2014b; Remme et al., 2015; Sumarga et al., 2015), and are being applied, for example in the World Bank's WAVES programme (World Bank, 2013). There is increasing evidence for the link between ecosystem services and biodiversity (e.g. MA, 2005; Mace et al., 2012; Reyers et al., 2012; Harrison et al., 2014; Schröter et al., 2014a). Also, biodiversity as a service in itself is an important consideration in ecosystem management (Hein, 2010; Mace et al., 2012; UN et al., 2014a). Nevertheless, there is as yet limited experience with analysing how biodiversity accounts could be developed as a complementary account in the SEEA-EEA framework. To date, only Bond et al. (2013) have tried to integrate biodiversity into an ecosystem accounting framework by accounting for changes in biodiversity in Victoria, Australia based on land cover change. Although biodiversity accounting is part of the SEEA-EEA (Hein et al., 2015), how to specifically include biodiversity in the framework is still being discussed (UN et al., 2015).

Biodiversity accounting, as tested to date, has focussed primarily on species diversity (Jones and Solomon, 2013). Biodiversity accounting is not entirely new, although it is in the early stages of development. Jones (1996) initiated a pilot study to outline a general model for biodiversity accounting. Nearly a decade later this pilot study was followed up by a case study at a company scale (Jones, 2003). Since then a number of case studies have been published, focussing mostly on biodiversity accounting for corporations and projects (Gardner et al., 2013; Rimmel and Jonäll, 2013; van Liempd and Busch, 2013; Virah-Sawmy et al., 2014). The only case where a large scale account was developed was by Bond et al. (2013) for Victoria, Australia. That study is one of the first biodiversity accounting studies to apply a spatial approach. For accounting according to the SEEA-EEA framework, a spatial approach is required to capture the spatial heterogeneity of ecosystem services and biodiversity (UN et al., 2014a). The SEEA-EEA approach focuses on accounting for large administrative areas, such as provinces and countries, but it does not aim to develop a global account. For biodiversity accounting in line with SEEA-EEA, methodologies can be developed that draw upon a wide range of studies on biodiversity indicators (e.g. Noss, 1990; Gregory et al., 2005; Butchart et al., 2010; Mace et al., 2010; Feest, 2013) and biodiversity monitoring systems, such as GEO BON (Pereira et al., 2013) or WWF's Living Planet Index (WWF, 2014).

By including biodiversity components that underpin different services (cf. Mace et al., 2012), such as game species or plant primary production, several ecosystem accounting studies indirectly cover biodiversity aspects that are important for providing certain ecosystem services (e.g. Remme et al., 2014; Schröter et al., 2014b). However, to date such studies have not included species diversity, which is considered a key biodiversity component (Noss, 1990; Chiarucci et al., 2011). The importance of species diversity is reflected by the value people ascribe to the existence of species or biodiversity in certain areas (Reyers et al., 2012; Boykin et al., 2013). Accounting for species diversity is complementary to other sets of ecosystem accounts and important for ecosystem management. Biodiversity accounting can cover aspects of cultural appreciation that have so far not been included in ecosystem accounts, such as the existence of populations of wild animals or the appreciation of species diversity, including diversity within individual species (Mace et al., 2012). To account for species diversity a comprehensive set of indicators needs to be identified, which can be frequently measured and recorded at aggregated scales.

Data on biodiversity is always location specific and spatial analysis is a key component of biodiversity assessments (Buckland et al., 2005). Therefore, in addition to assessing time series, many biodiversity indicators are spatially monitored and recorded, at a range of different scales (e.g. Noss, 1990; Chiarucci et al., 2011). Integrating such indicators into a larger accounting framework will give a more comprehensive overview of changes in ecosystems and biodiversity. In this study, we aim to assess which spatially explicit biodiversity indicators would potentially be suitable to include in biodiversity accounts. We identify criteria for selecting indicators that are relevant for biodiversity accounting, and test a set of biodiversity indicators. Given that countries will generally face data shortages and limited resources for collecting additional data to prepare biodiversity accounts, we explicitly examine how indicators sets can be simplified while still capturing essential information required to support policy-making. We provide a first assessment of biodiversity accounting indicators, which is an important step in the development of spatial biodiversity accounts, embedded in a larger ecosystem accounting system (as described by SEEA-EEA (UN et al., 2014a)). We build on the suggestions for biodiversity accounting done by SEEA-EEA (UN et al., 2014a), and focus on the need for governments to account for changes at large administrative scales. We analyse species richness, rare and threatened species and species abundance for a relatively data-rich region, Limburg province, the Netherlands.

## 4.2 Methods

### *4.2.1 Study area*

Limburg province is located in the south-east of the Netherlands and covers approximately 2,200 km$^2$. Limburg is densely populated (522 inhabitants per km$^2$ in 2010), with a total population of 1.1 million people (Statistics Netherlands, 2013f). The province has a varied cultural landscape, which has been intensively managed for many centuries (Berendsen, 2005; Jongmans et al., 2013). Most natural ecosystems have been converted, and those that remain are highly fragmented (Jongman, 2002). Nevertheless, Limburg harbours numerous species of national and even international importance, and provides habitats that are unique in the Netherlands (Willems, 2001; Statistics Netherlands et al., 2008).

### *4.2.2 Criteria for biodiversity accounting*

Biodiversity is inherently spatially explicit and spatial variation is an important aspect of biodiversity (e.g. Noss, 1990; Chiarucci et al., 2011). Spatial explicitness enables assessing spatial distribution and abundance and is an important requirement in the ecosystem accounting framework (UN et al., 2014a). The focus on spatial explicitness of biodiversity indicators allows for compatibility with the various accounts that are suggested within the SEEA-EEA. In this study, we also focus on the spatial relations between various indicators. In this section we assess which criteria biodiversity indicators should meet in order to provide adequate information for biodiversity accounting.

In order to develop biodiversity accounts, indicators need to be selected that fit the purpose of accounting. Accounting for a single indicator would not provide sufficient information on such a multidimensional and complex concept as biodiversity. Therefore there is a need for a set of indicators (Vačkář et al., 2012). A large range of biodiversity indicators exist, addressing a large array of biodiversity aspects (Pereira et al., 2013). Likewise, in scientific literature extensive lists of criteria for selecting and assessing biodiversity indicators have been developed (e.g. Gregory et al., 2005; van Strien et al., 2009; Heink and Kowarik, 2010; Vačkář et al., 2012). Similarly, criteria have been distinguished biodiversity-related for environmental indicators (e.g. Niemeijer and de Groot, 2008; van Oudenhoven et al., 2012). Based on biodiversity indicator literature we identified six key criteria to select and assess indicators for biodiversity accounting based on the criteria developed in literature on biodiversity and ecosystem services. These criteria are conceptually used to analyse the biodiversity indicators we apply in this study.

First, for biodiversity accounting an indicator should be quantitative. A key objective of biodiversity accounting is to monitor trends (Vačkář et al., 2012; UN et al., 2014a), for which indicators have to be quantitative and provide the possibility to track changes (van Strien et al., 2009; Heink and Kowarik, 2010; van Oudenhoven et al., 2012). Quantitative indicators allow for statistical analysis, while sensitivity to change potentially allows for analysing causal relationships. Second, biodiversity indicators need to be feasible in terms of data collection and analysis. Feasibility means that the analysis is repeatable and reproducible (Niemeijer and de Groot, 2008; Heink and Kowarik, 2010) and that the collection and application of indicators is feasible over regular time intervals. Third, universality refers to the wide applicability of an indicator. An indicator should be applicable in at different spatial scales, for different purposes (Heink and Kowarik, 2010) to accommodate different types of accounts, and be flexible so that it can be consistently applied in different regions (van Oudenhoven et al., 2012). Fourth, biodiversity indicators need to be comprehensive, representing a larger part of biodiversity with minimal information loss (van Strien et al., 2009; Heink and Kowarik, 2010; van Oudenhoven et al., 2012). Fifth, indicators need to be credible to users. The information the indicator displays must be acceptable for the end user or decision maker, while maintaining its information value (van Strien et al., 2009; Heink and Kowarik, 2010; van Oudenhoven et al., 2012). Sixth, and related, indicators should be easily understandable for their end users (van Strien et al., 2009; Heink and Kowarik, 2010; van Oudenhoven et al., 2012).

### *4.2.3 Biodiversity aspects covered by biodiversity accounting*

Several components of species diversity could be included in a biodiversity account, for which we tested several indicators. First, a biodiversity account should monitor trends of species and species groups. A general and straightforward way of doing this is studying species richness. Species richness is the most commonly studied measure of biodiversity (Fleishman et al., 2006; Balvanera et al., 2014). It is well-defined, relatively easy and relatively inexpensive to monitor at large scales (Kéry and Plattner, 2007). Besides species richness there are other types of indicators to measure species diversity. There are a wealth of biodiversity indices which include not only the number of species, but also species abundance (e.g. the Shannon-Wiener diversity index, the Simpson's index) (Yoccoz et al., 2001; Chiarucci et al., 2011). Species abundance provides detailed information on stability of community and habitat quality, but is data intensive and difficult to apply at large scales for a wide range of species. In addition to general species trends, rare and threatened species should be accounted for (UN et al., 2014a). One commonly used measure for identifying threatened species are Red Lists (Butchart et al., 2005; Vačkář et al., 2012). Originally developed by IUCN, the Red Lists classify species in terms of their risk of extinction (Butchart et al., 2005). Red List species can be analysed both for species richness as well as abundance. Both single species, specific species groups and composite indicators can be used to acquire a range of information on the status of biodiversity in the studied region.

In this study we addressed several biodiversity indicators which can be analysed spatially. First, we addressed species richness, looking both at richness of single species groups, as well as composite indicators. We studied species rarity and threat by analysing Red List species, as well as the importance of particular areas for rare species. Finally, we analysed species abundance, based on the Shannon-Wiener diversity index. After applying them for Limburg, we qualitatively assessed the indicators based on the six criteria.

### *4.2.4 Data analysis*

To map biodiversity indicators, ESRI ArcGIS software was used. For statistical analysis SPSS was used. We applied Pearson's r to analyse spatial correlation between biodiversity indicators. Pearson's r analyses linear correlation between two variables, and gives values between -1 and 1. Positive values indicate a positive correlation between the variables, negative values indicate a negative correlation and 0 indicates no correlation.

### *4.2.5 Species richness and Red List species*

#### *Available data*

Spatially explicit species richness data was obtained from the Dutch national database for flora and fauna (NDFF) for 14 species groups at a 1 km$^2$ resolution for Limburg (NDFF, 2014b). The dataset included information on species richness per grid cell, the number of Red List species per grid cell and an assessment of the completeness of the data per species group for each cell (NDFF, 2015). The completeness of the data for each cell was assessed by the organisations in charge of data collection, at the time it was made available to the NDFF. The completeness of each cell could be qualified as 'good', 'moderate', 'poor' or 'unknown', depending on the monitoring activity in the area. The species richness dataset did not distinguish between individual species. The data was collected between 2008 and 2012. Appendix II, Table AII.1, shows the full list of species groups and the data completeness and quality for each species group. We used the five species groups with the most complete data to develop the composite indicator BD5: butterflies, vascular plants, birds, dragonflies and amphibians. For all species groups in BD5 at least 75% of the grid cells were assessed for completeness (scores 'good', 'moderate' and 'poor'), and at least 25% of the grid cells had a data completeness of average to good.

In addition, for butterflies a dataset was obtained with a 250x250m resolution from Dutch Butterfly Conservation (De Vlinderstichting) (van Swaay, 2013). The dataset distinguished between individual butterfly species, and it was generated at a 250x250m scale to have a complete coverage of the Netherlands. As real observations on this scale are only available for a minority of the 250x250m grid cells, a combination of real observations, probability maps and occupancy modelling was used. Real observations at 250x250m resolution or smaller were used. Probability maps estimated the probability of a species occurring in a grid cell based on vegetation type, soil type and land use, also at a 250x250m resolution. Occupancy modelling estimated the distribution of species based on a 1 km$^2$ grid cell (van Swaay, 2013). Occupancy maps account for detection bias of an observer, by analysing detection and non-detection data and provide estimates of the percentage of occupied sights (van Strien et al., 2011; van Strien et al., 2013). Based on the three map types five quality classes were set up, of which we applied the highest three: (1) real observations with a precision of at least 250x250m or better, (2) real observations with a precision of at most 1 $km^2$, and (3) a 1x1 km grid cell with an average occupancy score higher than 0.5. In the case of classes 2 and 3 all 250x250m grid cells selected in the probability maps received the same quality score (van Swaay, 2013). We assumed that the butterfly species was present if a 250x250m grid cell had one of these three quality classes, and for each species we developed a presence/absence map. To obtain a butterfly species richness map, all maps for individual species were added together.

### *Species richness indicators*

For all 14 species groups the spatial correlation with all other species groups was tested, in order to identify whether species diversity in one group can be assumed to be representative for biodiversity at large in Limburg. To analyse species richness, four indicators were developed, BD1, BD2, BD5 and BD14. BD1 represented the species richness of the species group with the most complete dataset. For Limburg this was the species group butterflies, for which 95% of the grid cells had an average to high data completeness. The first composite indicator (BD2) comprised of butterflies and birds, two species groups which are commonly used for monitoring biodiversity in the European Union (e.g. EEA, 2012). The second composite indicator (BD5) comprised of the five species groups with the highest data completeness as mentioned in 2.5.1. The final composite indicator (BD14) comprises of all 14 species groups. The three composite indicators for species richness were tested to assess which type of indicator could best reflect the spatial distribution of species richness in a biodiversity account. To develop composite indicators the data was normalised for each included species group, resulting in a score between 0 and 1:

$$S_{ij} = \frac{s_{ij} - s_{min}}{s_{max} - s_{min}} \qquad (1)$$

where $S_{ij}$ is the normalised species richness of species group $i$ in grid cell $j$, $s_{ij}$ is the species richness of species group $i$ in grid cell $j$, $s_{min}$ is the minimum species richness of the species group found in Limburg and $s_{max}$ is the maximum species richness of the species group found in Limburg. The normalised species group summed to develop a composite indicator, after which all composite indicators were normalised for comparability, as follows:

$$BD_{nj} = \frac{\sum S_{ij}}{\sum S_{imax}} \quad (2)$$

where $BD_{nj}$ is the normalised species richness of the composite indicator in grid cell $j$, $n$ represents the number of species groups included in the composite indicator, $S_{ij}$ is the normalised species richness of species group $i$ in grid cell $j$, and $S_{imax}$ is the maximum normalised score of species group $i$. Each species richness indicator was mapped. Spatial correlation analysis was done between each composite indicator and the 14 species groups and the mean Pearson's r was calculated to test how well the composite indicators represented spatial variation of the individual species groups. In addition, spatial correlation analysis was done between the composite indicators. The above method was repeated to analyse the richness of Red List species, using the same four indicators (BD1, BD2, BD5 and BD14).

*Spatial resolution*

To assess the impact of spatial resolution on accounting outcomes, for butterflies we compared the 1 km$^2$ resolution NDFF data (NDFF, 2014b) with the higher resolution (250 m x 250 m) butterfly richness data of de Vlinderstichting (van Swaay, 2013), using Pearson's r. For the analysis we used the original 250x250m resolution Vlinderstichting data, as well as a version of the dataset that was upscaled to 1 km$^2$. We upscaled the dataset to assess how comparable the Vlinderstichting model and the NDFF data were. We applied spatial correlation analysis for all butterfly species, as well as for Red List species. To illustrate differences in results we analysed mean butterfly species richness for eight land cover types in Limburg (grassland, cropland, built-up areas, water, forest, heathland, peatland and other nature), which could be one of the levels at which ecosystem accounts could be set up (c.f. Remme et al., 2014).

### *4.2.6 Rarity of species*

In addition to species richness, rarity of species is a relevant biodiversity component to account for (Noss, 1990). Monitoring areas that are important for rare species could be one approach to monitoring rare species. Using the butterfly dataset from de Vlinderstichting (van Swaay, 2013), we developed a method to assess the importance of individual grid cells (1 km$^2$ resolution) for rare butterfly species in Limburg, similar to the method Crisp et al. (2001) used to map endemism of Australian flora. Each butterfly species was given 1 point, which was divided over the number of grid cells in which it was present, where the maximum number of grid cells for the province was 2484. For example, if a butterfly species if present in 10 grid cells, each grid cell where the species is present gets a score of 0.1 for this species. All other grid cells get a score of 0 for this species. Summing all the scores of the butterfly species per grid cell gave an importance score for rare species per $km^2$. This approach was used both for all butterfly species and for Red List species only.

### *4.2.7 Species abundance*

As a final indicator species abundance was analysed for Red List butterfly species in southern Limburg. A dataset on all observed butterfly and bird species and number of individuals was obtained from the NDFF (NDFF, 2014a). Observations with an accuracy of 2 $km^2$ or smaller were considered, using the centre of the observation as reference for analysis. Species observations were allocated to grid cells with 1 $km^2$ resolution. Species abundance was analysed per grid cell using the Shannon-Wiener diversity index for Red List butterflies, birds and BD2 (birds and butterflies combined) as follows:

$$H_j = -\sum_{i=1}^{R} P_{ij} * lnP_{ij} \tag{3}$$

where $H_j$ is the Shannon-Wiener index value of grid cell $j$, $R$ is the total number of species in grid cell $j$, and $P_{ij}$ is the number of individuals of species $i$ as a proportion of the total number of individuals of all species in grid cell $j$. In order to assess differences in spatial distribution, the outcomes were compared to 1 $km^2$ resolution species richness maps based on Red List species (NDFF, 2014b) using Pearson's r.

## 4.3 Results

### *4.3.1 Species richness*

Spatial correlations between richness of different individual species groups are generally not high in Limburg, with Pearson's r varying from 0.07 to 0.62 (Table 4.1). *Grasshoppers and crickets* have the highest mean spatial correlation with the 13 other species groups (Pearson r = 0.41), followed by *butterflies* (Pearson r = 0.40). *Lichens* (Pearson r = 0.18) and *mosses* (Pearson r = 0.20) have the lowest mean spatial correlations with other species groups. The correlation analysis results imply that it is difficult to include a single species group in the accounts and use this as an overall indicator for biodiversity. Even if, in the case of Limburg, *grasshoppers and crickets* are chosen as indicator species for the account (which have the highest mean correlation with other species), high biodiversity in the many cells that are rich in other biodiversity but low in *grasshopper and cricket* biodiversity would not show up in the account. Moreover, information on the spatial distribution of one species group does not provide information on the number of species available from other species groups.

The normalised species richness indexes are shown in Figure 4.1. The general spatial patterns between the different species richness indicators are similar, with similar areas with the highest normalised scores. Spatial correlation between the four composite species richness indicators is high (Pearson's r between 0.75 and 0.94, see Appendix II, Table AII.2). However, the intensity and concentration of hotspots differ between the indicators. The BD2 and BD5 indicators show larger areas with relatively high values, especially compared to BD1. BD14 has smaller, more highly concentrated areas of high values compared to other indicators.

**Table 4.1** Pairwise Pearson correlation matrix for 14 species groups, and mean correlation. Red indicates the lowest correlations found, green indicates the highest correlations found.

| | Butterflies | Vascular plants | Dragonflies | Birds | Amphibians | Mosses | Lichens |
|---|---|---|---|---|---|---|---|
| Butterflies | - | 0.39 | 0.56 | 0.51 | 0.51 | 0.18 | 0.19 |
| Vascular plants | | - | 0.28 | 0.39 | 0.39 | 0.20 | 0.19 |
| Dragonflies | | | - | 0.51 | 0.61 | 0.19 | 0.18 |
| Birds | | | | - | 0.37 | 0.07 | 0.09 |
| Amphibians | | | | | - | 0.24 | 0.16 |
| Mosses | | | | | | - | 0.12 |
| Lichens | | | | | | | - |
| Mushrooms | | | | | | | |
| Mammals | | | | | | | |
| Macro moths | | | | | | | |
| Micro moths | | | | | | | |
| Grasshoppers and crickets | | | | | | | |
| Other vertebrae | | | | | | | |
| Reptiles | | | | | | | |

**Table 4.1 (continued)**

| | Mushrooms | Mammals | Macro moths | Micro moths | Grasshoppers and crickets | Other vertebrae | Reptiles | Mean |
|---|---|---|---|---|---|---|---|---|
| Butterflies | 0.28 | 0.39 | 0.38 | 0.31 | 0.59 | 0.45 | 0.43 | *0.40* |
| Vascular plants | 0.18 | 0.48 | 0.31 | 0.27 | 0.32 | 0.33 | 0.31 | *0.31* |
| Dragonflies | 0.31 | 0.33 | 0.31 | 0.27 | 0.62 | 0.42 | 0.41 | *0.38* |
| Birds | 0.16 | 0.42 | 0.21 | 0.18 | 0.47 | 0.34 | 0.35 | *0.31* |
| Amphibians | 0.31 | 0.36 | 0.34 | 0.23 | 0.52 | 0.35 | 0.52 | *0.38* |
| Mosses | 0.32 | 0.17 | 0.20 | 0.19 | 0.22 | 0.28 | 0.16 | *0.20* |
| Lichens | 0.15 | 0.14 | 0.21 | 0.22 | 0.28 | 0.21 | 0.23 | *0.18* |
| Mushrooms | - | 0.23 | 0.24 | 0.22 | 0.37 | 0.46 | 0.25 | *0.27* |
| Mammals | | - | 0.27 | 0.21 | 0.32 | 0.32 | 0.25 | *0.30* |
| Macro moths | | | - | 0.59 | 0.36 | 0.38 | 0.30 | *0.31* |
| Micro moths | | | | - | 0.31 | 0.50 | 0.23 | *0.29* |
| Grasshoppers and crickets | | | | | - | 0.50 | 0.47 | *0.41* |
| Other vertebrae | | | | | | - | 0.31 | *0.37* |
| Reptiles | | | | | | | - | *0.32* |

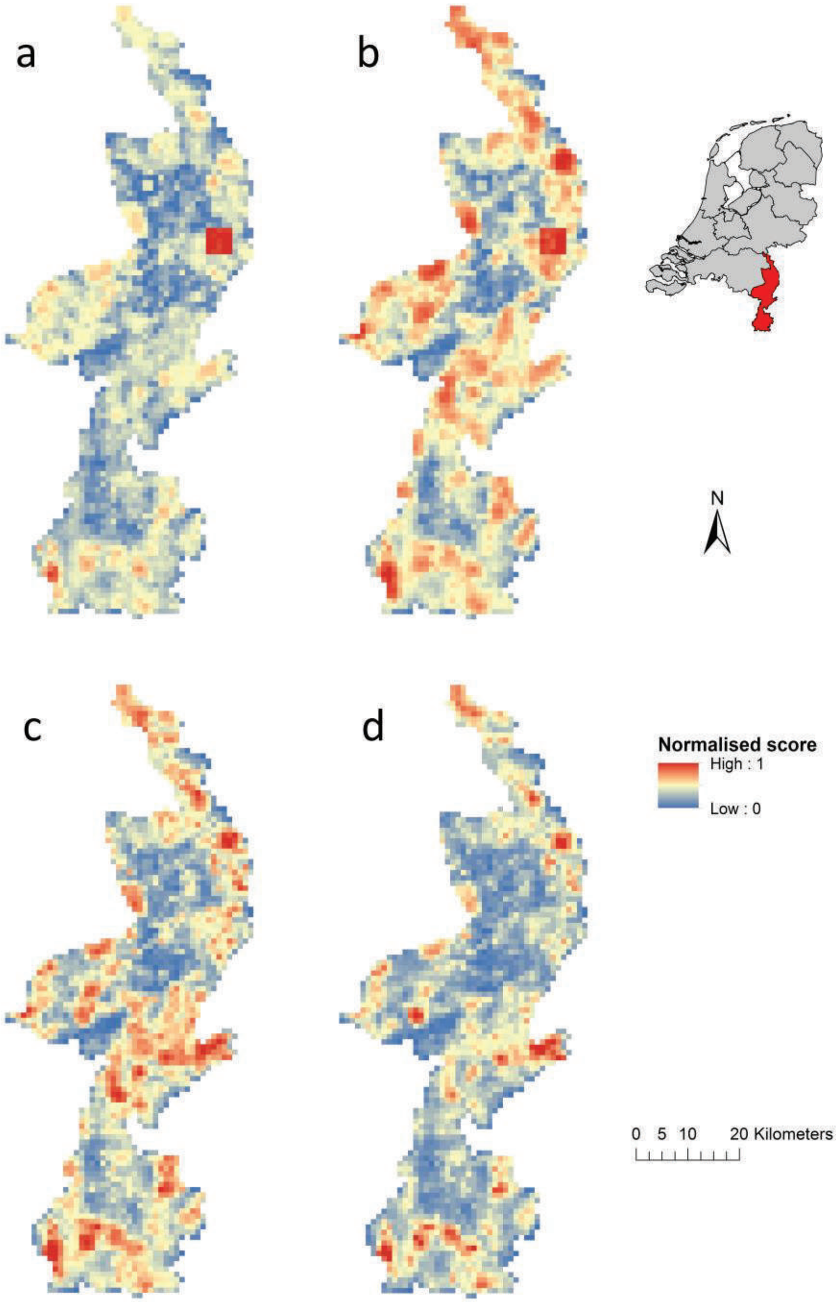


**Figure 4.1** Normalised maps of species richness for (a) BD1 (butterflies) (b) BD2 (birds and butterflies), (c) BD5 (butterflies, vascular plants, birds, dragonflies and amphibians), and (d) for BD14 (all 14 species groups).

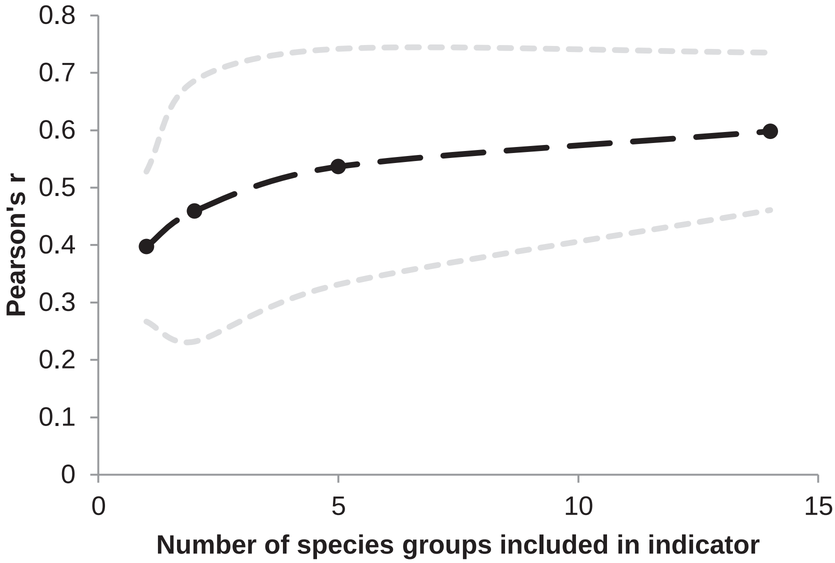


**Figure 4.2** Average Pearson's r in relation to the number of species groups included in the indicator. Grey lines indicate the margins of uncertainty (standard deviation).

Using an increasing amount of species groups for a composite indicator increases the spatial correlation with the individual species groups (Figure 4.2). The butterfly indicator (BD1) has the lowest mean spatial correlation with the separate species groups (Pearson's r = 0.40), while the composite indicator for BD14 has the highest spatial correlation (Pearson's r = 0.60). Although composite indicators increase overall spatial correlation with all species groups, the overall increase compared to a single species indicator is small. Composite indicators still do not cover a large part of species diversity, but do provide a more comprehensive indicator for species diversity than a single species group for biodiversity accounting. These results are relevant for deciding which indicators to include in biodiversity accounting. They show that to capture the spatial distribution of species richness, using a limited number of species groups could be sufficient. In our case, the Pearson's r between BD5 and BD14 is 0.94, meaning that data on five species groups captures nearly the same spatial distribution as data on fourteen species groups. Accounting for five species groups would in this case increase cost-effectiveness.

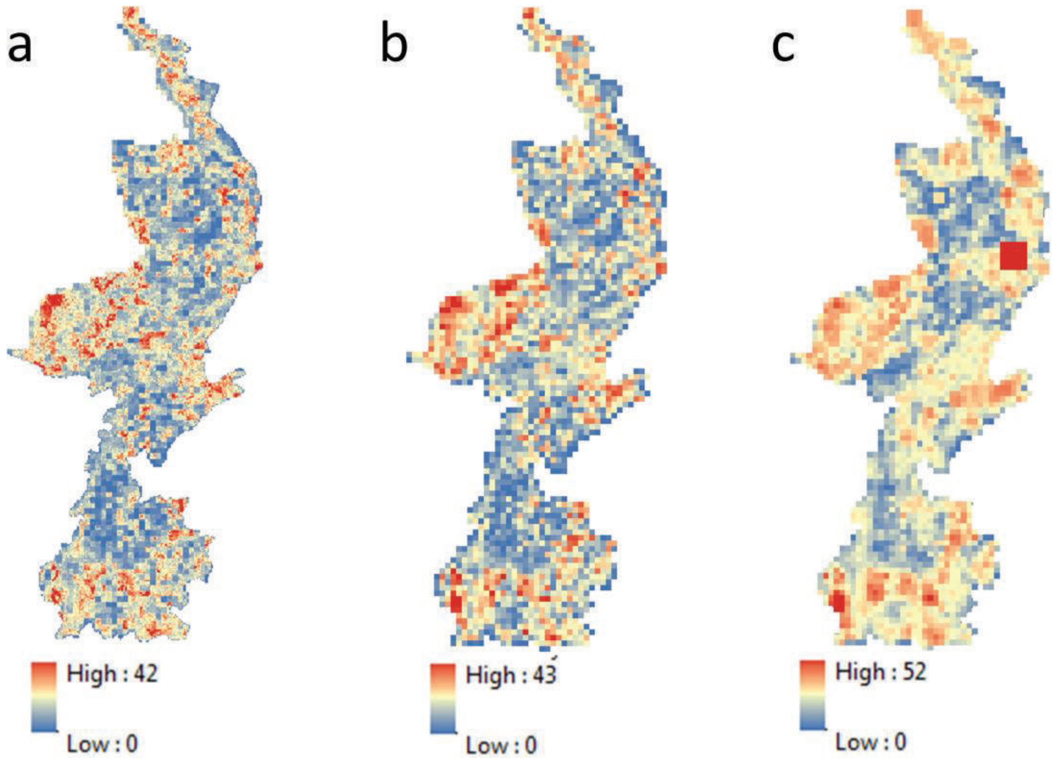


**Figure 4.3** Species richness for butterflies based on (a) the Vlinderstichting model with 250 m resolution (b) the Vlinderstichting model upscaled to 1 km resolution and (c) the NDFF data with 1 km resolution. Species richness for Red List butterfly species based on (d) the Vlinderstichting model with 250 m resolution (e) the Vlinderstichting model upscaled to 1 km resolution and (f) the NDFF data with 1 km resolution.

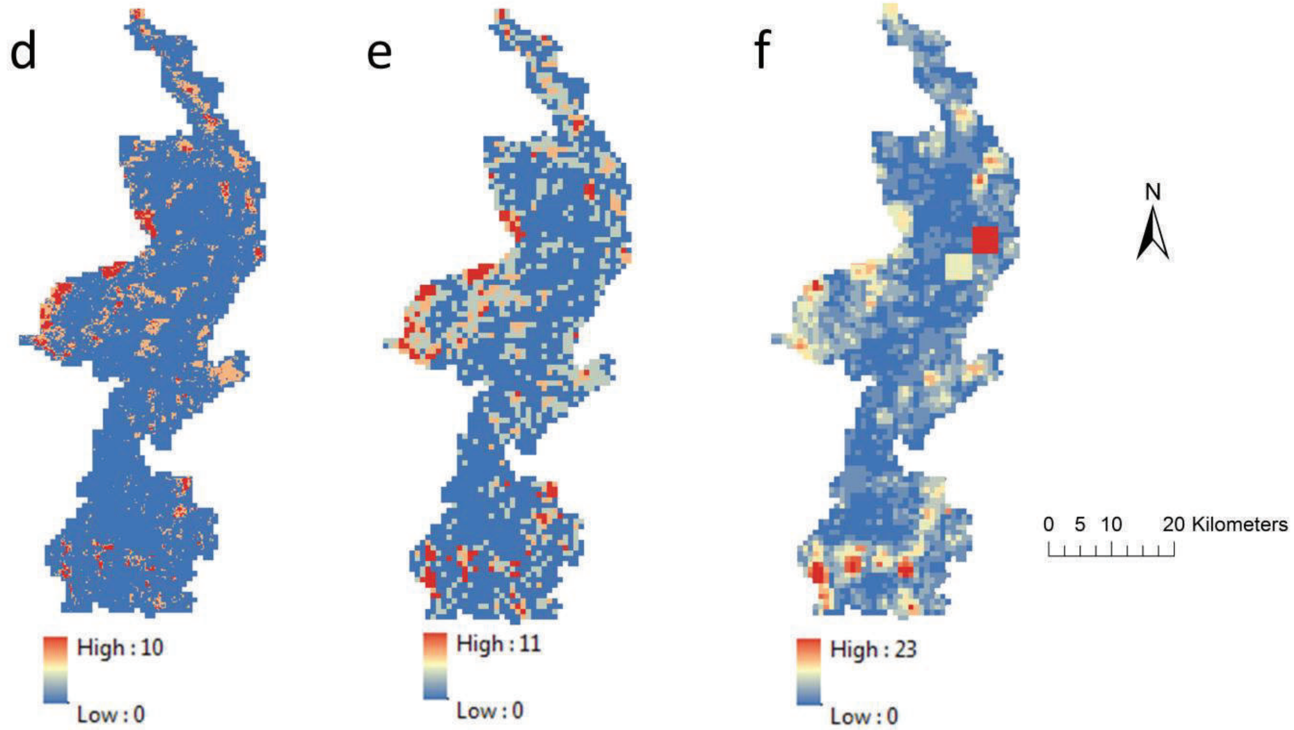


Figure 4.3 (continued)

### *4.3.2 The importance of data resolution*

The Vlinderstichting and NDFF datasets for butterfly species richness with different resolutions were compared spatially (Figure 4.3). The maps of higher resolution Vlinderstichting data (Figure 4.3a and 4.3d) show more concentrated areas with high numbers of species than the 1 $km^2$ resolution maps from the NDFF data (Figure 4.3c and 4.3f). The maps of high resolution Vlinderstichting data show more hotspots, but show a similar pattern in coldspots as the lower resolution maps from NDFF data. Spatial correlation between the high resolution Vlinderstichting model and the NDFF data was intermediate for species richness (Pearson r = 0.54) and poor for Red List species (Pearson r = 0.26). The upscaled Vlinderstichting model for all butterflies (Figure 4.3b) shows spatial similarities with both the high resolution Vlinderstichting model (Pearson r = 0.85) and the NDFF data (Pearson r = 0.66), but less so for Red List species. The mean butterfly richness per grid cell is higher in the NDFF dataset that in the Vlinderstichting model, both for all butterfly species, as well as for Red List species (Figure 4.4 and Appendix II, Table AII.3). Figure 4.4 shows that for all land covers using the 1 $km^2$ resolution results in a higher species richness. The trend between land cover types remains quite similar for the three map sets, with the more natural land covers showing higher mean species richness than the more human dominated land covers. The highest butterfly species richness was found in peatland and the lowest in cropland. For all land cover types the NDFF dataset the mean species richness per $km^2$ was higher than for the upscaled Vlinderstichting model, which is the result of the higher number of observations in the NDFF dataset. Overall, the results show that the application of different datasets results in different outcomes in terms of richness per $km^2$ and in terms of spatial distribution. The 250x250m resolution maps appear to more sharply delineate locations important for butterflies and are more informative for biodiversity accounting than maps with a coarser resolution, as landscape units in Limburg are often heterogeneous at less than 250x250m. The analysis shows that the chosen resolution for biodiversity accounting can strongly influence the outcomes. In addition, the results show that the resolution of the biodiversity indicators should match landscape heterogeneity as much as possible.

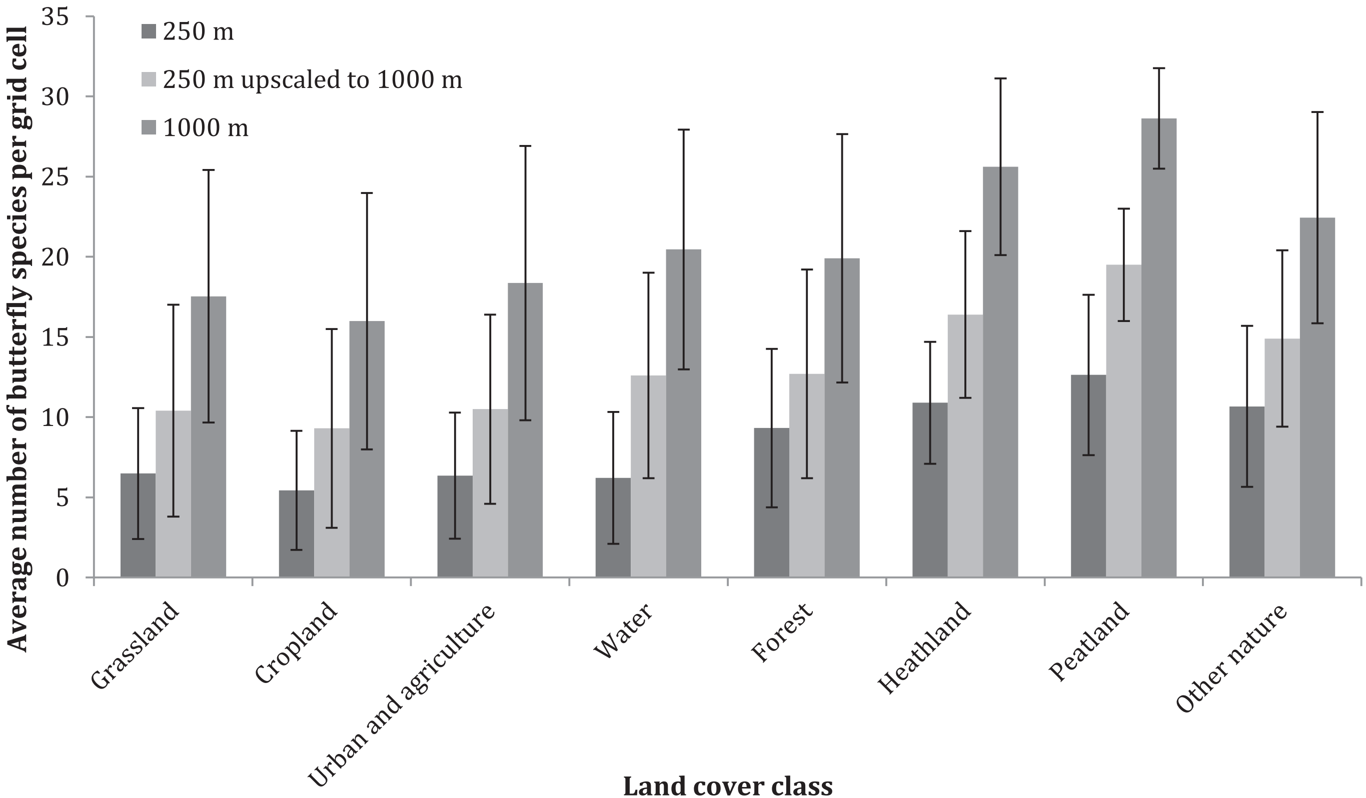


**Figure 4.4** Mean number of butterfly species per land cover class for the Vlinderstichting data (250 m resolution, and upscaled to 1000 m resolution) and the NDFF data (1000 m resolution, red). Error bars indicate standard deviation.

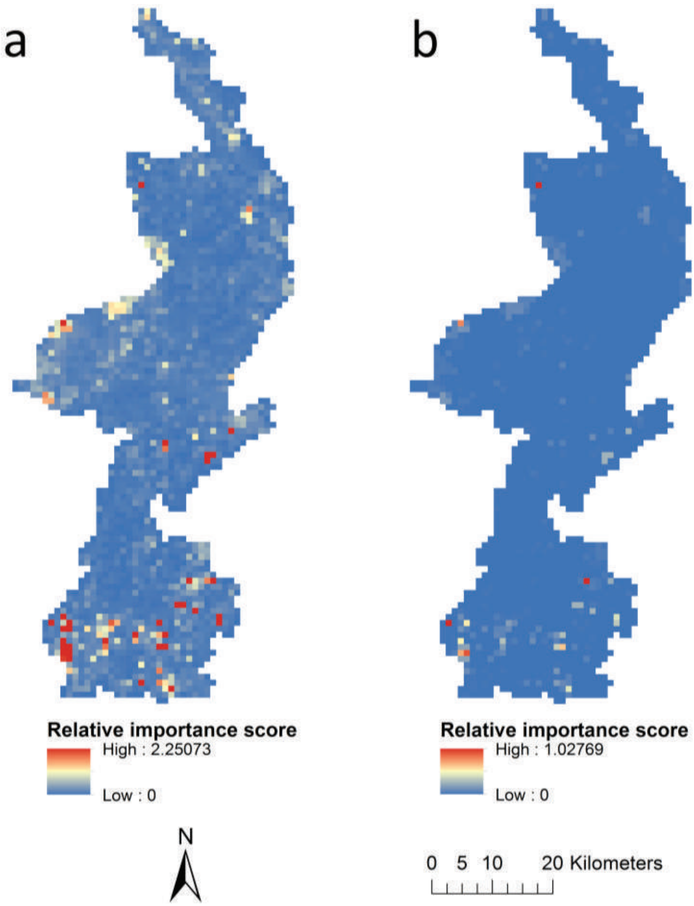


**Figure 4.5** Maps of importance of areas for (a) all butterfly species and (b) Red List butterfly species.

### *4.3.3 Importance of areas for rare species*

The most important areas for rare butterfly species are mostly found in the south of Limburg (Figure 4.5). There is a small amount of grid cells with very high scores (9 cells with scores above 0.5), as a result of a small number of rare butterfly species occurring in Limburg. The central region of the province scores low, both for all butterfly species, as well as for Red List species. Figure 4.5 shows that, with the exception of some overlapping hotspots, the spatial distribution of important areas between all butterfly species and Red List species is clearly different. When assessing which areas are important for species in biodiversity accounts, it would be more informative to include an indicator for a broader range of species, rather than only Red List species. Only accounting for Red List species would be too limited, as there are too few Red List butterfly species in Limburg to reflect the importance of different habitats.

There were no strong correlations between species richness indicators and the rarity indicator (Table 4.2). Spatial correlation was highest between species richness of 14 species groups and the importance of areas for all butterfly species (Pearson's r = 0.32) Spatial correlation between species richness indicators and importance of areas for rare Red List butterflies was lower than importance for all rare butterfly species. The clear differences in spatial correlation between species richness and important areas shows that both types of indicators would provide different, but complementary, sets of information for biodiversity accounting.

**Table 4.2** Pearson's r for correlation between species richness indicators and rarity indicators for butterflies.

| Species richness indicators | Rarity indicators for butterflies | |
|---|---|---|
| | All butterflies | Red List butterflies |
| Butterflies | 0.30 | 0.25 |
| Red List butterflies | 0.29 | 0.26 |
| BD2 | 0.24 | 0.20 |
| BD5 | 0.28 | 0.23 |
| BD14 | 0.32 | 0.27 |

### *4.3.4 Species abundance*

The spatial distribution for the Shannon-Wiener index for birds and butterflies is very heterogeneous in southern Limburg (Figure 4.6). For butterflies the Shannon-Wiener index shows only a few cells with high diversity. The Shannon-Wiener index shows an intermediate correlation with species richness, based on species observation data of birds and butterflies (NDFF, 2014a). Butterfly abundance shows the strongest correlation with butterfly species richness (Pearson's r = 0.61). Birds have a strong influence on the spatial distribution of both abundance and species richness of BD2. The results yield interesting lessons for biodiversity accounting. Indicators for species abundance and species richness show clear different spatial patterns, meaning that both indicator types provide different information for accounts. Also, combining different species groups for a composite species abundance indicator (the Shannon-Wiener index, in our case) could dilute information on the less dominant species group. Therefore, it could be relevant to account for species abundance of different species groups separately.

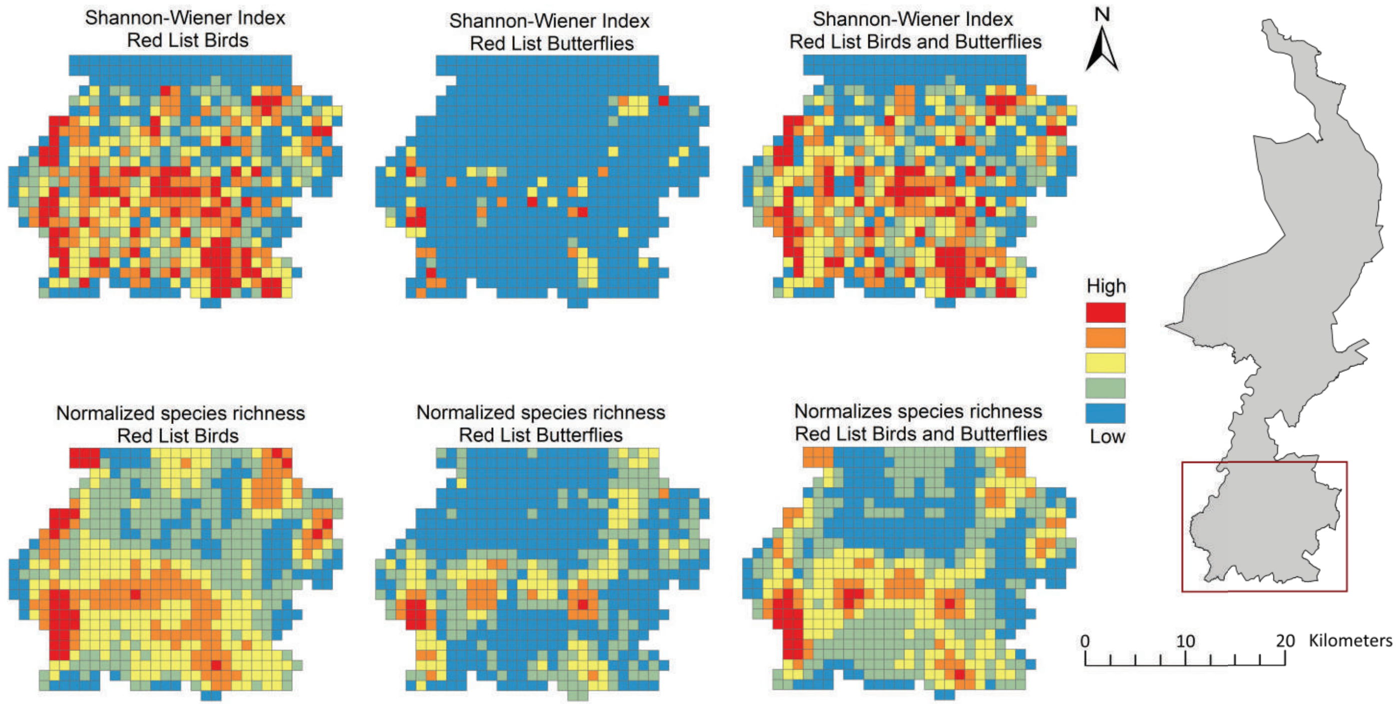


**Figure 4.6** Distribution of Red List bird and butterfly species for southern Limburg at 1 km² resolution measured using (a) the Shannon-Wiener index and (b) species richness. For birds Pearson's r = 0.50, for butterflies Pearson's r = 0.61 and for birds and butterflies combined Pearson's r = 0.53.

## 4.4 Discussion

### *4.4.1 Biodiversity accounting indicators*

The indicators we analysed yielded clear differences in terms of spatial distribution, but they also have clear differences in terms of which biodiversity accounting criteria they meet. In this section, we shortly synthesize our findings based on the six criteria we proposed. We then proceed to assess the indicators for biodiversity accounting in a broader context.

#### *Assessment of individual indicators*

Although all applied indicators were in principle quantifiable, species richness was the most straightforward, at least for single species groups. The indicator is expressed in absolute values, although normalisation is advisable when combining species groups. The other tested indicators are always expressed in relative values. Species richness for a single species group is the most feasible indicator to monitor because it requires relatively little data. The indicator for important areas requires the same type of data, but requires additional calculation. Nevertheless, feasibility of this indicator is also relatively high. Indicators that include abundance require substantially more and more detailed data and more complex analysis. Species richness indicators have been widely applied in many regions around the world and at many different spatial scales, and remain unaffected by changes in the size of a research area. Relative indicators such as the important areas and abundance indicators applied here, are more strongly affected by the spatial extent and boundaries of the analysis and are less comparable between areas, unless the same set of species is used. In terms of comprehensiveness, a more ecologically complex indicator, such as the Shannon-Wiener index is more comprehensive than the other indicators because it includes information on species abundance. It is also the most ecologically credible indicator used as it takes into account multiple aspects of biodiversity (i.e. presence and abundance) (Chiarucci et al., 2011). The difference in quantitative approach between species richness and the other indicators (i.e. absolute vs. relative values) makes species richness more understandable for the general public. Species richness of a single species group is most understandable, because species numbers can be easily communicated.

When dealing with spatially explicit indicators for accounting, some additional aspects need to be considered. Our research clearly shows that data resolution can affect biodiversity accounting output. High resolution data increases the level of detail at which an area can be analysed. However, for Limburg, increasing the resolution also decreased the measured species per area unit. The applied scale correction did not compensate for this lower count in species richness. Discrepancies between resolutions of input data and boundary effects affect biodiversity accounts (Bond et al., 2013; UN et al., 2014a). Also, when applying a dataset for a single species or species group indicator and applying this same dataset in a composite indicator, autocorrelation should be assessed (Bond et al., 2013), determining to what extent the dataset affects the clustering of both indicator types.

### *Biodiversity accounting with a set of indicators*

In this study we have tested a range of different spatial indicators and aspects that are of importance for biodiversity accounting. Species richness was the easiest indicator to account for, as data for this indicator is readily available, i.e. the feasibility of this indicator type is high. This was the case for Limburg, but is also true for many species globally (e.g. for vertebrates the Living Planet Index, WWF, 2014). However, accounting for species richness alone does not provide sufficient information on the state of biodiversity in Limburg province. Involving multiple biodiversity indicators with different qualities in accounting allows for a wider coverage of the criteria. For example, while species richness is a biodiversity indicator which is quantitative, readily available, feasible to monitor and understandable for the general public, ecologists state that there are more informative biodiversity indicators (Fleishman et al., 2006; Feld et al., 2009). Measures for species abundance, such as the Shannon-Wiener index, are ecologically more credible, but are less understandable for the general public and data is often not widely available (Chiarucci et al., 2011). By combining these different indicators a more comprehensive account can be developed. On the one hand, species richness can be a useful biodiversity indicator in combination with other metrics, such as distribution of abundance and rarity (Fleishman et al., 2006). On the other hand, some indicators can provide duplicate information, which could make some indicators redundant. For example, for species richness in Limburg little difference was found between the spatial distribution of all species and the spatial distribution of Red List species. Therefore, accounting only for species richness of Red List species could be sufficient, also providing an indication of the general species richness distribution. However, it should be noted that Red Lists are regularly updated and therefore change, which could lead to inconsistencies in biodiversity accounts. Species richness indicators and abundance and distribution indicators such as the Shannon-Wiener index complement each other. The strengths of Shannon-Wiener index compensate for the weaknesses of species richness indicators (comprehensiveness and credibility) and vice versa (easily quantifiable, feasible and understandable). In addition, the necessary data to derive the indicators partially overlap, i.e. by collecting data on species abundance, species richness data is inherently also collected. Therefore, when developing biodiversity accounts, it would be advisable to incorporate both species richness indicators and species abundance indicators. Rarity indicators were spatially very different from species richness. Therefore, also the rarity indicators provide additional spatial information for biodiversity accounting, showing specific hotspot areas.

### *4.4.2 Illustrating the use of biodiversity accounting*

Biodiversity accounts have been applied, for instance, to inform companies that want to apply conservation measures (Jones, 2003; Rimmel and Jonäll, 2013). To illustrate what a species account could look like, Bond et al. (2013) present accounting tables for the Great Barrier Reef Region and Victoria in Australia. However, besides the development of accounting tables, the applicability of the spatial perspective of biodiversity accounts needs to be further explored. Here we briefly address one potential application of spatial biodiversity accounts. We apply the addressed biodiversity indicators to assess the difference between biodiversity inside and outside a conservation network. In Limburg, conservation areas included in the Dutch national ecological network (NNN, Natuurnetwerk Nederland) have higher species richness than unprotected areas outside the network (Figure 4.7). The total mean species richness is 33% higher inside the NNN than in the rest of Limburg. The mean species richness is significantly higher inside the NNN than in the rest of Limburg for all 14 measured species groups. Similar results were found for Red List species (not shown). In addition, the rarity scores were twice as high inside the NNN as in the rest of Limburg (not shown). The analysis shows that the ecological network is positively related to species diversity in Limburg, confirming that the protected status of the areas in this network contributes to the conservation of their biodiversity. This finding is in line with other studies on the NNN that conclude that the biodiversity is highest inside the network (Strijker et al., 2000; Jagers op Akkerhuis et al., 2006). If such an analysis is repeated in the future, changes can be assessed and related to implemented policies, to analyse which policies have been successful or detrimental to biodiversity. Including multiple biodiversity indicators would give further information on which areas need additional attention.

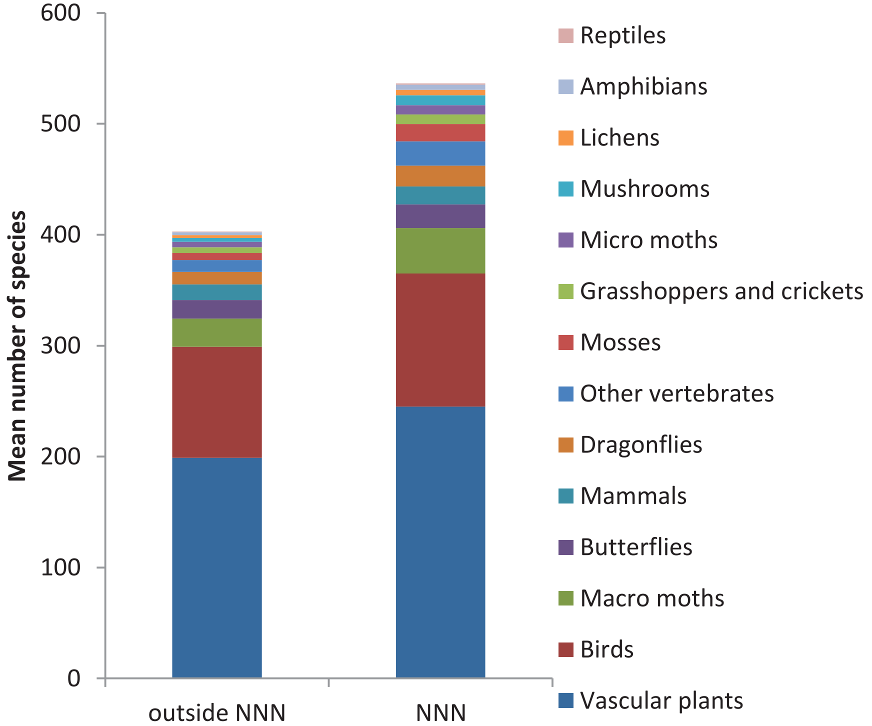


**Figure 4.7** Mean species richness outside and inside the Dutch national ecological network NNN. Means for all species groups are significantly higher within the NNN than outside the NNN ($P < 0.001$).

### *4.4.3 Developing biodiversity accounts*

*Biodiversity accounting in the Netherlands*

In this study we have focussed on accounting for species diversity from a spatial perspective using a limited number of indicators, as was the focus of prior biodiversity accounting studies (e.g. Bond et al., 2013; Jones and Solomon, 2013). The focus is strongly related to the value humans give to the existence of biodiversity i.e. biodiversity as a cultural ecosystem service (Mace et al., 2012; Reyers et al., 2012). For this study we had access to a subset of data on biodiversity in Limburg province, the Netherlands. The Netherlands has a rich tradition in monitoring species, and many organisations exist that record and monitor specific species groups. Therefore, at a national scale data on temporal trends exist for several species groups (see for example Statistics Netherlands et al., 2014). National organisations and initiatives such as the Network for Ecological Monitoring (NEM), Statistics Netherlands and the Compendium voor de Leefomgeving frequently publish trends in biodiversity, both composite indicators as well as for single species. The large collected datasets have been used to study aspects of biodiversity, such as the applicability of single species groups as indicators for multiple species groups (Musters et al., 2013). Involving such initiatives in developing national biodiversity accounts as part of an integrated ecosystem accounting system would vastly increase the type of information that could be included in such accounts.

*Biodiversity accounting in an international context*

We acknowledge that data availability in Limburg is very high compared to many other regions in the world. We are therefore not suggesting that the approach we applied is equally applicable in other parts of the world. The ultimate goal would be to develop biodiversity accounts that can be adapted and used in all countries and at all spatial extents (Bond et al., 2013). We postulate that a flexible approach to biodiversity accounting is required so that countries can adjust their accounts to the data that is available. It is more important that accounts are comparable over time within a country or region than that biodiversity accounts are comparable between countries. A flexible approach would facilitate which biodiversity indicators can be included in accounts, focussing on which data is available and can be regularly collected for monitoring. This paper presents a number of lessons in this context.

First, if data are available, a selection of indicators needs to be made. Selecting indicators that can represent other aspects of biodiversity increases cost-effectiveness of an account, while still capturing spatial and temporal patterns in biodiversity. For example, in the case of Limburg, the richness of five species groups can be used as a proxy for the richness of fourteen species groups. The criteria we distinguished can guide such choices and can ensure that multiple perspectives on biodiversity are covered in accounts. Second, for countries that intend to develop accounts, our paper shows potentially relevant indicators that indicate priorities for which indicators to include in such accounts and for additional data collection. For example, in order for ecosystem accounting to be applied in regions with lower data availability than Limburg, more straightforward and widely available indicators such as species richness and Red List species can be included in biodiversity accounts. In combination with ecosystem condition accounts (UN et al., 2014a;

Hein et al., 2015), such indicators can give an indication of the general trends in ecosystems and biodiversity of a region. Indicators for important areas can be developed based on species richness data. Such indicators could be enriched by assessing relationships between species richness and land cover. Indicators for species abundance provide important information for biodiversity accounts, but their inclusion will probably require investments in biodiversity monitoring schemes in many countries. Multiple indicators should be included in accounts, in order to monitor different aspects of biodiversity. The main issue that remains here is the selection of the most adequate set of indicators, based on the available knowledge and data. To define a basic set of indicators that is applicable in a wide range of regions additional research is needed in other areas. Whether such a set of indicators can be developed remains to be seen. It is likely that the set of indicators that provides the most comprehensive information on biodiversity could differ between different areas (Jones and Solomon, 2013), not only due to data availability, but also depending on for example specific important species groups and threatened species.

The incorporation of information from existing biodiversity monitoring schemes is key to creating comprehensive biodiversity accounts. In Europe there are many biodiversity monitoring schemes in place (Geijzendorffer and Roche, 2013), but also globally spatial biodiversity datasets exist, e.g. the Map of Life (www.mol.org) and the Living Planet Index (WWF, 2014). Pereira et al. (2013) call for the development of a set of Essential Biodiversity Variables in order to consistently monitor a range of biodiversity components globally. The complementarity between the mentioned initiatives and programs, and the SEEA-EEA biodiversity accounting approach needs to be further examined. Global efforts to synthesize biodiversity information are increasing, and also national efforts are underway in different countries (e.g. the Netherlands (NDFF) and Australia (Bond et al., 2013)). Technological advancements, such as improved remote sensing techniques, will provide further opportunities to collect indicators for biodiversity (Pereira et al., 2013). This can lead to the further development of indicator sets for biodiversity accounts over time.

### *Broadening the scope of biodiversity accounting*

The biodiversity accounting approach as currently proposed by the SEEA-EEA can be expanded based on extensive biodiversity research. Besides species diversity, other biodiversity aspects could prove to be crucial additions to biodiversity accounts, to capture important changes and pinpoint relations. For example, research on the links between international trade and their impact on biodiversity can be made more explicit (Lenzen et al., 2012). By monitoring the effects of trade between countries on biodiversity threats, not only domestic, but also transboundary effects are accounted for. Such biodiversity analysis connects to the objectives of the SEEA-EEA to measure and monitor the relationship between ecosystems and economic activity (UN et al., 2014a). Also, the link between biodiversity and land use or specific habitats can be further incorporated (Bond et al., 2013). Another addition could be the inclusion of indicators such as the Biodiversity Intactness Indicator (Scholes and Biggs, 2005), or Mean Species Abundance (Alkemade et al., 2009), that assess changes in biodiversity compared to a reference state.

An important aspect of biodiversity that is often discussed in relation to ecosystem services, is functional diversity. Functional diversity is the aspect of biodiversity that influences ecosystem properties and also enables ecosystems to provide services (Hooper et al., 2005). Functional diversity can be a critical underlying component for the provision of many different ecosystem services (Díaz et al., 2006; Mace et al., 2012). Existing biodiversity accounting efforts focus on species diversity, and do not cover functional diversity. Ecosystem accounting according to the SEEA-EEA guidelines (partially) covers other aspects of biodiversity already, such as ecosystem diversity and functional diversity, namely in condition accounts (UN et al., 2014a; Hein et al., 2015), and to a lesser extent in the ecosystem service accounts, by including services that are directly related to biodiversity (Remme et al., 2014; Schröter et al., 2014b). Accounting for functional diversity requires improved understanding of the underpinning role of biodiversity in the provision of the different types of ecosystem services, for which strong evidence is not always available (Kremen, 2005; Cardinale et al., 2006; Ridder, 2008). Functional diversity accounts would require a more integrated approach with other ecosystem accounts, especially related to ecosystem condition and the capacity of the system to provide services. In order to mainstream biodiversity accounting, the initial focus should be on species diversity. Nevertheless, the inclusion of functional diversity and more complex biodiversity metrics should be further explored, in order to get a more encompassing picture of the relationship between biodiversity and ecosystem services.

## 4.5 Conclusion

We explored how information on biodiversity can be included in a biodiversity account, using a spatial perspective. We show that for Limburg province the spatial variation between species groups is large. Spatial distribution of a single species group can only capture a small portion of the overall spatial distribution of other species groups. Applying a composite indicator that includes a small number of species groups is capable of capturing nearly as much of the spatial variation as all species groups combined. Accounting for the spatial variation of a selective number of well-monitored species groups can significantly decrease the data needs of biodiversity accounting efforts. We show that incorporating multiple types of biodiversity indicators into accounts can provide complementary information, which can assist policy makers to make informed decisions. In the development of biodiversity accounts, indicators should be included that are ecologically significant, but also indicators that are understandable for the general public. The choice for resolution strongly influences the spatial outcomes of biodiversity accounting. Although our study shows that aggregated variation between land cover types remain comparable when analysed at different resolutions, spatially the correlation between the outcomes at different spatial resolutions is weak. The explored set of species diversity indicators provide relevant, complementary information to develop biodiversity accounts, and can be used as a starting point to development more extensive accounts. The inclusion of functional diversity indicators would provide additional information that would increase the comprehensiveness of biodiversity accounts. Our case study shows that by combining multiple indicators, species diversity can be monitored in a comprehensive way, addressing species richness, threatened species, species abundance as well as the importance of specific areas for conservation.

## Acknowledgements

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# Chapter 5 - How budget constraints affect conservation network design for biodiversity and ecosystem services: optimizing planning in Limburg, the Netherlands


## Abstract

Limited budgets and budget cuts hamper the effective development of biodiversity conservation networks. Optimizing the spatial configuration of conservation networks given such budget constraints remains challenging. Systematic conservation planning addresses this challenge. Furthermore, this planning approach can integrate both biodiversity and ecosystem services as conservation targets, and hence address the challenge to operationalize ecosystem services as an anthropocentric argument for conservation. We create two conservation scenarios to expand the current conservation network in the Dutch province of Limburg. One scenario focuses on biodiversity only and the other integrates biodiversity and ecosystem services. We varied conservation budgets in these scenarios and used the Marxan software to assess differences in the resulting network configurations. In addition, we tested the network's cost-effectiveness by allocating a conservation budget either in one or in multiple steps. We included twenty-nine biodiversity aspects and five ecosystem services. The inclusion of ecosystem services to expand Limburg's conservation network only moderately changed prioritized areas, compared to only conserving biodiversity. Network expansion in a single time-step is more efficient in terms of compactness and cost-effectiveness than implementing it in multiple time-steps. Therefore, to cost-effectively plan conservation networks, the full budget should ideally be available before the plans are implemented. We show that including ecosystem services to cost-effectively expand conservation networks can simultaneously encourage biodiversity conservation and stimulate the protection of conservation-compatible ecosystem services.



Based on:
Remme, RP, Schröter, M. How budget constraints affect conservation network design for biodiversity and ecosystem services: optimizing planning in Limburg, the Netherlands. *Ecological Complexity* (submitted).

## 5.1. Introduction

Creating protected areas is a much seen strategy for biodiversity conservation (Rands et al., 2010). For instance, Natura 2000 sites have been established throughout Europe to conserve biodiversity as part of the European Union (EU) Biodiversity Strategy (European Commission, 2011). In addition, EU member states have made their own efforts to conserve and manage biodiversity. For example, the Netherlands has been developing the National Ecological Network since the 1990s to connect protected areas and to enhance the mobility of species (LNV, 1989). In recent years, severe governmental budget cuts have hampered the completion of this national ecological network (Buijs et al., 2014) and individual provinces must now arrange their own spatial conservation efforts. Limited budgets often constrain current conservation efforts (James et al., 1999; Brooks et al., 2006). Optimizing the spatial configuration of the expansion of such conservation networks under constrained conservation budgets is challenging, in particular in face of other societal and economic interests in land use. Accounting for conservation costs, such as costs for acquiring land (opportunity costs), can improve the effectiveness of conservation planning (Naidoo et al., 2006). Systematic conservation planning is an approach to address this challenge. The approach systematically identifies surrogates for conservation features (biodiversity and ecosystem services), sets quantitative and operational targets, recognizes how these targets can be met by conservation areas and uses explicit, yet simple, methods to locate and design conservation areas (Margules and Pressey, 2000; Margules et al., 2007; Moilanen et al., 2009).

Traditionally, conservation efforts, such as the creation of protected areas, have focused on biodiversity (Cimon-Morin et al., 2013; Castro et al., 2015). However, ecosystem services (ESs), which are defined as the contributions of ecosystems to human well-being (Haines-Young and Potschin, 2010b), have been introduced as an additional argument for conservation (Armsworth et al., 2007; Chan et al., 2011; Schröter et al., 2014a). Increasing amounts of quantitative information are being gathered to spatially model ESs (Maes et al., 2012b; Martínez-Harms and Balvanera, 2012; Nemec and Raudsepp-Hearne, 2013; Schägner et al., 2013). Spatial ES models are increasingly being used for ecosystem accounting, i.e. the systematic, spatially explicit monitoring of ES provision (Schröter et al., 2014b; Remme et al., 2015; Sumarga et al., 2015). However, appropriate policy purposes and applications of accounting still need to be further explored (Schröter et al., 2015). One such application could be systematic conservation planning. Biodiversity conservation networks could potentially both conserve and enhance the provision of specific ESs (Castro et al., 2015). Additionally, the inclusion of ESs in systematic conservation planning could well improve biodiversity conservation (Cimon-Morin et al., 2013), as important areas for ES conservation could provide additional areas to conserve biodiversity. However, including ESs in systematic conservation planning is a relatively recent, and yet underdeveloped research field (Chan et al., 2011; Schröter et al., 2014c) that requires further research, especially given the complex relationship between ES and biodiversity (Mace et al., 2012; Balvanera et al., 2014). For example, a distinction should be made between conservation-compatible ESs and ESs that are not compatible with biodiversity conservation (Chan et al., 2011). Conservation-compatible ESs can reasonably be used as an additional conservation argument as their inclusion creates potential synergies or at least no conflicts with biodiversity conservation. Generally, regulating and cultural services are conservation-compatible, while provisioning services are likely incompatible due to material extraction necessary to make use of the ES (Schröter and Remme, 2015).

We aim to assess the impact of limited conservation budgets on cost-effective spatial network conservation strategies for ESs and biodiversity. The conservation site selection software Marxan offers an approach to integrate ESs and biodiversity targets as well as cost information in the context of systematic conservation planning. Marxan is based on an optimization algorithm (Ball et al., 2009) and follows three main principles to solve conservation problems: comprehensiveness (i.e. reaching multiple conservation targets), cost-effectiveness (i.e. cheaper solutions are preferred to costly solutions) and connectivity (i.e. a low edge-to-area ratio of a conservation area) (Wilson et al., 2010). Recent studies using Marxan have integrated ES and biodiversity targets to develop conservation networks (e.g. Chan et al., 2011; Izquierdo and Clark, 2012; Egoh et al., 2014; Schröter et al., 2014c). These studies have included different types of cost data, ranging from restoration costs (Egoh et al., 2014) to opportunity costs for alternative land uses (Chan et al., 2011; Schröter et al., 2014c) and accumulated threats to ESs (Izquierdo and Clark, 2012). To date, direct costs of land acquisition have not been applied to develop conservation areas with Marxan. In addition, most studies assume one single time step to develop a conservation network in their analysis In this study, we aim to also assess the impact of applying a budget over multiple time steps on the conservation network's cost-effectiveness. To address our aims we use the expansion of the current conservation network in Dutch province of Limburg as a case study.

## 5.2. Methodology

### *5.2.1 Study area*

Limburg province is located in the south-east of the Netherlands and covers approximately 2,200 km$^2$. Limburg is densely populated with over 500 inhabitants per km$^2$ and a total population of 1.1 million (Statistics Netherlands, 2013f). The province has a varied cultural landscape, which has been intensively managed for many centuries (Berendsen, 2005; Jongmans et al., 2013). Most natural ecosystems have been converted, and those that remain are highly fragmented (Jongman, 2002). Limburg harbours numerous species of national and even international importance (Statistics Netherlands et al., 2008) and provides many habitats that are rare in the Netherlands, such as calcareous grasslands (Willems, 2001). The provincial Nature Policy Plan has distinguished different protection zones (Provincie Limburg, 2013). The core nature areas (the so-called gold-green areas) are included in the National Ecological Network (Figure 5.1). These are priority areas from ecological, landscape and recreational perspectives (Provincie Limburg, 2013). They include both Natura 2000 areas and areas that are still actively used and managed by humans (cf. category IV Habitat/Species Management Areas of the IUCN's Global Protected Areas Programme (IUCN, 2014)). Silver-green areas are newly created nature areas and zones with agro-ecological management regimes. These were planned in the original National Ecological Network but have not been fully included yet due to budget cuts. Bronze-green areas are locally-relevant priority zones that were excluded from the National Ecological Network (Provincie Limburg, 2013). The silver-green and bronze-green areas form the basis to find new conservation areas in our scenarios.

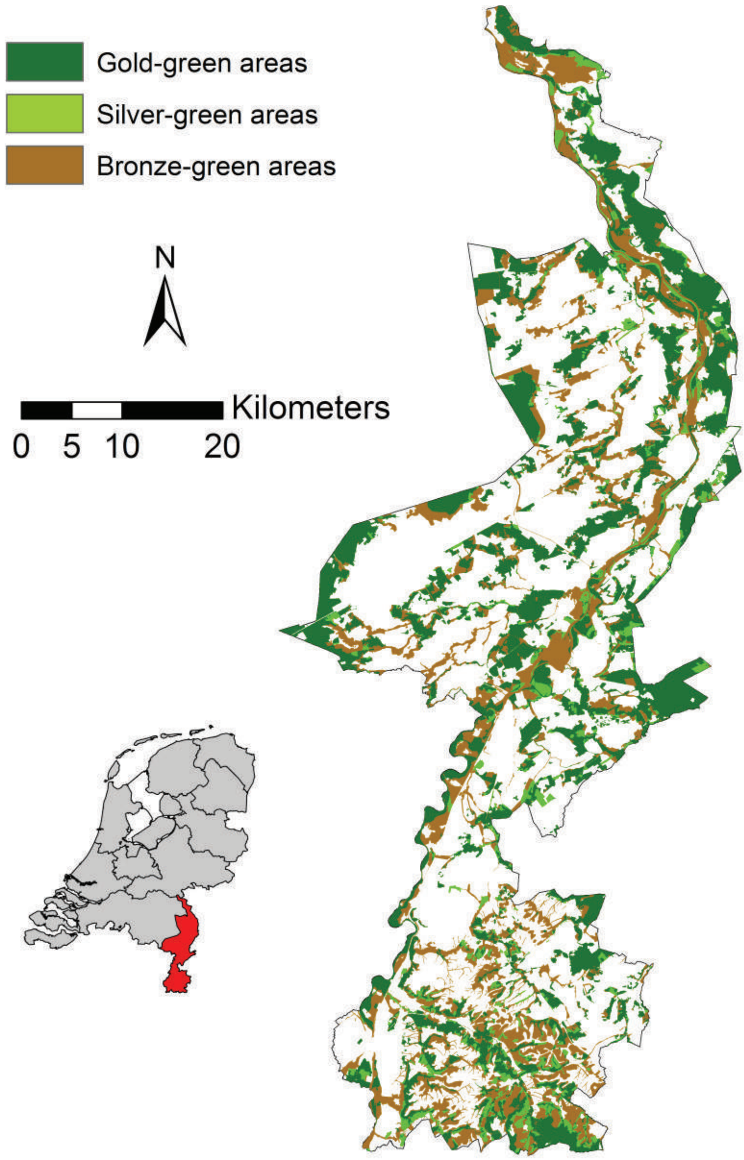


**Figure 5.1** Zonation of Limburg for a conservation network, with the gold-green (current conservation network), and silver-green and bronze-green areas (both possible network expansion areas). White represents areas that are not suitable for conservation. Derived from Provincie Limburg (2014).

### *5.2.2 Marxan*

The Marxan software (version 2.43) was used to assess the consequences of our scenarios for prioritizing network expansion areas. Marxan uses a heuristic optimization algorithm with the help of simulated annealing to develop spatially explicit solutions for conservation problems (Ball et al., 2009). The output provides cost-effective solutions based on an available budget, network development costs and multiple features that represent biodiversity aspects and ESs. Marxan combines three major principles of systematic conservation planning: comprehensiveness, cost-effectiveness and connectivity. Comprehensiveness implies that the software tries to simultaneously reach multiple conservation targets. Cost-effectiveness implies that the software minimizes costs while approaching the selected targets. Penalties are applied for not meeting conservation targets and breaching a given cost threshold (Game and Grantham, 2008). Connectivity refers to compactness of the conservation network. Marxan tries to minimize the edge-to-area ratio to develop a comprehensive and compact conservation network. The software was developed to minimize an objective function that contains the conservation costs and the boundary length of the conservation area. Different types of quantitative data for conservation features, such as presence data and metric data, can be combined in Marxan. Marxan provides two types of spatially explicit outputs, namely a best run and a selection frequency. The best run is the run that minimizes the objective function (of all runs, 100 in our case), and the selection frequency is the number of times that a planning unit was selected in the runs (ranging from 0 – never selected to 100 – always selected).

### *5.2.3 Data*

We analysed twenty-nine biodiversity aspects and five ESs, which were each included in Marxan as separate features (Appendix III, Table AIII.1). The biodiversity aspects included six features for which official policy goals from the provincial nature policy plans exist (Provincie Limburg, 2013) and twenty-three agro-ecological conservation focus species (Provincie Limburg, 2015a). The features with policy goals are farmland birds, meadow birds, foraging areas for geese, habitat for the European hamster *(Cricetus cricetus)*, hedgerows and traditional orchards. To represent the twenty-three focus species, we included presence data for seven nesting bird species at 250x250m resolution (Provincie Limburg, 2015b) and sixteen plant and animal species with presence data at 1km$^2$ resolution (Alterra, 2015). The five ESs, for which spatial models have been developed (Remme et al., 2014; Remme et al., 2015) are annual drinking water provision (m$^3$ per ha), annual carbon sequestration (ton C per ha), annual air quality regulation (€ per ha), annual nature tourism (€ per ha) and annual recreational cycling (cycling trips per ha). We included only ESs that are conservation compatible (i.e. can be reasonably used as an additional conservation argument and do not conflict with biodiversity; Chan et al., 2011). Detailed methods for the spatial models for drinking water provision, carbon sequestration and recreational cycling are described in Remme et al. (2014) and for air quality regulation and nature tourism in Remme et al. (2015). All the characteristics of the ESs, policy goals and focus species, and their targets are listed in Table AIII.1 of Appendix III.

To estimate conservation network expansion costs, we applied registered cadastral land prices as a proxy. Average monthly transaction prices of arable land, grassland and an aggregation of all other land were available for south and north Limburg for October 2013 to September 2014 (Boerderij, 2014). The annual average land price was applied to three different land-cover types: arable land, agricultural grasslands and a composite class including all non-agricultural and non-urban land use classes, based on the Dutch 25x25m land cover map LGN6 (Hazeu, 2009). Built-up areas were excluded from the analysis. The land-price data were used to estimate the costs to obtain new conservation areas. Other costs, such as maintenance costs were Ignored because we focused on the major costs of implementing the network. Limburg was divided into planning units of 100x100m grid cells, each containing information about the present conservation features and opportunity costs.

### *5.2.4 Scenarios, targets and budget simulations*

Two conservation scenarios were developed. The first scenario (Biodiversity) focuses on achieving biodiversity targets only as is the case in the current provincial conservation policy. Conservation targets were set for each of the twenty-nine biodiversity aspects. The six policy targets for habitat conservation were applied as proposed in Limburg's policy plan (Provincie Limburg, 2015a). For the twenty-three focus species the target was set to increase conservation by an additional 10% of each species' total occurrence area (Table S1). For the Corn crake *(Crex crex)* only four known habitat locations exist in Limburg. Three of them are currently protected. In this case the target was set at protecting all locations. The species targets were set arbitrarily as there are no specific species goals defined for Limburg. The 10% targets were chosen because they require substantial additional conservation efforts, while limited room exists to expand conservation in this densely populated province. The ESs were not included with specific targets in the

Biodiversity scenario, however, we recorded the amount of ESs that would be protected as co-benefits of biodiversity conservation in the selected areas.

The second scenario (Biodiversity and ESs) included both biodiversity and ES targets. The biodiversity targets were the same as in the Biodiversity scenario. For ESs, conserving an additional 10% of the total amount of ES provision in Limburg compared to the current situation was targeted. ESs targets are seldom set in policy plans so far (Luck et al., 2012), and do currently not exist for Limburg. Therefore, the ESs targets are arbitrarily selected.

According to Limburg’s nature conservation policy plan for the period 2013-2020 approximately €32 million is available to conserve biodiversity conservation in silver-green and bronze-green zones between 2013-2015 (Provincie Limburg, 2013). No budget information was available for the period 2016-2020. We assumed that this budget is approximately €10.5 million per year and that this full budget would be used to fund the expansion of the conservation network. For the period up to 2020 we varied the budget to test how this influenced network expansion. For both scenarios we started with a budget simulation of €32 million. Next, a budget simulation was tested in which it was assumed the €10.5 million per year was available up to 2020, equating to €84 million for the period 2013-2020. The budget simulation without a cost threshold helped to establish the maximally required budgets for achieving all conservation targets (€705 million for the Biodiversity scenario and €1447 million for the Biodiversity and ESs scenario). Finally, for both scenarios hypothetical budgets of €200 million, €400 million, €500 million, €600 million and €700 million respectively were simulated, and for the Biodiversity and ESs scenario budgets of €800 million, €1000 million and €1200 million were additionally simulated because implementing ES targets could be more expensive. Figure 5.2 provides a schematic overview of the scenario analysis. Selection frequency maps were produced to assess differences between the scenarios and between budget simulations.

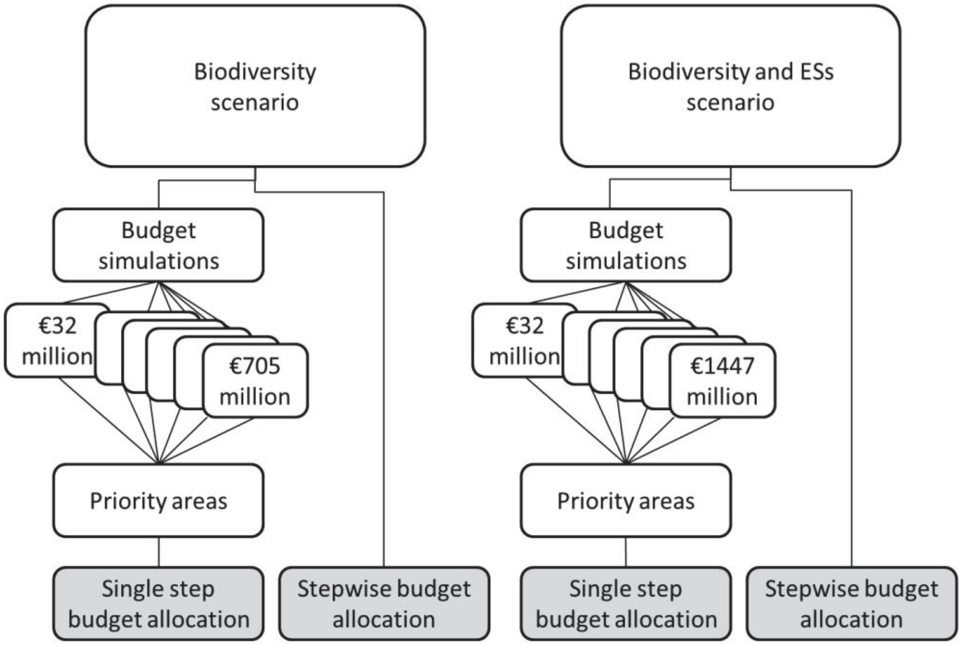


**Figure 5.2** Schematic overview of analysis for the two conservation scenarios.

### *5.2.5 General parameters*

Parameters were set according to developed best practice for this software (Ardron et al., 2008). Marxan's Species Penalty Factor was kept at 1.0 for all features. Marxan's Boundary Length Modifier was set to the most appropriate value of 5000 after testing its sensitivity. Each scenario simulation was repeated in 100 hundred times with a slightly altered Cost Threshold Penalty. Higher Cost Threshold Penalties were used for simulations with lower budgets and lower Cost Threshold Penalties for higher budgets. All simulations are summarized in Table AIII.2 of Appendix III.

Limburg was split into three area types. First, the gold-green areas were locked in, so that these areas were always included as prioritized sites in the Marxan output. Second, the silver-green and bronze-green areas were identified as areas to possibly expand the network. Finally, all areas outside the gold-green, silver-green and bronze-green areas were excluded from the analysis (i.e. these areas were not prioritized for network expansion. These areas include mainly built-up areas, such as towns, roads and industry, and economically important agricultural areas.

### *5.2.6 Target achievement-cost relationship*

For both scenarios we created a curve for average target achievement versus costs of the network expansion. To create these curves Marxan simulated different cost thresholds, ranging from €32 million to €705 million (i.e. maximum target achievement) for the Biodiversity scenario and from €32 to €1,447 million for the Biodiversity and ESs scenario. Average target achievement was calculated for ESs, biodiversity aspects and all features combined. Target achievement was measured per feature as the percentage of the additional conservation achieved in the best solution of a scenario run. To assess the added value of conserving ESs next to biodiversity, the average ES target achievement for the Biodiversity and ESs scenario was compared to the curve for the Biodiversity scenario (which included ESs as co-benefits only).

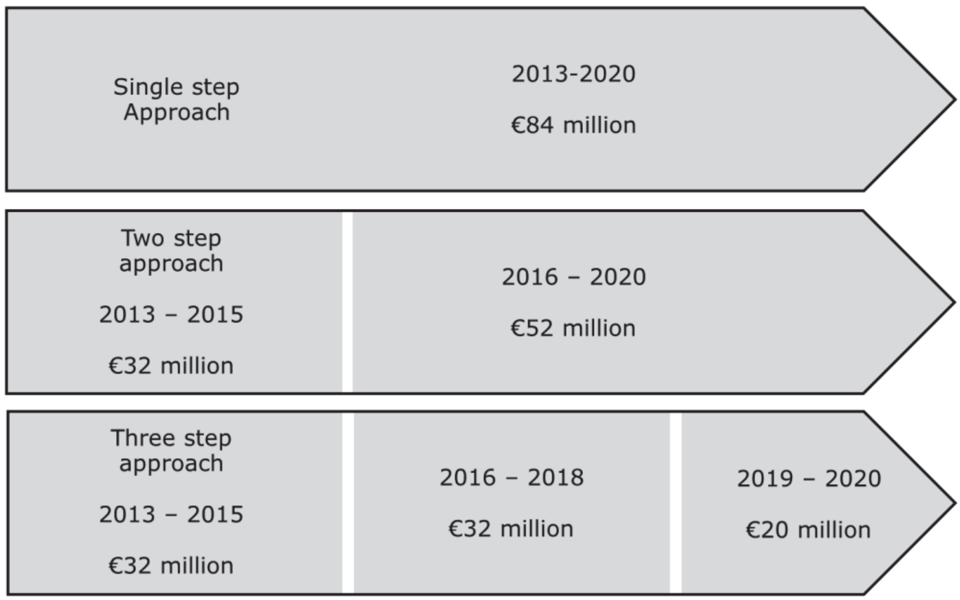


**Figure 5.3** Schematic representation of allocation of budgets over time according to a single step, two step and three step approach.

### *5.2.7 Step-wise approach for conservation network expansion*

In practice, the national and provincial conservation networks have been developed gradually over time, assigning new areas over the years. Marxan, however, assumes a single step approach for developing or expanding such a network. To assess the effects of this single step approach as opposed to a more gradual longer-term development, we developed a stepwise approach to allocate the available budget with Marxan (Figure 5.2 and Figure 5.3). We tested the stepwise approach for both scenarios and applied a constant annual budget for the period 2013-2020. First, we applied a single step approach, immediately allocating the total budget. Next, we applied a two-step approach, allocating €32 million in the first step (2013-2015) and allocating the remaining €52 million subsequently in a second step (2016-2020). The selected planning units from the best run in the first step were locked into the conservation network before running the second step, so that these areas were automatically selected as conserved areas in the second step. Finally, a three-step approach was applied, allocating €32 million in the first step (2013-2015), 32 € in the second step (2016-2018), and €20 million in the third step (2019-2020). Results from the three stepwise simulations were compared based on their average target achievements and the spatial distribution of selected planning units.

### *5.2.8 Optimizing conservation network expansion based on selection frequencies*

To assess which areas were prioritized for conservation by Marxan, we developed priority area maps. For both scenarios we created planning unit selection frequency maps for the different budget simulations. This resulted in six maps per scenario. From each map, we selected all planning units with a selection frequency higher than 50 to determine high priority areas (i.e. areas chosen in at least 50 of the 100 runs). For both scenarios we overlayed the six priority area maps to establish by how many budget simulations planning units were prioritized, between zero (i.e. not prioritized by any) and six (prioritized by all). The Spearman correlation coefficient was determined to estimate the spatial correlation between the priority maps of the two scenarios.

## 5.3. Results

### *5.3.1 Achievement-cost curves for different sizes of conservation budgets*

The expansion of the conservation network achieving all scenario targets would be more than twice as expensive for the Biodiversity and ESs scenario (€1447 million), compared to the Biodiversity scenario (€705 million) (Figure 5.4). This large difference in necessary budgets is due to the higher number of targets in the Biodiversity and ESs scenario. In both scenarios an average biodiversity target achievement of 43% was reached with a €32 million budget and 64% with a €84 million budget. In the Biodiversity scenario an average biodiversity target achievement of 98% was already reached with the €400 million budget and higher budgets mainly increased the connectedness of the network (Figure 5.5). In the Biodiversity and ESs scenario 98% of the biodiversity targets were met with the €500 million budget, but this budget only supported an average target achievement of 57% for ESs. For the €1000 million budget a 95% average target achievement for ESs was reached. The biodiversity target achievement differed little between the two scenarios. However, the scenarios strongly differed for the ES target achievements. For the €32 million and €84 million budget simulations the differences were relatively small, but larger in the Biodiversity and ESs scenario. For budgets between €200 and €700 million the Biodiversity and ESs scenario achieved between 7 to 17 percentage points higher ES targets than the Biodiversity scenario. ES target achievement was significantly higher for the Biodiversity and ESs scenario than for the Biodiversity scenario for budgets of €200 million ($P < 0.05$), €400 million ($P < 0.01$) and €700 million ($P < 0.01$). Noticeably, average target achievement for ESs showed a linear increase with increasing budgets, while the average target for biodiversity features increased asymptotically. This might be due to the fact that ES features such as carbon sequestration and recreational cycling are more widely spread than biodiversity features and planning units containing these features are increasingly prioritised with increasing budgets. Furthermore, it seems that large proportions of the targets for ESs are costly to achieve.

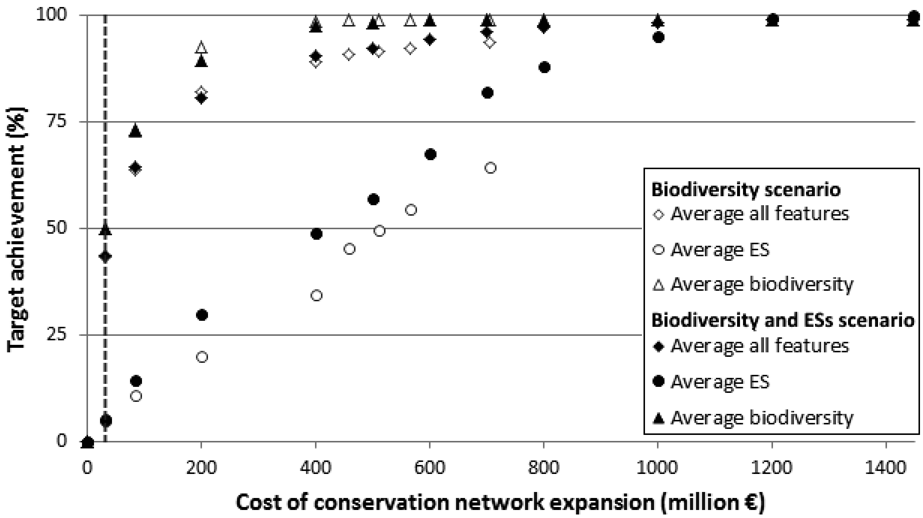


**Figure 5.4** Relation between costs of the conservation network expansion and average target achievement for the biodiversity scenario (dark-red markers) and ES and biodiversity scenario (blue markers).The dashed line indicates the conservation budget for 2013-2015 (€32 million).

Figure 5.5 shows the selection frequency maps for the two scenarios with budgets increasing from left to right. The frequency maps for the €84 million budget simulation show small clustered areas with high selection frequencies for both scenarios. In the €200 million budget simulation (middle column) the Biodiversity scenario shows a larger number of planning unit clusters with high selection frequencies, while the Biodiversity and ESs scenario shows a more scattered result with more isolated planning units with high selection frequencies. For the highest budget simulations both scenarios show large clusters of high selection frequencies that generally border the current conservation network. Selection frequencies decreased as distance from the current conservation network increased.

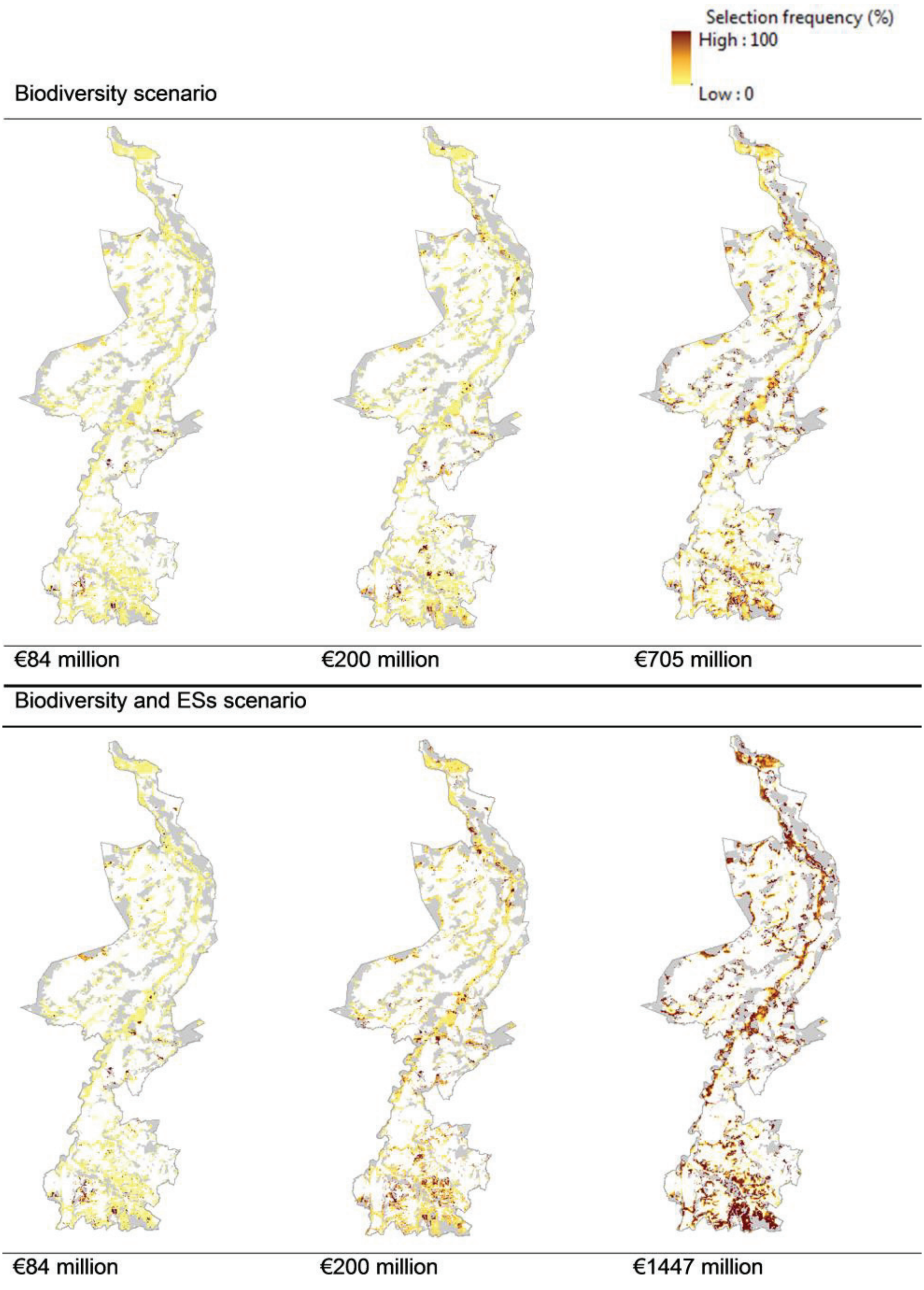


**Figure 5.5** Selection frequency of the planning units for different budget sizes for the Biodiversity scenario (above) and the Biodiversity and ESs scenario (below). Grey is the current network.

### *5.3.2 Single versus stepwise approach*

The single time step approach performed better than the two or three step approach in terms of average target achievement and network compactness (Table 5.1). Similar average targets were achieved in both scenarios, except for the ESs, for which the Biodiversity and ESs scenario performed slightly better. In both scenarios the single step approach achieved 2 percentage points higher average targets for all conservation features compared to the two step approach and 9% higher average targets for all conservation features compared to the three step approach. In the Biodiversity and ESs scenario the single step approach achieved the highest average targets for ESs, biodiversity aspects and all features combined. In the Biodiversity scenario, the two step approach achieved the highest average targets for ESs, but it achieved lower average targets than the single step approach for biodiversity aspects. The approaches with multiple time steps resulted in a more scattered network expansion (Figure 5.6). The single step approach generally selected multiple new conservation areas that are connected to the current conservation network. Although the two step approach generally resulted in a similar pattern, the network was spatially more scattered with more single isolated planning units. The three step approach resulted in the most scattered spatial distribution, with few clustered planning units.

**Table 5.1** Average target achievements and boundary lengths of the conservation network of the stepwise approaches for the two conservation scenarios.

| **Approach** | **Average target achievement (%)** | | | **Boundary length (km)** |
|---|---|---|---|---|
| | All features | ESs | Biodiversity aspects | |
| **Biodiversity scenario** | | | | |
| Single step | 64 | 11 | 73 | 3748 |
| Two-step | 62 | 13 | 71 | 3751 |
| Three-step | 55 | 10 | 63 | 3854 |
| **Biodiversity and ESs scenario** | | | | |
| Single step | 64 | 14 | 73 | 3719 |
| Two-step | 62 | 13 | 70 | 3742 |
| Three-step | 55 | 12 | 63 | 3851 |

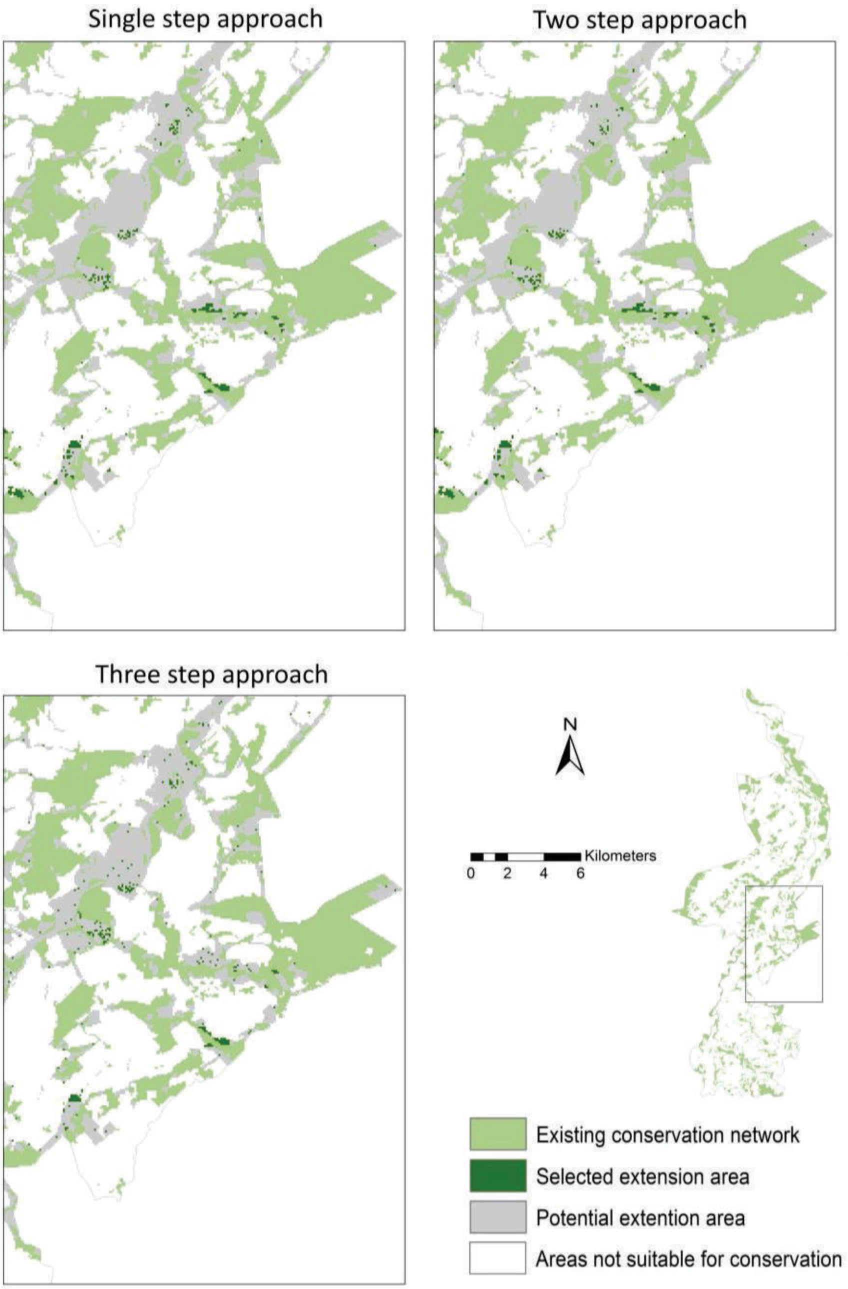


**Figure 5.6** Outcomes of the stepwise approach for budget allocation of the Biodiversity and ESs scenario, showing the selected conservation network extension areas (dark green) for a single step, two step and three step approach for a section of Limburg.

### *5.3.3 Priority areas for conservation*

Approximately 1200 planning units were prioritized by more than three budget simulations for both scenarios (Table 5.2). In the Biodiversity scenario nearly twice as many planning units were prioritized at least once, compared to the Biodiversity and ESs scenario. The two scenarios also showed overlap in area prioritization. 81 planning units were selected by all budget simulations of both scenarios. 792 planning units were selected by at least four budget simulations of both scenarios. The priority area maps for both scenarios (Figure 5.7) were weakly correlated with each other (Spearman's rho = 0.42). Some areas are always prioritized, as can be seen in the zoomed insets, but clear differences also exist between the scenarios. The Biodiversity and ESs scenario shows a more scattered spatial pattern than the Biodiversity scenario. This is in line with the selected area under different budgets (Figure 5.5). The Biodiversity scenario map contains many areas that were prioritized by one or two budget simulations.

**Table 5.2** Number of budget simulations prioritizing quantities of planning units per conservation scenario.

| Number of budget simulations prioritizing planning units | Number of selected planning units | |
|---|---|---|
| | **ES & biodiversity scenario** | **Biodiversity scenario** |
| **0** | 37,702 | 31,461 |
| **1** | 3,752 | 4,923 |
| **2** | 1,135 | 6,421 |
| **3** | 1,237 | 995 |
| **4** | 624 | 644 |
| **5** | 468 | 500 |
| **6** | 123 | 97 |

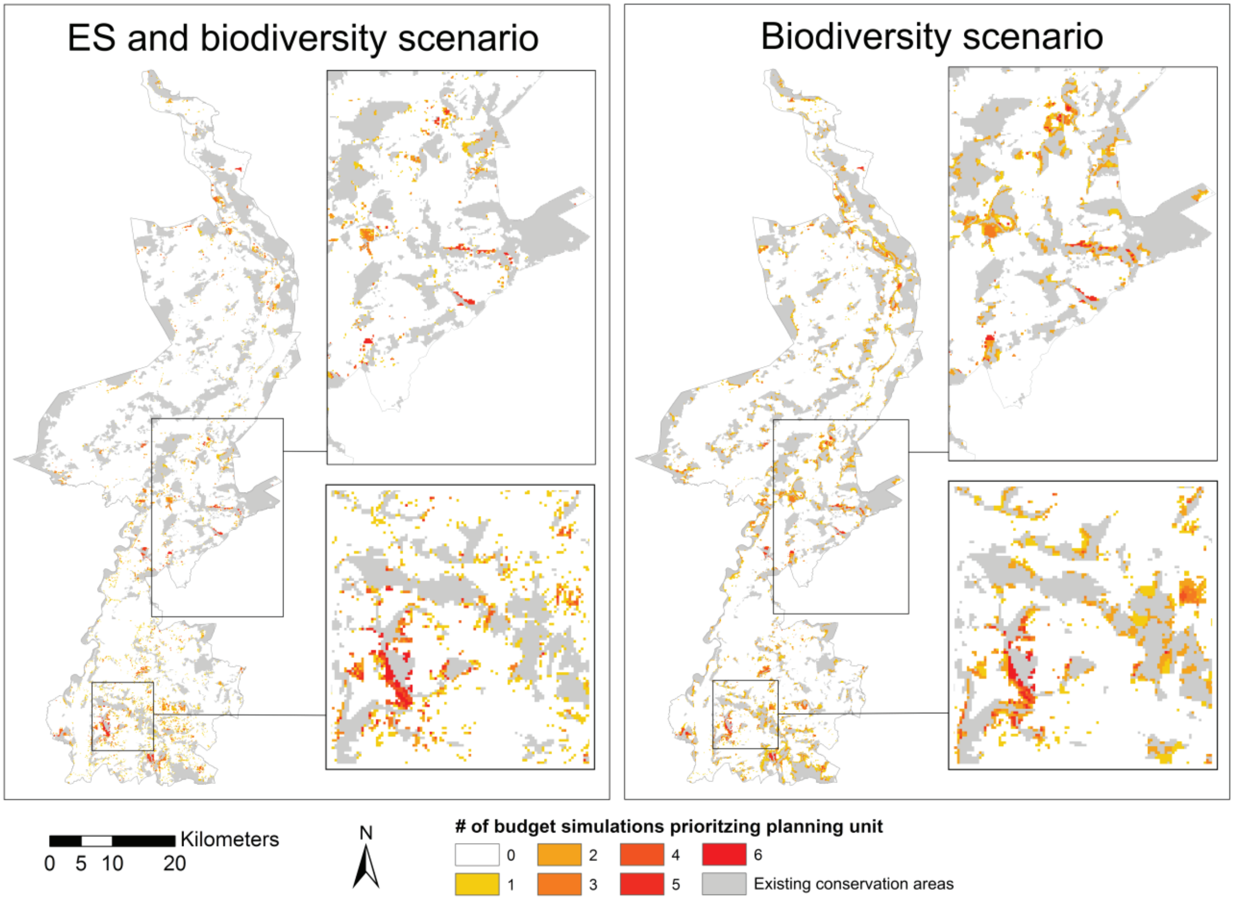


**Figure 5.7** Number of budget simulations for which a planning unit was selected to expand the network in more than 50% of the runs for both conservation scenarios. Grey areas indicate the current conservation areas.

## 5.4. Discussion

### *5.4.1 The added value of conserving ecosystem services*

Our results showed that including ESs into conservation planning for Limburg had only a small negative impact on achieving the overall biodiversity conservation targets. Planning did clearly positively influence achieving targets of conservation-compatible ESs, even for restricted conservation budgets. Considering both biodiversity and ESs in the conservation network expansion could therefore preserve more features of ecological importance than traditional conservation policies that were solely aimed at biodiversity. This is in line with results of other studies (e.g. Chan et al., 2011; Cimon-Morin et al., 2013). The area needed to conserve both biodiversity and ESs was larger than that for a dedicated biodiversity conservation network, as was also shown in earlier studies (Egoh et al., 2010; Chan et al., 2011; Schröter et al., 2014c). The larger network required more budget to substantially increase conservation of both biodiversity and ESs. The spatial distribution and abundance patterns clearly differ between biodiversity and ESs (Cimon-Morin et al., 2013). In Limburg, the ESs were less clustered than the biodiversity aspects and this resulted in a more widespread conservation network.

Conserving conservation-compatible ESs has consequences for provisioning services such as agricultural ESs and hunting, because of the necessary management changes and restrictions of use. If all areas prioritized by at least four budget simulations in the Biodiversity and ESs scenario (approximately 1200 ha, Figure 5.7) were included in the conservation network, this would result in an annual loss of approximately €0.4 million in terms of the ESs crop production, fodder production and hunting, as modelled by Remme et al. (2015). However, the network expansion would sustainably conserve €0.8 million annually in terms of conservation-compatible ESs (drinking water production, carbon sequestration, air quality regulation and nature tourism) (Remme et al., 2015). This example shows that including ESs in a conservation analysis can be used to analyse whether benefits exceed conservation costs, as has been found in other studies (e.g. Naidoo and Ricketts, 2006; Polasky et al., 2012). In addition, the conservation network and decreased human pressures therein could further improve the conditions to provide ESs. Areas with lower human pressure (e.g. protected areas) generally have a high capacity to deliver regulating and cultural services (Schneiders et al., 2012). The changes in ecosystem conditions could also provide opportunities for new biodiversity and ESs to develop in an area. This, however, requires further research to further integrate biodiversity and ESs and on how to provide incentives to increase conservation budgets (Cimon-Morin et al., 2013). Besides budget constraints, poor availability of spatial data on ESs was one of the most important limiting factors to include ESs in conservation network planning (Knight et al., 2006; Egoh et al., 2007). Such data constraints are, however, now rapidly becoming less relevant with the stringent calls for mapping ecosystem and their services, such as those in the European Union Biodiversity Strategy (European Commission, 2011), and the rapid increase in the number of ES mapping studies and available methods (Martínez-Harms and Balvanera, 2012).

### *5.4.2 Limitations of the conservation scenarios*

In this study relatively few target species were selected to be conserved in primarily agro-ecological areas (cf. Provincie Limburg, 2015a). We used presence data at relatively course resolutions compared to the heterogeneity of local landscapes. Our study does not aim to comprehensively assess all species diversity in Limburg. How changes in the conservation network affect all biodiversity therefore remains unclear. By contrast, Wamelink et al. (2013) assessed the effects of budget cuts on Limburg's ecological network based on 249 species from the EU's Bird- and Habitat directives. Although they did not assess which areas should be prioritized for network expansion, they concluded that the budget cuts negatively affected all species throughout the province.

We analysed the conservation network expansion based on the full protection of areas. This is a simplification of current conservation policies. In Limburg's silver-green and bronze-green areas partial use strategies are also applied. These strategies combine conservation measures with extractive land-uses (Provincie Limburg, 2013). Jongeneel et al. (2012), for example, showed that agriculture-based nature management is cheaper than full protection. Limburg's policy makers also indicate that conserving an area based on partial use is five to six times cheaper than land price-based full protection. Although an alternative version of Marxan, Marxan with Zones, allows to include partial use zones, we did not use this version because adequate spatial information on how effective partial use strategies are for conservation (Makino et al., 2013) and opportunity costs in these zones were lacking. Furthermore, including partial use when developing a conservation network likely vastly increases the area needed for biodiversity and ES conservation (Schröter et al., 2014c).

We aggregated land-cost data for north and south Limburg due to a lack of site specific data and therefore could not fully consider the local differences in land prices. Also, the (un)willingness of local land owners, whose areas are prioritized for conservation, to sell their land could change land prices and could exclude areas from a network that would be optimal to connect other areas. The priority areas that resulted from the scenario analysis, are, however, proper starting points for provincial planners to expand the network. Negotiations with land owners and local stakeholders will be necessary to ensure broader societal acceptance of the conservation network expansion.

This study included five conservation-compatible ESs (Appendix III, Table AIII.1) that are frequently included in biodiversity and ES assessments (Chan et al., 2011; Schröter et al., 2014c; Adame et al., 2015). Whether increasing the amount and diversity of ESs when developing conservation networks will subsequently improve the combined conservation of biodiversity and ESs remains unclear (Cimon-Morin et al., 2013). In addition, which ESs are fully compatible with biodiversity is uncertain (Gos and Lavorel, 2012; Geijzendorffer and Roche, 2013). Some of the selected ESs could potentially negatively impact biodiversity. For example, if recreation and tourism overcrowd a conservation area, species could be disturbed, damaged or avoid the area. The integration of ESs and biodiversity is improving (Mace et al., 2012; Reyers et al., 2012; Harrison et al., 2014), but scientific understanding on optimising conservation management needs to be further developed.

Target setting for ESs remains a major challenge for further operationalizing the ES concept for conservation purposes (Luck et al., 2012). While for our study some targets for biodiversity features were available, we had to assume targets for the ESs due to lack of policy-based targets for our study area. A change in targets, however, has been shown to have an effect on the size of the area selected for conservation (Egoh et al., 2010) and likely also on the spatial configuration of the conservation network. In future studies, targets could potentially be derived from an analysis of the demand for ESs in a particular area (Wolff et al., 2015), or through deliberative discourses with stakeholders and decision-makers on how many ESs need to be sustained in order to fulfil human needs.

### *5.4.3 Conservation strategy of the Netherlands and Limburg*

The call to align the Dutch National Ecological Network with a greener economy (Buijs et al., 2014) could provide an important incentive to include ESs in the further development of conservation networks, as ESs form a valuable part of the local economy (Remme et al., 2015). Given the Dutch decentralization of conservation policies towards provinces, Limburg could include ES targets in their strategy to select conservation areas, provided that the current core areas are maintained. The provincial conservation plan clearly aims to also conserve areas of importance for recreation and tourism (Provincie Limburg, 2013). This aim is incorporated in our study in the form of two cultural ESs. Including additional ESs would ensure that other important ESs are also provided to society, also ensuring landscape multifunctionality (Mastrangelo et al., 2014). Limburg plans to obtain an additional 3500ha for conservation purposes by 2020 (Provincie Limburg, 2013). Based on the Biodiversity and ESs scenario, we therefore suggest to focus on those areas that were prioritized by at least two budget scenarios in our analysis (i.e. 3587 ha, Table 5.2 and Figure 5.7). Given the obvious budget constraints, areas selected by five or six budget simulations in either of the scenarios, should be selected first to ensure that high priority areas are conserved.

## 5.5. Conclusion

Our analysis has shown that currently Limburg's conservation budget does not suffice to substantially increase the conservation network, while achieving specific conservation targets. This limited budget calls for alternative strategies, such as the inclusion of ESs, which cause opportunity costs for society but also provides it with clear benefits. The inclusion of ESs in the expansion of Limburg's conservation network only moderately changes prioritized areas compared to only conserving biodiversity. In both cases approximately 1200ha are prioritized for conservation independent of the size of the restricted conservation budget. These areas should be considered first to expand the conservation network. Conserving both biodiversity and ESs results in a more scattered spatial distribution of the conservation network than when only biodiversity is conserved. A higher budget is needed to conserve both ESs and biodiversity. Keeping the current provincial conservation budget constant until 2020 approximately achieves 64% of the biodiversity conservation targets but only between 11-14% of the ES targets. For biodiversity, small increases in the current conservation budget would strongly increase target achievement. To effectively achieve more ES targets, however, a substantially higher budget is needed than the one that is currently planned. We have also shown that the conservation network expansion in a single time-step is more efficient in terms of compactness and cost-effectiveness than implementing the expansion in multiple time-steps. Therefore, when planning cost-effective conservation networks, the budget should be available before development and implementation starts. Although including ESs in the expansion of the conservation network is more costly than conserving only biodiversity, this increases the efficiency of protecting conservation-compatible ESs and ensures their continued future provision. Finally, including ESs in expanding Limburg's conservation network does not negatively affect biodiversity and likely enhances the maintenance of conservation-compatible ESs. Our study provides an example of how ESs could be combined with biodiversity aspects in planning conservation networks.

## Acknowledgements

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# Chapter 6 – Synthesis, discussion and conclusions

Partly based on:

Schröter, M., Remme R.P., Sumarga, E., Barton, D.N., Hein, L., 2015. Lessons learned for spatial modelling of ecosystem services in support of ecosystem accounting. *Ecosystem Services* 13, 64-69.

## 6.1 Scoping spatial ecosystem services and biodiversity modelling approaches for ecosystem accounting, conservation and management

Ecosystem service (ES) research is rapidly advancing, for instance on modelling (Crossman et al., 2013b), the development of ecosystem accounting (Hein et al., 2015) and understanding linkages between biodiversity and ESs (Harrison et al., 2014). This thesis contributes to these advancing scientific fields by analysing ESs and biodiversity in the context of ecosystem accounting, conservation and management. The main aim of this thesis is to empirically assess how spatial models for ES flows and biodiversity can be operationalized in this context. In this thesis, I analyse biophysical ES flows and apply monetary valuation techniques for ecosystem accounting, test spatial indicators for biodiversity accounting and use accounting information on ESs and biodiversity to develop plausible conservation scenarios. These issues are addressed through four research questions (RQs):

RQ5) How can biophysical ES flows be spatially modelled for ecosystem accounting?

RQ6) How can ESs be value in monetary terms for ecosystem accounting?

RQ7) Which species diversity indicators can be applied to develop a comprehensive biodiversity accounting framework?

RQ8) What are the effects of including ESs in planning for an expansion of a biodiversity conservation network?

In Sections 6.2 to 6.5 I will answer these research questions and discuss which further improvements are needed in the domains of spatial modelling, monetary valuation and biodiversity indicators to operationalise ecosystem accounting, and integrate biodiversity and ESs in support of ecosystem management and conservation. Subsequently, I will synthesize and discuss the main findings of my research and propose several specific ways forward for ecosystem accounting, ES and biodiversity conservation and management.

## 6.2 Biophysically modelling ecosystem services for ecosystem accounting

### *6.2.1 How to spatially model biophysical ecosystem service flows for ecosystem accounting*

Over the past decade the amount of studies dealing with spatial ES modelling has grown exponentially (Maes et al., 2012b; Martínez-Harms and Balvanera, 2012; Crossman et al., 2013b; Figure 6.1) and ES modelling research is also rapidly increasing in the context of ecosystem accounting (e.g. Schröter et al., 2014b; Sumarga and Hein, 2014; Duku et al., 2015; Schröter et al., 2015).

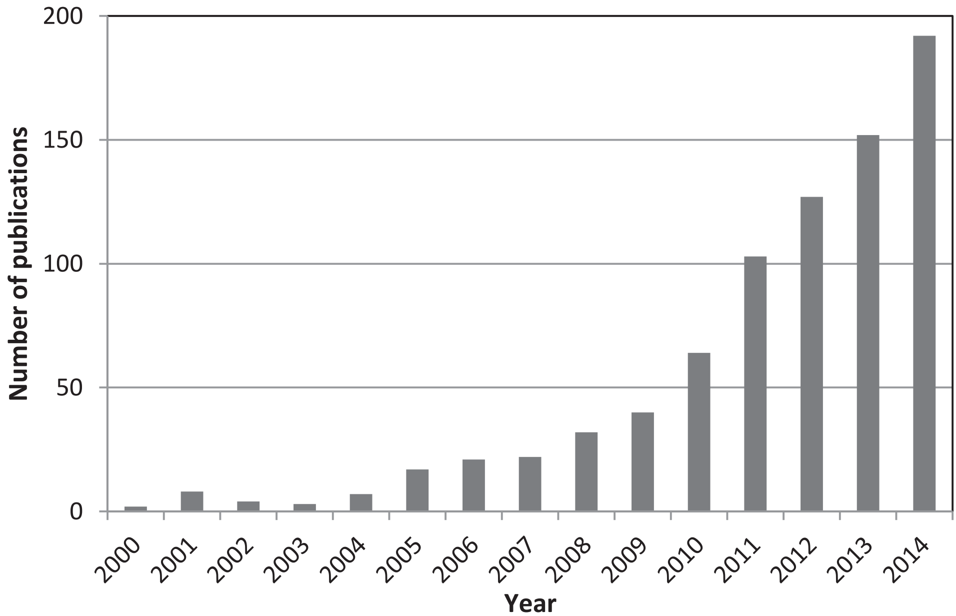


**Figure 6.1** Number of publications on spatial modelling of ecosystem services, based on a Scopus search with terms *"ecosystem service*"* and *spatial* model** in the topic. Search date: 06 August 2015.

In Chapter 2, I demonstrate that spatial modelling of biophysical ES flows in the context of accounting is feasible. This analysis included seven ES flow models for biophysical ecosystem accounting in Limburg. Each ES required a unique modelling approach, depending on specific environmental processes, ecosystem types and socio-economic conditions. The approaches included aggregated statistics look-up table (LUT) approaches, a multiple layer LUT approach and approaches based on causal relationships. The models differed in accuracy, defined as the degree of agreement between spatial modelling results and the modelled object of phenomenon (Harvey, 2008). Results from an analysis of the ES models for Limburg, combined with models for Telemark (Norway) and Central Kalimantan (Indonesia) in the context of the Ecospace project, show that the models based on causal relationships were generally more accurate than the models based on LUT approaches (Schröter et al., 2015). Spatial ES flow models can provide information for ecosystem accounts at multiple levels of spatial aggregation, ranging from local landscapes to large administrative units such as provinces or ultimately nations.

Modelling ESs for ecosystem accounting involves a trade-off between modelling feasibility and accuracy. Modelling feasibility is defined as the inverse of information costs attached to ecosystem accounting (Schröter et al., 2015). The information costs, and therefore also modelling feasibility, are influenced by several constraints in study designs. The first most important constraint is that feasibility is a function of the study area's extent. Larger areas are more likely to have a larger diversity of ecosystems (Turner et al., 1989), decreasing the likelihood that representative data for all ecosystems is available. A second important constraint is the variation of heterogeneity within ecosystems. Large, monotonous ecosystems, such as deserts, are more feasible to accurately model than small fragmented ecosystems (Schröter et al., 2015), such as most ecosystems in Limburg. A third constraint that affects modelling feasibility, is the available budget and time for data collection and model development. This constraint strongly affects choices for applied methods and potential data collection (Schröter et al., 2015). This constraint is one of the main reasons that currently ES models for ecosystem accounting have excluded interactions between ESs, because this requires increased process knowledge, time and data. These constraints affect modelling choices when balancing accuracy and modelling feasibility.

The level of accuracy and feasibility required for an end purpose determines how complex the applied models need to be (Tallis and Polasky, 2009). ES data collected specifically for ecosystem accounting or assessments is still limited, although the amount of data is increasing. To model biophysical ES flows for accounting, a 'satisficing' approach could be chosen, rather than going for the optimal approach. A satisficing approach permits "satisfaction at some specified level of all its needs" (Simon, 1956, p. 156), balancing accuracy and modelling feasibility. Such an approach compensates for the limited data availability in the current phase ecosystem accounting development. A relevant ES modelling approach needs to be coupled to the end purpose of the accounting exercise and the characteristics of a study area (e.g. extent, ecosystem heterogeneity, accessibility) (Schröter et al., 2015). For a large homogeneous area (e.g. a desert), applying complex modelling methods with high resolution spatial data is not needed, whereas in a very heterogeneous area more complex methods are likely required to accurately cover spatial ES distribution. A satisficing approach to ES modelling for ecosystem accounting allows to prioritise data collection efforts for those ESs that are most heterogeneous and most relevant for an area. If improved ES datasets become available over time, modelling methods for ecosystem accounting can also be gradually improved in terms of complexity and spatial resolution to more accurately account for changes.

### *6.2.2 Integrating spatial ecosystem service models for ecosystem accounting*

Although current knowledge and data on ES flows suffices to model many ESs for ecosystem accounting, still more efforts are needed to include regulating and especially cultural services that are rarely modelled (Chan et al., 2012; Tallis et al., 2012). Recent ecosystem accounting studies have included an increasing number of regulating ESs such as air quality regulation (this thesis), carbon storage and sequestration (this thesis; Schröter et al., 2014b; Sumarga and Hein, 2014), snowslide prevention (Schröter et al., 2014b), water purification and soil erosion control (Duku et al., 2015). The latter ESs are an important recent inclusion, as hydrological services have often been overlooked. In terms of cultural ESs, usually tourism and recreation are modelled (this thesis; Schröter et al., 2014b; Sumarga and Hein, 2014), but also habitat of flagship species (Sumarga and Hein, 2014) and the existence of wilderness areas (Schröter et al., 2014b). Cultural services related to inspiration, spirituality and education have not been modelled in an ecosystem accounting context to date. As ecosystem accounting is gaining momentum, the opportunities for applying spatial ecosystem accounting models are becoming clear, but also the gaps in current scientific knowledge.

A supply and use account for ES flows is a core aspect of ecosystem accounting, but this account becomes more informative when combined with the ecosystem condition, capacity, demand and biodiversity accounts (cf. Hein et al., 2015). For instance, the information potential of ES flows substantially increases if this information can be compared with the capacity of ecosystems to provide services. Spatially assessing capacity and flow provides information on under- or overuse of a service and can support monitoring of an ecosystem's sustainable use (Schröter et al., 2014b). Research on ecosystem capacity to provide ESs is increasing (e.g. Villamagna et al., 2013; Schröter et al., 2014b), as well as capacity mapping methods (Bagstad et al., 2014; Schröter et al., 2014b; Duku et al., 2015). However, ecosystem capacity is yet to be included in a full set of ecosystem accounts. An ecosystem condition account monitors ecosystem processes and components that influence the ecosystems state, functioning and extent (Hein et al., 2015). A spatially explicit ecosystem condition account would complement spatial ES flow and capacity accounting, by giving insight into the state of different ecosystems and the underlying processes needed to provide ESs. To develop condition accounts an assessment of the relevant condition indicators related to ESs and biodiversity is still needed. Spatial assessments of demand are increasing in ES research (e.g. Burkhard et al., 2012; Kroll et al., 2012; Nedkov and Burkhard, 2012; Burkhard et al., 2014), but demand is yet to be included in empirical ecosystem accounting studies. To include demand in ecosystem accounting the users of ESs and their location need to be assessed. Such an account would require further conceptualisation of aspects such as import and export of ESs across the boundary of the study area and between countries, and clearly delineating the included users. Sufficient scientific knowledge currently exists to develop models for many of the different ecosystem accounts. A next step in ecosystem accounting research is to test an integrated ecosystem accounting system with models for conditions, capacity, flow and demand.

## 6.3 Monetary valuation for ecosystem accounting

### *6.3.1 How to value ecosystem service flows for ecosystem accounting in monetary terms*

Similar to biophysical ES modelling studies, spatially explicit ES valuation studies have also rapidly increased over the past decade (Schägner et al., 2013). ES value maps for ecosystem accounting are the result of combining spatially explicit biophysical ES models and monetary information. In Chapter 3, I show that monetary valuation of different ESs is feasible with exchange value methods in line with the System for National Accounts (SNA; UN et al., 2009) and taking into account spatial heterogeneity. Valuation methods included in the System of Environmental-Economic Accounting - Experimental Ecosystem Accounting (SEEA-EEA), such as the resource rent method, avoided damage cost method and the replacement cost method, can be used to value many provisioning, regulating and cultural ESs (UN et al., 2014a; Obst et al., 2015). The applicability of a range of these monetary valuation methods for ecosystem accounting has also been shown by Sumarga et al. (2015). The authors spatially model and value seven ESs in an ecosystem accounting context for Central Kalimantan, Indonesia, using similar methods as I have applied for Limburg and conclude that for their case study the methods included in SEEA-EEA are appropriate.

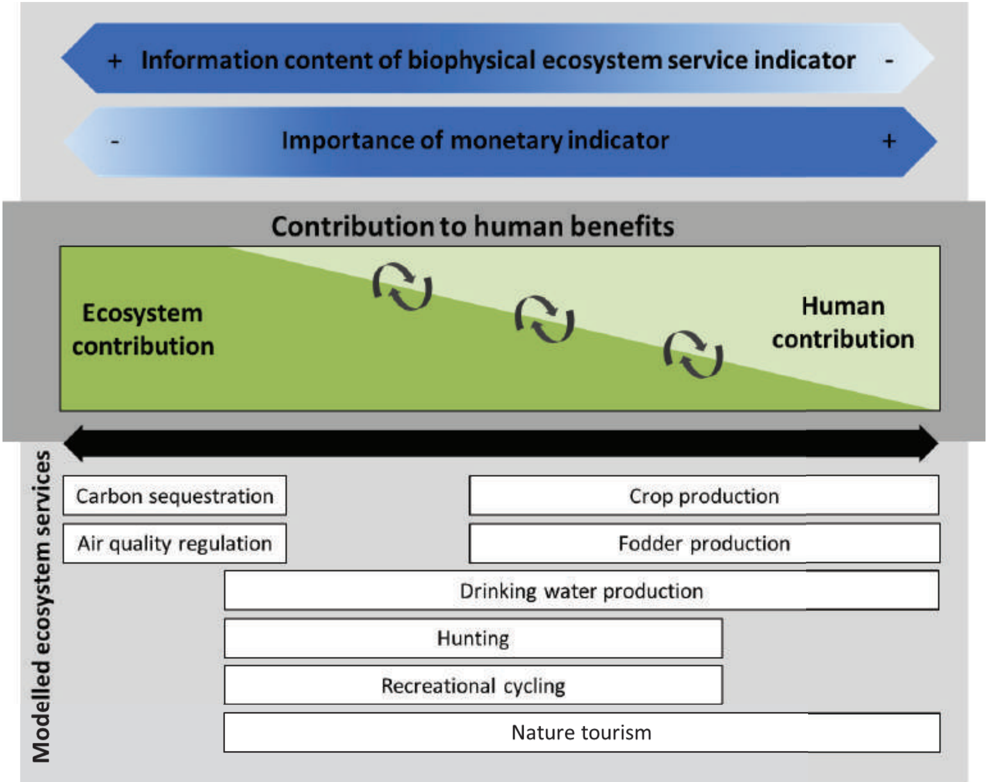


**Figure 6.2** The green bar illustrates the relation between ecosystem and human contributions to benefits. The ecosystem contribution decreases from left to right, while the human contribution increases. The circular arrows indicate entanglement of both contributions. The top arrows indicate the information content of biophysical indicators and the importance of a monetary indicator in addition to a biophysical indicators along the contribution gradient. Darker shading indicates higher importance. The bottom section indicates where the modelled ecosystem services are placed along the contribution gradient for Limburg. Adapted from Van Reeth et al. (2014).

Accounting for monetary values of ESs is important to disentangle ecosystem and human contributions, as I concluded in Chapter 3. In Figure 6.2, I illustrate the relation between the two types of contributions and the information value of a monetary indicator, besides biophysical indicators. The importance of a monetary indicator increases when a benefit is derived from a combination of ecosystem and human contributions, as opposed to a benefit derived fully from an ecosystem contribution. The primary reason for this increased importance is that the different contributions can be disentangled with monetary valuation methods, while this is often more difficult with biophysical indicators (as presented in Chapters 2 and 3). In Limburg, provisioning and cultural ESs were most entangled with human contributions (Figure 6.2).

A valuation method that is particularly useful for disentangling ecosystem and human contribution is the resource rent method, as it distinguishes costs related to different human contributions (labour, fixed capital, intermediate production) and costs related to the ecosystem (residual costs). The resource rent method is applicable to many provisioning and some cultural ESs (i.e. ESs that fall within the production boundary of SNA, in particular tourism and recreation) and is feasible because cost data on human and produced capital inputs are often collected for many of the economic activities involved. For example, due to the extensive monetary data on crop production in the Netherlands, I was able to more accurately disentangle the ecosystem contribution (€ per ha, Chapter 3), than with a biophysical indicator that reflected the combined product of ecosystem and human contributions (tons produce per ha, Chapter 2). The resource rent method allowed for disentangling ecosystem and human contributions based on costs. Biophysically disentangling ecosystem and human contributions would have required more data and a more complex modelling methodology. Note that the resource rent is not applicable in cases where the rent is zero or negative (Obst et al., 2015). Zero or negative rents are likely to result from existing market structures, such as subsidies, related to an ES. In cases where market structures do not allow the incorporation of a reasonable exchange value for an ES into the observed market price, the resource rent should not be used.

ESs that do not usually require a human contribution (e.g. carbon sequestration and air quality regulation, Figure 6.2), do not require the disentangling function that a monetary indicator provides. A monetary indicator is of less added value in this respect, but still provides relevant information to assess changes in ES provision over time (Radermacher and Steurer, 2015). Biophysical and monetary indicators cannot easily be compared in terms of importance, but an assessment should be made of the type of information both types of indicators reflect. In some cases, biophysical and monetary values reflect different aspects of an ES. For example for Limburg, the biophysical hunting indicator reflects the provisioning aspect of the ES (consumable meat, Chapter 2), while the monetary indicator reflects the cultural aspect (cost of hunting rights, Chapter 3).

The use of a monetary indicator provides the possibility to measure and sum ESs using a commensurable unit (Daily et al., 2009) and is therefore able to add information to a biophysical accounting approach. In addition to the resource rent method, other monetary valuation methods have proven to be applicable in an ecosystem accounting context. Avoided damage cost and replacement cost methods can be applied for various regulating services, such as carbon sequestration and air quality regulation (Sumarga et al., 2015; Chapter 3). Such methods are crucial for valuing many regulating services as market data does not exist. Although many ESs can be included in ecosystem accounts, with the current set of SNA-aligned valuation methodologies the exclusion of consumer surplus from monetary accounting limits the inclusion of all ESs (Bartelmus, 2013). Many ESs that mainly generate a consumer surplus, such as ESs related to education or inspiration, can currently not be included in ecosystem accounts. Such ESs, related to non-use values, are inherently challenging to value in monetary terms, but also in biophysical quantities (Radermacher and Steurer, 2015). Experimental methods that derive exchange values from consumer surplus, such as the simulated exchange value approach (Campos and Caparrós, 2006; Oviedo et al., 2010), need to be further tested to expand the ESs that can be included in ecosystem accounts. In addition, further research on valuation methods for hydrological services, such as water purification or flood prevention, is needed (Sumarga et al., 2015). To include many regulating and cultural ESs a flexible approach is necessary. A flexible approach will allow for the use of a wider range of valuation methods, including, for example, avoided damage costs, the travel cost method and the simulated exchange value approach. Further assessment is needed to determine whether all ESs can be valued consistently with the rigorous SNA guidelines.

### *6.3.2 Combining monetary and biophysical information*

Several challenges and weaknesses in monetary valuation for ecosystem accounting clearly still need to be overcome. In addition to lacking suitable valuation methods for some ESs (mainly cultural services) (Chan et al., 2012; Bartelmus, 2015), underlying weaknesses of existing valuation methods remain unresolved (Chee, 2004) and applicable valuation methods for ecosystem accounting are still being debated (Bartelmus, 2015; Obst et al., 2015). Nevertheless, a major strength of monetary valuation remains that an indicator is provided to compare different ESs through a commensurable unit. To date no biophysical indicator or other indicators allow to compare between many different provisioning, regulating and cultural services. Therefore, monetary ecosystem accounting complements biophysical ecosystem accounts. Vice versa, a wide range of biophysical indicators are available to cover more ESs than can currently be done with monetary values. As shown in Figure 6.2, monetary accounting also provides an opportunity to disentangle ecosystem and human contributions. This is often not possible with biophysical indicators. In addition, monetary values are easily understandable for decision makers. However, caution is needed as monetary values in ecosystem accounting do not measure welfare and can substantially differ from welfare analyses. This is clearly illustrated in Section 3.4.2 for the ES air quality regulation, were a welfare approach results in a monetary value that is over five times higher than when an ecosystem accounting approach is applied.

To ensure that ecosystem accounting is as comprehensive as possible accounting in both biophysical and monetary terms is essential. Biophysical data is necessary for monetary valuation, but should not be seen as merely an intermediate step in ecosystem accounting. Comparing biophysical quantities over time gives an accurate indication of changes in ecosystem condition, flows and use, and possibly a more objective insight into the sustainability of the ecosystem-human relationship than monetary values. Combining the strengths of both types of indicators is critical for developing comprehensive ecosystem accounts.

## 6.4 Integrating biodiversity into ecosystem accounting

### *6.4.1 Species diversity indicators for biodiversity accounting*

The role of biodiversity in an ecosystem accounting system is currently being established in the SEEA-EEA (UN et al., 2015). Biodiversity underpins ES provision, but is also an ES in itself (Mace et al., 2012). To incorporate the latter aspect in ecosystem accounting, a biodiversity account is included that measures and monitors different aspects of species diversity (UN et al., 2014a). In Chapter 4, I addressed different types of species diversity indicators that could be included in a biodiversity account, as part of an ecosystem accounting system. Indicators for species diversity aspects, such as species richness of multiple species groups, important areas or habitats for rare species and species abundance, all give different insights into the spatial heterogeneity of biodiversity. The information content differs between indicators. Some indicators are easy to understand for the general public, but are limited in their ecological importance (e.g. richness of a single species group), whereas other indicators are ecologically more informative but difficult to understand for decision makers and the general public (e.g. the Shannon-Wiener index). Variations in spatial patterns are large between different indicators. Therefore, to comprehensively account for biodiversity from a species diversity perspective, multiple types of species diversity indicators should be combined. This finding is also supported by research on biodiversity indicators (e.g. Purvis and Hector, 2000; Costelloe et al., 2015). To measure and monitor species diversity a biodiversity account should ideally include a composite indicator for richness of multiple species groups, an indicator which incorporates species abundance, especially of threatened species and an indicator for important habitats and areas for species diversity. As most countries have a poorer biodiversity data coverage than the Netherlands, in most cases the indicators that I tested in Chapter 4, cannot likely (all) be included in a biodiversity account in data-scarce countries. Biodiversity accounting needs to be tested in more countries to better understand which biodiversity indicators are applicable in different contexts. A first phase of biodiversity accounting could be to focus on indicators with relatively low information needs such as species richness and the availability of important habitats for endangered species. If internationally ecosystem accounting systems prove to be successful and available budgets increase, addressing ecologically more complex indicators, such as species abundance (at least for endangered species), would substantially enrich biodiversity accounting.

To efficiently develop an indicator set for biodiversity accounting existing indicator lists, such as that of the Convention on Biological Diversity (CBD; UNEP, 2006) or that of the European Union (Streamlining European Biodiversity Indicators, EEA, 2012), can be used as guides. Walpole et al. (2009) assess the completeness of the CBD's indicator list and conclude that mainly the indicators related to biodiversity components are well developed. Other indicators related to the categories 'ecosystem integrity, goods and services', 'status of knowledge, innovations, and practices', 'status of access and benefits sharing' and 'status of resource transfers' remain underdeveloped or even undeveloped. For the assessment of individual indicators, see Walpole et al. (2009). Even with their current shortcomings, internationally recognized indicator lists provide comprehensive general indicator types for different aspects of biodiversity, within which specific indicators could be chosen for biodiversity accounting. In this thesis, I focussed mainly on indicators developed with species data, but indicators that are more indirectly linked to species, could also be used, especially in regions that are less data-rich than Limburg.

A relevant addition could be mean species abundance (MSA), which calculates biodiversity as the remaining mean abundance of original species relative to their abundance in an undisturbed ecosystem (Alkemade et al., 2009). MSA is calculated using pressures, such as land use change, fragmentation and climate change, and does not require data on individual species. Through its modelling approach, with a limited amount of data that needs to be accounted for, the indicator is easy to implement and monitor. Especially in data scarce areas such an indicator could prove to be a valuable indicator in a biodiversity account. Alkemade et al. (2009) do stress that MSA should be used in combination with complementary indicators, as MSA does not completely cover the complex biodiversity concept. A species related composite indicator that has global coverage and a time series for trend analysis is the Living Planet Index (LPI), that measures the average trends in populations of vertebrate species from around the world (Loh et al., 2005). The LPI has collected trend data for thousands of vertebrates, providing a useful source of information for national biodiversity accounts. An essential initiative that can provide (remote sensing) data for regional biodiversity monitoring and biodiversity accounting is the Group on Earth Observation – Biodiversity Observation Network (GEO-BON) (Scholes et al., 2008; Jones et al., 2011; Pereira et al., 2013).

### *6.4.2 Combining biodiversity and ecosystem services in accounting*

Two distinctly different roles of biodiversity need to be included in an ecosystem accounting framework, reflecting the two ways it is being included in ecosystem service research (Malinga et al., 2015). First, as outlined above, an account that covers indicators related to species diversity (i.e. biodiversity as a cultural ES). The account fills a gap that is not covered by the supply and use account for ES flows, as appreciation of species existence is not covered in the classifications by the Millennium Ecosystem Assessment (MA, 2005), The Economics of Ecosystems and Biodiversity (TEEB, 2010) and the Common International Classification of Ecosystem Services (CICES; Haines-Young and Potschin, 2010b). Second, the role of biodiversity in regulating ecosystem processes and conditions should be included an ecosystem accounting system, at least for those conditions that generate capacity to provide ESs. Such information would connect biodiversity accounts and condition accounts. Determining which aspects of biodiversity are important for generating ESs is notoriously difficult (Hooper et al., 2005; Cardinale et al., 2012), but for an increasing amount of biodiversity aspects their effects on ES provision are becoming clearer (Cardinale et al., 2012; Balvanera et al., 2014; Harrison et al., 2014). Information on relationships between biodiversity and ESs are likely to become more readily available in the coming years, with the establishment of the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) (Inouye, 2014; Díaz et al., 2015). One of IPBES's main aims is to synthesize information on the relationship between biodiversity and ESs (Balvanera et al., 2014). A main challenge in linking biodiversity and ecosystem condition in an accounting system will be to identify the most relevant biodiversity indicators to include in accounts. To select relevant indicators an approach could be to focus on functional diversity relevant for generating ESs (Maes et al., 2012a; Díaz et al., 2013).

## 6.5 Including ecosystem services in spatial biodiversity conservation approaches

Exploiting one or more ESs could causes losses of other ESs, especially of regulating and cultural services (Raudsepp-Hearne et al., 2010; Goldstein et al., 2012). To minimize such losses, conservation-compatible ESs (cf. Chan et al., 2011) should be included in conservation. However, including ESs in conservation should ideally not go at the expense of biodiversity conservation. In Chapter 5, I show that including such ESs in expanding Limburg's conservation network does not negatively affect the effectiveness of achieving biodiversity targets and increases the effectiveness of achieving set ES targets. Therefore, the inclusion of ESs into the biodiversity conservation network is likely to enhance the maintenance of conservation-compatible ESs. Combining ES and biodiversity in the conservation network does substantially increase the necessary implementation budget. In terms of increased effectiveness of combining ESs and biodiversity in a conservation network, similar results have been presented for other areas where ESs are included in spatial conservation assessments (Izquierdo and Clark, 2012; Schröter et al., 2014c; Adame et al., 2015). A side effect of including conservation-compatible ES in conservation planning could be that some other ESs are negatively affected (e.g. agricultural ESs, Maes et al., 2012a), as the area for their production decreases. Nevertheless, integrating biodiversity and conservation-compatible ESs into conservation planning would provide a more effective way to safeguard landscape multifunctionality (Mastrangelo et al., 2014).

For Europe, Maes et al. (2012a) found that areas with high biodiversity are positively spatially correlated with the provision of ESs. In general, for local and national ES conservation planning, the spatial relationship between ES and biodiversity is positive, but ranges from low to moderate (Cimon-Morin et al., 2013). For Limburg, I analysed the spatial correlation between the ESs presented in Chapters 2 and 3 and species richness for butterflies, birds, vascular plants, dragonflies and amphibians presented in Chapter 4. This analysis is additional to the research presented in previous chapters. The spatial correlation between each ES and the different species groups was weak (Pearson's $r < 0.25$, Appendix IV

Table AIV.1). Spatial correlations were positive for most ESs, including all conservation-compatible ESs (drinking water production, carbon sequestration, air quality filtration, recreational cycling and nature tourism, see Chapter 5). Two not conservation-compatible ESs (crop production and fodder production) are weakly, negatively correlated with species richness. A functional relationship between the tested species groups and the ESs is not evident, which could explain the weak correlations. Spatial correlation between biodiversity and, for example, carbon sequestration may have been higher if tree diversity had been used as a biodiversity indicator. Differences in spatial patterns between biodiversity and conservation-compatible ESs provide an extra incentive for including ESs into conservation planning, as purely biodiversity focussed conservation networks are unlikely to fully ensure future ES provision. Integrated conservation networks provide the most effective approach to spatially ensure biodiversity protection and ESs provision.

Conservation networks are one approach that can be taken to conserve biodiversity and ESs, but other approaches should also be assessed. For example, different gradients of protection can be applied, using stringent protection zones without human influence, but also partial use can be applied, combining human land-uses with conservation measures (Schröter et al., 2014c). Although protected areas have shown to be a cost-effective form of biodiversity conservation (Balmford et al., 2002), their effectiveness for ES conservation remains to be further explored. Fully protecting areas may not be the most effective form of conservation for ESs, especially since humans need to be present on location or in the proximity to use many ESs (Fisher et al., 2009). In further research, a broader range of measures needs to be assessed, including management practices, that can be applied for ES conservation in combination with maintaining biodiversity.

## 6.6 Challenges and ways forward for ecosystem accounting

### *6.6.1 Conceptual model for ecosystem accounting*

Ecosystem accounting is rapidly developing into an applicable approach to measure and monitor ecosystems. A major step in this respect, laying the basis for future studies and pilots, was the publication of the SEEA-EEA guidelines by UN et al. (2014a). Currently, most of ecosystem accounting work has been conceptual and even the conceptual basis is still evolving (with new technical guidelines scheduled to be published by UNSD in November 2015). Based on current knowledge on ecosystem accounting, building upon the SEEA-EEA guidelines, I propose a conceptual model that integrates its different components and relationships (Figure 6.3). The model applies the various accounts introduced in the SEEA-EEA (UN et al., 2014a), including a condition account, capacity account, supply and use account, demand account and a biodiversity account. The model integrates the preliminary ecosystem accounting model of the SEEA-EEA (UN et al., 2014a), a conceptual model introduced on assets and ESs (Obst et al., 2015), on condition, capacity and flow (Schröter et al., 2014b), on the human contribution (Chapter 2) and on the role of biodiversity (Chapter 4). The conceptual model shows the relations between the different accounts.

The condition account and capacity account together describe ecosystem state. The capacity account and supply and use account for ES flows together give an indication of under- or overuse, and ultimately sustainable use of ESs (Schröter et al., 2014b). The supply and use account and demand account together indicate who is currently using ESs and whether current demand can be met locally. Biodiversity has both an underpinning role related to ecosystem processes and conditions and as a contribution to human well-being (i.e. an ES). A biodiversity account in its suggested form focusses mainly on the latter role, but could be expanded in the future to also further incorporate the underpinning role. The human contribution is likely often inextricably entangled with ecosystem processes and conditions (Chapter 2), and therefore will be partially included in the different accounts. Moreover, human induced pressures and management also function as drivers of change and are therefore an integral part of the conceptual model. The conceptual model reflects the relation between ecosystems and society in a simplified way, but shows that an integrated set of accounts is needed to cover the multifaceted system. Only an integrated set of accounts will allow to measure and monitor changes in ecosystems and how humans use affects them. Further testing several of the ecosystem accounts is needed, but also of an integrated set of accounts, as whether the suggested accounts offer comprehensive information needed to monitor ecosystems, is currently unknown.

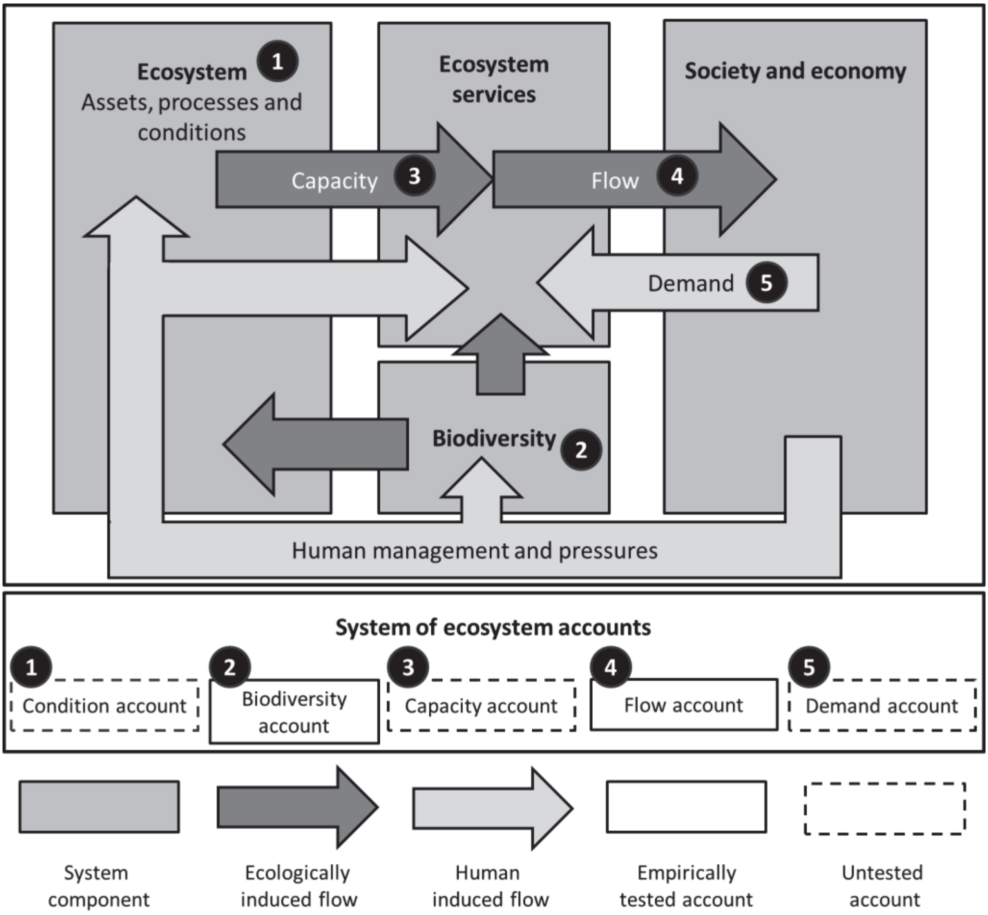


**Figure 6.3** Conceptual model of the human-ecosystem relationship aspects that are captured by ecosystem accounting. The framework is adapted from Schröter et al. (2014b), UN et al. (2014a) and Figure 2.1.

### *6.6.2 Towards integrated ecosystem accounts*

Calls for standardising definitions, terminology, data collection and appropriate methods in ES research in different use contexts are increasing (Boyd and Banzhaf, 2007; Nahlik et al., 2012; Polasky et al., 2015). Ecosystem accounting represents such a use context for which the road to standardisation is rapidly being paved. Currently no international statistical standard for ecosystem accounting exist. However, the SEEA-EEA has laid the foundation for developing such a standard (UN et al., 2014a) and is rapidly preparing for further operationalisation of ecosystem accounting (e.g. UN et al., 2015). The main purpose and guidelines for ecosystem accounting have been conceptualised, and its boundaries have been set (UN et al., 2014, with further technical details on specific topics provided by Obst and Vardon, 2014; Hein et al., 2015; Obst et al., 2015 among others). At least sixty-nine countries have currently committed to some form of accounting for ecosystem services (Polasky et al., 2015), and there is a growing interest on the topic from decision makers. The first empirical ecosystem accounting modelling studies have been assessed to develop best practices (Schröter et al., 2015). All in all, the progress in operationalising ecosystem accounting is rapid and is only likely to increase with increasing demand for information from decision makers.

Nevertheless, the development of ecosystem accounting is not complete and several challenges still remain (UN et al., 2014a; Hein et al., 2015; Obst et al., 2015). A first challenge is the integration with national accounting, as ecosystems are inherently different from economic sectors included in national accounts. Ecosystems provide many different ESs with incommensurable biophysical units and cannot always be valued using standard market value approaches. Whether ecosystem accounting can be fully integrated into national accounting is uncertain. Open challenges need to be solved in terms of connecting ecosystems to economic sectors and double counting certain ecosystem goods that are already in national accounts (Obst et al., 2015). In a critical commentary, the approach proposed in the SEEA-EEA has been described as a detour for greening national accounts, as this approach does not provide full integration of ecological information and economic data (Bartelmus, 2015). This author argues that the suggested approach is not sufficiently rigorous to integrate with the rigid SNA, stating that "[classical accounting] conventions and rules are bound to distort complex processes within and between ecosystems". Although full integration of ecosystem accounting and SNA is yet to be established, my thesis shows that integration of ecological information and economic data in the context of ecosystem accounting is already feasible for many ESs.

Besides integration of ecosystem accounting with SNA, another important challenge is that remains is the integration of the different types of ecosystem accounts. For full integration several challenges remain. First, the types of accounts that are yet to be empirically assessed (the condition account, capacity account and demand account), require attention. Apart from separately testing the different types of accounts, their compatibility with each other should be assessed. Second, as an important part of ecosystem accounting is monitoring change, future research should focus on analysing temporal changes. Such analysis is necessary to evaluate if detecting ecosystem degradation is possible. A large hurdle in current accounting research has been the limited data availability and accessibility (Hein et al., 2015; Schröter et al., 2015; Sumarga et al., 2015). Calls for investments in ecosystem data specifically geared at accounting have been made (e.g. Obst et al., 2013), and technological developments are improving data collection opportunities (e.g. satellite data, citizen science initiatives through mobile technology) (Starr et al., 2014; Petrou et al., 2015). Therefore, an increase in the quantities of available environmental data is very plausible. A crucial challenge will be to focus data collection efforts on aspects that are relevant for ecosystem accounting.

In the further development and integration of ecosystem accounting the focus on a spatial approach should be maintained. The major strength of ecosystem accounting is its ability to capture multiple relations between ecosystems and humans from a spatial perspective, and actively recording these spatial variations and changes to feed into and evaluate policies at various administrative tiers. This strength is even underlined by its critics (e.g. Bartelmus, 2015). Ecosystem accounting is an extensive, integrated information system that can be used as an instrument for evaluating and developing policies. A system for ecosystem accounting should align itself with SNA as much as possible, but some deviations might be inescapable. For example, whether all regulating and cultural ESs can be monetarily valued without including components of consumer surplus, remains unclear (Hein et al., 2015). Without further research on SNA-conform valuation methods, a rigid SNA-aligned system could result in the exclusion of such ESs, and, therefore, in accounts that are not fully comprehensive. As comprehensiveness is one of the key components of ecosystem accounting, compromising SNA-alignment to some extent might be necessary on this issue. Where possible monetary information from ecosystem accounting should feed into national accounting, to inform end users on the influence of ecosystems in various economic sectors (i.e. SNA benefits, cf. Chapter 3) and a minimal value ecosystems contribute that are not included in national accounts (i.e. non-SNA benefits, cf. Chapter 3). I refer to a minimal value because currently not all cultural and regulating ESs can be included in ecosystem accounting, and if included, they may reflect only part of the ES. For example, Sumarga et al. (2015) value orang-utan habitat as a surrogate for the ES wildlife habitat. To obtain the full ES value, habitat for all species would need to be valued. For many cases indicators that only partially cover the ES are necessary, due to low data availability.

## 6.7 Applying ecosystem service and biodiversity modelling for accounting, conservation and management

The CBD's Aichi Biodiversity Targets (UNEP, 2010) and the EU's Biodiversity Strategy (European Commission, 2011) both explicitly include ESs alongside biodiversity in several individual targets. The Aichi targets address conservation, protection and restoration of ESs and biodiversity. Both strategies include calls to develop accounting and reporting systems for ESs and biodiversity. The EU strategy explicitly requires member states to "map and assess the state of ecosystems and their services in their national territory" (European Commission, 2011). The rapid conceptual development of ecosystem accounting and the SEEA-EEA supports such calls, and the international interest for ecosystem accounting indicates that the calls are being taken seriously. The spatial modelling work on ESs and biodiversity I present in this thesis is an essential contribution to achieving the various goals and developing reporting and accounting systems. Without models the information demand of ecosystem accounting systems cannot be met.

A question that remains to be answered, is to what extent ecosystem accounts can meet expectations and needs of decision makers (Radermacher and Steurer, 2015), and how spatial modelling can contribute to meeting these needs. Spatial biodiversity and ES modelling outcomes can support different policy purposes including awareness raising, accounting, priority setting and instrument design (Gómez-Baggethun and Barton, 2013). Biodiversity and ES maps can raise awareness on how humans use and depend on them. In this thesis I show how spatial models can be used for accounting (Chapters 2, 3 and 4) and priority setting (Chapter 5). Finally, spatial models can be used for instrument design (e.g. to develop payment schemes for ESs). For each of the policy purposes a niche can be defined related to which modelling methods can reliably be applied (Schröter et al., 2015). The niches overlap and a spatial modelling method can serve multiple purposes. For example, the ES models that I developed for ecosystem accounting (Chapters 2 and 3), could also be applied for priority setting (Chapter 5). Ecosystem accounting is primarily an integrated information system (Obst and Vardon, 2014). However, the approach can be used to evaluate policy related to ESs, biodiversity conservation and land management, as a result of the necessary monitoring efforts. The output allows for monitoring trends and assessing management strategies, and can be used as input for assessing scenarios and developing new management strategies related to sustainability and land and resource management (Hein et al., 2015; Radermacher and Steurer, 2015).

The exact policy implications of ecosystem accounting will become more clear in coming years. Many countries have committed to forms of ecosystem accounting (Polasky et al., 2015), and the first reports from statistical agencies on different types of ecosystem accounts have appeared (e.g. Bond et al., 2013; Kahn et al., 2014). The work of the statistical agencies is still experimental, but the important steps towards operationalising ecosystem accounting are being made. The ES research from this thesis contributes to the first experimental ecosystem account in the Netherlands, which is being developed for Limburg by Statistics Netherlands in cooperation with the Environmental Systems Analysis group. Such an experimental account could provide the necessary insights to upscale the approach to the national scale. Combining the insights from this experimental account with the data available from the Dutch national ES mapping project Atlas Natural Capital (www.atlasnatuurlijkkapitaal.nl), could result in an encompassing Dutch ecosystem account.

## 6.8 Conclusions

To operationalize ecosystem accounting and improve spatial conservation planning a suite of different indicators are needed that represent both ESs and biodiversity. In this thesis, I show that biophysical and monetary indicators are complementary to account for ESs flows and that multiple indicators are needed to comprehensively account for biodiversity. I showed how spatial ES and biodiversity models can support ecosystem accounting, management and conservation. My research has applied several main concepts and modelling strategies that have been discussed in the ecosystem accounting community, and tested to what extent they can be used to develop actual accounts. An addition, I have assessed how spatial information on ESs and biodiversity can contribute to more cost-effective planning of conservation networks.

My research provides one of the first empirical assessment of both biophysical and monetary ES flows in the context of ecosystem accounting. I showed that developing spatial models for flows of many different ES types and valuing these ESs with SNA-consistent methods is feasible. I conclude that, if both an ecosystem and human contribution are needed for societal benefit, monetary valuation methods are more effective to disentangle these contributions than biophysical indicators. Especially the resource rent method provides an effective method to disentangle such contributions. In many cases biophysical and monetary indicators provide different insights in ESs flows, highlighting that they are complementary. Moreover, biophysical indicators are necessary to develop monetary accounts. Currently, not all ESs can be monetarily valued in ecosystem accounting yet. SNA-aligned methods to value some cultural and regulating ESs still need to be further developed and tested in an ecosystem accounting context to ensure reliable applications.

Relevant data for ecosystem accounting exists on many ESs, environmental, socio-economic variables and biodiversity, but this data tends to be very scattered and may not always be accessible due to, for example, financial constraints or concerns for confidentiality. This hampers effective accounting. Therefore, in addition to further investing in the improvement of spatial modelling and valuation techniques, investments are needed in increasing data availability and accessibility for ecosystem accounting. Modern techniques, such as high resolution and up-to-date satellite imagery and big data opportunities through advanced and readily available technology (e.g. smartphones) increase the potential for data collection. Such techniques increase the potential to develop ecosystem accounts for data poor areas, but also increase the potential to collect all necessary environmental and socio-economic data in data rich areas such as the Netherlands.

This thesis shows that ESs and biodiversity should both be included in ecosystem accounting, ecosystem management and developing conservation networks. The spatial distributions of ESs and biodiversity differ. One aspect cannot be used to indicate the other. Therefore, both aspects should be measured and monitored regularly and combined in an ecosystem accounting system. In addition, combining ESs and biodiversity in ecosystem management and spatial conservation planning can increase the cost-effectiveness of the applied conservation strategy.

ESs are increasingly being included in different types of ecosystem management and biodiversity conservation strategies. An operational ecosystem accounting system ensures that sufficient information is collected on ESs and biodiversity to monitor not only the conditions, capacity, flow and demand, but also the effectiveness of management and conservation. This thesis has substantially contributed to operationalising ecosystem accounting by empirically testing applicable indicators, spatial modelling methods and monetary valuation methods, delineating caveats and assessing uncertainties. The quantity of studied ESs can be expanded and more ecosystem accounting aspects, such as ecosystem condition, capacity and societal demand need to be covered and integrated, before ecosystem accounting is put into full practice. Nevertheless, my thesis shows that ecosystem accounting withstood important empirical tests regarding ES flows and valuation. This thesis has provided a number of necessary insights and approaches to further develop ecosystem accounting and link these approaches to management and conservation strategies. Specifically, I show that including ESs in the design of biodiversity conservation networks likely increases their cost-effectiveness and conserves more aspects of biodiversity and ESs with societal and ecological importance. Ecosystem accounting, management and conservation are all necessary to ensure the safeguarding of natural and managed ecosystems and for sustaining human well-being.

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# Appendix I

Additional information for Chapter 3

**Biophysical nature tourism model**

The biophysical allocation model for nature tourism estimated the number of tourists in nature areas, assuming that they stayed within a 15 km radius of their accommodation. This 15 km radius was also used in a comparable model for nature-based recreation in the Netherlands (de Vries and Goossen, 2002). The model was set up based on a point dataset with the location of hotels, camp sites and holiday parks (Stichting Recreatie, 2007). Further data was only available at an at the scale of three sub-regions of Limburg, in the form of aggregated statistics. The number of different accommodation types and total overnight stays for the three sub-regions were derived from Statistics Netherlands (2013d) and ZKA Consultants & Planners (2011) (Table A1). To calculate the total overnight stays the number of beds per accommodation type (Statistics Netherlands, 2013d) were multiplied by the average annual occupation rate of accommodations in the Netherlands, derived from Statistics Netherlands (2014a). To calculate the number of tourists visiting nature the total overnight stays were divided by the average length of a holiday (Statistics Netherlands, 2013d) and multiplied by the fraction of tourists visiting nature areas (ZKA Consultants & Planners, 2011). The fraction of tourists visiting nature areas depends on how urbanized the surroundings of an accommodation are, assuming that the portion of tourists visiting nature increases with decreasing degree of urbanization (Table A2). To determine the degree of urbanization a five point scale similar to that of Statistics Netherlands (2014b) has been used, with the difference that the categories used here are based on population densities. Based on the combination of statistics described above nature tourists were allocated evenly to the nature areas within the 15 km radius around each the accommodation. Nature areas were considered to be all areas that fall under a form of nature protection policy.

**Table AI.1** Statistics on tourism in three sub-regions of Limburg. Data sources: ZKA Consultants & Planners (2011) and Statistics Netherlands (2013d).

| | North Limburg | Central Limburg | South Limburg |
|---|---|---|---|
| **Number of accommodations** | | | |
| Total | 144 | 104 | 452 |
| *Hotels* | *49* | *48* | *323* |
| *Camp sites* | *70* | *30* | *38* |
| *Holiday parks* | *25* | *26* | *91* |
| | | | |
| **Overnight stays** | | | |
| Total | 3,804,469 | 1,410,534 | 3,918,207 |
| *Hotels* | *359,370* | *218,509* | *2,047,738* |
| *Camp sites* | *487,862* | *273,426* | *612,850* |
| *Holiday parks* | *2,957,237* | *918,599* | *1,257,620* |
| | | | |
| **Average length holiday (days)** | 5.2 | 5.8 | 5.0 |
| **Tourists visiting nature (%)** | 58 | 55 | 54 |

**Table AI.2** Percentage of tourists visiting nature, depending on the degree of urbanization of the area surrounding their accommodation. The bold figures indicate the average percentage for tourists walking in nature as indicated by ZKA Consultants & Planners (2011).

| Category for degree of urbanisation | Population density (inhabitants/ km2) | # of accommodations | % of tourists visiting nature | | |
|---|---|---|---|---|---|
| | | | North Limburg | Central Limburg | South Limburg |
| **1** | <500 | 327 | 68 | 65 | 64 |
| **2** | 500-1000 | 80 | 58 | 55 | 54 |
| **3** | 1000-1500 | 59 | 48 | 45 | 44 |
| **4** | 1500-2500 | 132 | 38 | 35 | 34 |
| **5** | >2500 | 102 | 28 | 25 | 24 |

**Table AI.3** Health impact categories, including categories related to willingness-to-pay, resulting from PM10 concentration change, their physical impact on a person and the monetary value of the treatment costs. Physical impacts and treatment costs are adapted from Preiss et al. (2008), unless stated otherwise.

| Health impact categories | Physical impact per person per µg $PM_{10}$ (1/(µg/m³)) | Treatment costs per case for 2010 (€) | Costs per person per µg $PM_{10}$ (€/person/µg/m³) |
|---|---|---|---|
| Work loss days | $1.39 * 10^{-2}$ | 362 | 5.03 |
| New case chronic bronchitis | $1.86 * 10^{-5}$ | 22,748[a] | 0.42 |
| Respiratory hospital admission | $7.03 * 10^{-6}$ | 2453 | 0.02 |
| Cardiac hospital admission | $4.34 * 10^{-6}$ | 2453 | 0.01 |
| Medication/bronchilator use child | $4.03 * 10^{-4}$ | 1.23 | 0.00 |
| Medication/bronchilator use adult | $3.27 * 10^{-3}$ | 1.23 | 0.00 |
| Lower respiratory symptoms adult | $3.24 * 10^{-2}$ | 47 | 1.51 |
| Lower respiratory symptoms child | $2.08 * 10^{-2}$ | 47 | 0.97 |
| Years of life lost[b] | $6.51 * 10^{-4}$ | 49,062 | 31.94 |
| Net restricted activity days[b] | $9.59 * 10^{-3}$ | 159 | 1.53 |
| Minor restricted activity days[b] | $3.69 * 10^{-2}$ | 47 | 1.72 |
| Increased mortality risk infants[b] | $6.84 * 10^{-8}$ | 3,679,677 | 0.25 |
| Total avoided costs per person per avoided $PM_{10}$ concentration increase | | | 43.40 |

[a] adapted from RIVM (2012).

[b] categories including willingness-to-pay.

# Appendix II

Additional information for Chapter 4.

**Table AII.1:** Analysed species groups and an assessment of data quality based on the completeness of the spatial data per group. The grey shading indicates the species groups with a known completeness of at least 75% and an average to good completeness of at least 25%. These species groups were used for composite indicator BD5.

| | Completeness of data (% of cells) | | |
|---|---|---|---|
| **Species group** | **Average to good** | **Poor** | **Unknown** |
| Butterflies | 95 | 4 | 1 |
| Vascular plants | 91 | 8 | 1 |
| Birds | 79 | 18 | 2 |
| Dragonflies | 80 | 12 | 8 |
| Amphibians | 38 | 37 | 25 |
| Grasshoppers and crickets | 39 | 10 | 52 |
| Reptiles | 31 | 16 | 52 |
| Mosses | 22 | 26 | 52 |
| Macro moths | 12 | 51 | 38 |
| Lichens | 7 | 19 | 74 |
| Micro moths | 6 | 18 | 76 |
| Mushrooms | 2 | 6 | 91 |
| Mammals | 0 | 98 | 1 |
| Other vertebrae | 0 | 0 | 100 |

**Table AII.2:** Spatial correlation (Pearson's r) between species richness indicators.

| **Indicator** | BD1 | BD2 | BD5 | BD14 | Average |
|---|---|---|---|---|---|
| BD1 | - | 0.85 | 0.78 | 0.75 | 0.79 |
| BD2 | 0.85 | - | 0.87 | 0.80 | 0.84 |
| BD5 | 0.78 | 0.87 | - | 0.94 | 0.86 |
| BD14 | 0.75 | 0.80 | 0.94 | - | 0.83 |

**Table AII.3** Spatial correlation (Pearson's r) between butterfly richness indicators at 250 m and 1000 m resolution.

| Dataset | Vlinder-stichting 250m | Vlinder-stichting 250m upscaled to 1000m | NDFF 1000m |
|---|---|---|---|
| Vlinderstichting 250m | | 0.85 | 0.54 |
| Vlinderstichting 250m upscaled to 1000m | 0.85 | | 0.66 |
| NDFF 1000m | 0.54 | 0.66 | |

**Table AII.4** Spatial correlation (Pearson's r) between Red List butterfly richness indicators at 250 m and 1000 m resolution.

| Dataset | Vlinder-stichting 250m | Vlinder-stichting 250m upscaled to 1000m | NDFF 1000m |
|---|---|---|---|
| Vlinderstichting 250m | | 0.65 | 0.26 |
| Vlinderstichting 250m upscaled to 1000m | 0.65 | | 0.40 |
| NDFF 1000m | 0.26 | 0.40 | |

**Table AII.5** Mean number of butterfly species per grid cell and standard deviation.

| Dataset | All butterflies | | Red List butterflies | |
|---|---|---|---|---|
| | Mean per grid cell | SD | Mean per grid cell | SD |
| Vlinderstichting 250m | 6.6 | 4.4 | 0.2 | 0.5 |
| Vlinderstichting 250m upscaled to 1000m | 10.8 | 6.4 | 0.6 | 1.0 |
| NDFF 1000m | 17.9 | 8.2 | 1.8 | 2.9 |

# Appendix III

Additional information for Chapter 5.

**Table AIII.1:** Input features, their units, source, total availability and the set targets.

| Feature type | Feature name | Source | Unit | Provincial total | Protected in current network | Target |
|---|---|---|---|---|---|---|
| **Ecosystem service** | Air quality regulation | Remme et al. (2015) | €/yr | 2061887 | 795466 | 1001654 |
| **Ecosystem service** | Carbon sequestration | Remme et al. (2014) | tC/yr | 70014 | 43625 | 50627 |
| **Ecosystem service** | Drinking water extraction | Remme et al. (2014) | m3/yr | 26966482 | 6272014 | 8968662 |
| **Ecosystem service** | Nature tourism | Remme et al. (2015) | €/yr | 38690522 | 26954015 | 30823067 |
| **Ecosystem service** | Recreational cycling | Remme et al. (2014) | trips/yr | 9073242 | 1898117 | 2805441 |
| **Biodiversity policy goal** | Foraging area for geese | Provincie Limburg (2015a) | ha | 207 | 29 | 104 |
| **Biodiversity policy goal** | Habitat for European hamster | Provincie Limburg (2015a) | ha | 6261 | 466 | 766 |
| **Biodiversity policy goal** | Habitat for farmland birds | Provincie Limburg (2015a) | ha | 13434 | 310 | 330 |
| **Biodiversity policy goal** | Habitat for meadow birds | Provincie Limburg (2015a) | ha | 779 | 22 | 112 |
| **Biodiversity policy goal** | Hedgerows | Provincie Limburg (2015a) | meters | 581569 | 12560 | 87235 |
| **Biodiversity policy goal** | Standard orchards | Provincie Limburg (2015a) | ha | 3284 | 155 | 328 |

Table AIII.1(continued)

| Feature type | Feature name | Source | Unit | Provincial total | Protected in current network | Target |
|---|---|---|---|---|---|---|
| **Biodiversity, species** | Black-tailed godwit | Provincie Limburg (2015b) | ha | 350 | 26 | 61 |
| **Biodiversity, species** | Common midwife toad | Alterra (2015) | ha | 4455 | 1365 | 1811 |
| **Biodiversity, species** | Common spadefoot | Alterra (2015) | ha | 1099 | 633 | 743 |
| **Biodiversity, species** | Corn bunting | Provincie Limburg (2015b) | ha | 16 | 4 | 6 |
| **Biodiversity, species** | Corn crake | Provincie Limburg (2015b) | ha | 4 | 3 | 4 |
| **Biodiversity, species** | Dusky large blue | Alterra (2015) | ha | 814 | 155 | 236 |
| **Biodiversity, species** | Eurasian curlew | Provincie Limburg (2015b) | ha | 1168 | 106 | 223 |
| **Biodiversity, species** | European beaver | Alterra (2015) | ha | 23158 | 7666 | 9982 |
| **Biodiversity, species** | European tree frog | Alterra (2015) | ha | 1975 | 520 | 718 |
| **Biodiversity, species** | European weather loach | Alterra (2015) | ha | 236 | 133 | 157 |
| **Biodiversity, species** | Floating water-plantain | Alterra (2015) | ha | 2339 | 930 | 1164 |

Table AIII.1(continued)

| Feature type | Feature name | Source | Unit | Provincial total | Protected in current network | Target |
|---|---|---|---|---|---|---|
| **Biodiversity, species** | Geoffroy's bat | Alterra (2015) | ha | 3229 | 1164 | 1487 |
| **Biodiversity, species** | Green snaketail | Alterra (2015) | ha | 1530 | 666 | 819 |
| **Biodiversity, species** | Grey long-eared bat | Alterra (2015) | ha | 200 | 83 | 103 |
| **Biodiversity, species** | Grey partridge | Provincie Limburg (2015b) | ha | 4871 | 336 | 823 |
| **Biodiversity, species** | Hazel dormouse | Alterra (2015) | ha | 3417 | 1693 | 2035 |
| **Biodiversity, species** | Jersey tiger | Alterra (2015) | ha | 2580 | 780 | 1038 |
| **Biodiversity, species** | Moor frog | Alterra (2015) | ha | 8077 | 3116 | 3924 |
| **Biodiversity, species** | Northern crested newt | Alterra (2015) | ha | 8817 | 3888 | 4770 |
| **Biodiversity, species** | Red-backed Shrike | Provincie Limburg (2015b) | ha | 158 | 124 | 140 |
| **Biodiversity, species** | Skylark | Provincie Limburg (2015b) | ha | 17421 | 1605 | 3347 |
| **Biodiversity, species** | Stag beetle | Alterra (2015) | ha | 10605 | 1828 | 2889 |
| **Biodiversity, species** | Yellow-bellied toad | Alterra (2015) | ha | 792 | 290 | 369 |

**Table AIII.2:** Specific Marxan settings per budget simulation, including number of achieved features and achieved boundary length.

| ES and biodiversity scenario | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| **Network expansion costs (million €)** | **Boundary length modifier** | **Species penalty factor** | **Cost thres hold** | **Thres-hold penalty** | **Number of input features** | **Number of features for which targets were achieved** | **Boundary length (km)** | **Application in stepwise approach** |
| 32 | 5000 | 1 | 32 | 4000 | 34 | 11 | 3602 | Current conservation budget and step 1 in stepwise approach, period 2013-2015 |
| 84 | 5000 | 1 | 84 | 1500 | 34 | 19 | 3719 | |
| 200 | 5000 | 1 | 200 | 1000 | 34 | 24 | 4182 | |
| 400 | 5000 | 1 | 400 | 500 | 34 | 29 | 5047 | |
| 500 | 5000 | 1 | 500 | 400 | 34 | 30 | 5403 | |
| 600 | 5000 | 1 | 600 | 300 | 34 | 31 | 5685 | |
| 700 | 5000 | 1 | 700 | 200 | 34 | 32 | 5776 | |
| 800 | 5000 | 1 | 800 | 200 | 34 | 33 | 5593 | |
| 1000 | 5000 | 1 | 1000 | 50 | 34 | 34 | 4886 | |
| 1200 | 5000 | 1 | 1200 | 20 | 34 | 34 | 3838 | |
| 1447 | 5000 | 1 | 0 | 1 | 34 | 34 | 3214 | |
| 52 | 5000 | 1 | 53 | 2500 | 34 | 17 | 3742 | Stepwise (2/2), step 2016-2020 |
| 32 | 5000 | 1 | 32 | 4000 | 34 | 15 | 3728 | Stepwise (2/3), step 2016-2018 |
| 20 | 5000 | 1 | 21 | 6000 | 34 | 15 | 3851 | Stepwise (3/3), step 2019-2020 |

Table AIII.2 – continued.

| Biodiversity scenario | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Network expansion costs (million €) | Boundary length modifier | Species penalty factor | Cost thres hold | Thres-hold penalty | Number of input features | Number of features for which targets were achieved | Boundary length (km) | Application in stepwise approach |
| 32 | 5000 | 1 | 32 | 4000 | 29 | 11 | 3605 | Current conservation budget and step 1 in stepwise approach, period 2013-2015 |
| 84 | 5000 | 1 | 84 | 1200 | 29 | 16 | 3748 | |
| 200 | 5000 | 1 | 200 | 500 | 29 | 26 | 4090 | |
| 400 | 5000 | 1 | 400 | 200 | 29 | 28 | 5071 | |
| 458 | 5000 | 1 | 500 | 100 | 29 | 29 | 3283 | |
| 511 | 5000 | 1 | 600 | 50 | 29 | 29 | 3170 | |
| 566 | 5000 | 1 | 700 | 20 | 29 | 29 | 3089 | |
| 705 | 5000 | 1 | 0 | 1 | 29 | 29 | 2994 | |
| 52 | 5000 | 1 | 53 | 2500 | 29 | 12 | 3751 | Stepwise (2/2), step 2016-2020 |
| 32 | 5000 | 1 | 32 | 4000 | 29 | 14 | 3740 | Stepwise (2/3), step 2016-2018 |
| 20 | 5000 | 1 | 21 | 6000 | 29 | 15 | 3854 | Stepwise (3/3), step 2019-2020 |

# Appendix IV

Additional information for Chapter 6.

**Table AIV.1** Spatial correlation (Pearson's r) for Limburg between ecosystem service indicators (biophysical and monetary) from Chapters 2 and 3 and species richness indicators from Chapter 4. Composite species richness indicator BD5 consists of the species groups butterflies, birds, vascular plants, dragonflies and amphibians.

| Ecosystem service indicator | Species richness indicator | | | | | |
|---|---|---|---|---|---|---|
| | Composite indicator (BD5) | Butterflies | Birds | Vascular plants | Dragonflies | Amphibians |
| *Biophysical indicator* | | | | | | |
| Crop production | -0.136** | -0.128** | -0.114** | -0.080** | -0.083** | -0.106** |
| Fodder production | -0.082** | -0.067** | -0.038** | -0.073** | -0.052** | -0.076** |
| Drinking water production | 0.070** | 0.071** | 0.026* | 0.115** | 0.020* | 0.039** |
| Hunting | 0.176** | 0.089** | 0.046** | 0.070** | 0.219** | 0.228** |
| Air quality regulation | 0.105** | 0.086** | 0.002 | -0.082** | 0.203** | 0.156** |
| Carbon sequestration | 0.128** | 0.110** | 0.049** | 0.021* | 0.146** | 0.148** |
| Recreational cycling | 0.032** | 0.013 | -0.033** | 0.125** | -0.031** | 0.051** |
| Nature tourism | 0.196** | 0.170** | 0.137** | 0.199** | 0.089** | 0.154** |

**Correlation is significant at the 0.01 level (2-tailed).
*Correlation is significant at 0.05 level (2-tailed).

Table AIV.1 (continued)

| Ecosystem service indicator | Species richness indicator | | | | | |
|---|---|---|---|---|---|---|
| | Composite indicator (BD5) | Butterflies | Birds | Vascular plants | Dragonflies | Amphibians |
| *Monetary indicator* | | | | | | |
| Crop production | -0.138** | -0.126** | -0.112** | -0.102** | -0.078** | -0.107** |
| Fodder production | -0.082** | -0.067** | -0.038** | -0.073** | -0.052** | -0.076** |
| Drinking water production | 0.070** | 0.071** | 0.026** | 0.116** | 0.020* | 0.039** |
| Hunting | 0.031** | 0.014 | -0.019 | -0.067** | 0.104** | 0.069** |
| Air quality regulation | 0.106** | 0.075** | -0.002 | 0.133** | 0.079** | 0.100** |
| Carbon sequestration | 0.134** | 0.115** | 0.060** | 0.018 | 0.154** | 0.149** |
| Nature tourism | 0.225** | 0.203** | 0.116** | 0.246** | 0.097** | 0.200** |
| Total ES value | 0.017 | 0.015 | -0.032** | 0.064** | -0.008 | 0.029** |
| Public ES value | 0.237** | 0.214** | 0.118** | 0.261** | 0.106** | 0.209** |
| Private ES value | -0.156** | -0.142** | -0.123** | -0.119** | -0.088** | -0.122** |

**Correlation is significant at the 0.01 level (2-tailed).
*Correlation is significant at 0.05 level (2-tailed).

## Summary

Humans depend on biodiversity and ecosystem contributions to maintain their quality of life. Ecosystem contributions are referred to as ecosystem services and they link ecosystem processes and societal wellbeing. Biodiversity is the variability among all sources of living organisms, including the diversity within species, between species and of ecosystems. Ecosystem services and biodiversity need to be conserved to maintain a well-functioning earth system. Calls to systematically measure ecosystem services and biodiversity are increasing (e.g. the Aichi Targets, and the EU's Biodiversity Strategy), and ecosystem accounting is an approach that answers to this call.

Ecosystem accounting is a systematic approach to measure and monitor ecosystem services, ecosystem conditions and biodiversity over time and explicitly focusses on spatial approaches to capture the spatial heterogeneity of ecosystems and ecosystem service provision. The foundations for ecosystem accounting have been laid in recent years, with a clear relation to national accounting, a defined purpose and terminology, an analysis of the necessary components and their relationships and a preliminary set of internationally recognised guidelines developed under the auspices of the United Nations, the System of Environmental-Economic Accounting - Experimental Ecosystem Accounting (SEEA-EEA). The SEEA-EEA guides the implementation of ecosystem accounting and describe the different types of accounts within the accounting system. To fully operationalise ecosystem accounting multiple challenges still need to be overcome. The main aim of this thesis is to empirically assess how spatial models for ecosystem service flows and biodiversity can be applied in the context of ecosystem accounting, conservation and management. For this thesis, the Dutch province of Limburg is used as a case study. In this thesis, I address three challenges concerning the development of ecosystem accounting and the conservation of biodiversity and ecosystem services.

First, the different types of accounts within the ecosystem accounting framework require empirical testing. Although extensive experience with spatial ecosystem services analysis exists, the applicability of ecosystem service mapping and modelling methods for ecosystem accounting has rarely been rigorously tested. In this thesis, I assess the ecosystem service flow account, which is a crucial component of an ecosystem accounting system. In Chapter 2, I develop spatial biophysical models of seven ecosystem services in Limburg, in a way that is consistent with ecosystem accounting. The included ecosystem services are hunting, drinking water extraction, crop production, fodder production, air quality regulation, carbon sequestration and recreational cycling. In addition, I examine how human inputs can be distinguished from ecosystem services. Model outcomes are used to develop an ecosystem accounting table in line with SEEA-EEA guidelines, in which contributions of land cover types to ecosystem service flows are recorded. Furthermore, spatial accounts for single statistical units are developed. Based on an analytical assessment, I argue that for many ecosystem services fully disentangling human and ecosystem contributions is not possible. The model results show that land cover types that have the largest contribution to the total annual flow of an ecosystem service do not necessarily have the highest mean annual flow per hectare. While the total annual ecosystem service flow is generally lowest in the more natural land cover types with a smaller extent, such as heath and peat, the mean ecosystem service flow per hectare from these land cover types is highest for multiple ecosystem services, including hunting, air quality regulation and recreational cycling. In Chapter 3, I assess a monetary ecosystem service flow account for Limburg. Ecosystem accounting monitors ecosystem services and measures their monetary value using exchange values consistent with the System of National Accounts (SNA). Seven ecosystem services are modelled and valued: crop production, fodder production, drinking water production, air quality regulation, carbon sequestration, nature tourism and hunting. Monetary ecosystem accounts are developed that specify values generated by ecosystem services per hectare, per municipality and per land cover type. I analyse the relative importance of public and private ecosystem services. The SNA-aligned monetary value of modelled ecosystem services for Limburg was around €112 million in 2010, with an average value of €508 per hectare. Ecosystem services with the highest values are crop production, nature tourism and fodder production. Due to exclusion of consumer surplus in SNA valuation, calculated values are considerably lower than those typically found in welfare-based valuation approaches. I show that it is feasible to develop spatial models for flows of many different ecosystem service types and value these ecosystem services with SNA-consistent methods. The main uncertainties that underlie spatial modelling and monetary valuation of ecosystem service flows are assessed. The main uncertainties in spatial modelling stem from lacking data availability and accessibility, lack of model validation and high levels of spatial aggregation. The main uncertainties in monetary valuation stem from a lack of local data and the methodological inability to fully disentangle ecosystem and human contributions. SNA-aligned methods to value some cultural and regulating ecosystem services still need to be further developed and tested in an ecosystem accounting context to ensure reliable applications.

Second, the role of biodiversity in ecosystem accounting requires further research. The multi-layered relationship between biodiversity and ecosystem services, makes measuring and monitoring of biodiversity necessary to acquire comprehensive ecosystem accounts. Biodiversity aspects, such as species diversity, are not captured by accounts focussing on ecosystem services. For the purpose of ecosystem accounting an open challenge is to assess which set of indicators can be included in a biodiversity account. In Chapter 4, I address indicators that reflect human appreciation of ecosystems for biodiversity accounting. I assess various indicators for species diversity for Limburg in terms of their applicability in the SEEA-EEA framework. In particular, I analyse indicators reflecting species richness, the presence of rare and threatened species and species abundance using six different criteria. These criteria are whether indicators are quantifiable, feasible to measure and monitor, comparable between regions, comprehensive, credible and, finally, understandable for a broader audience. I show that for Limburg province spatial variation between different species groups is large, which implies that in the development of biodiversity accounts multiple species groups should be considered. Species richness is useful as an indicator to identify areas of particular importance for biodiversity conservation, with Limburg province showing a strong spatial correlation between species richness for all species and species richness of threatened species. Rarity indicators and species abundance indicators showed weak spatial correlation with species richness, providing complementary information on species distribution. All indicators have different strengths and weaknesses, implying that in the development of biodiversity accounts multiple species groups and multiple indicators need to be combined. However, the specific combination that provides the most comprehensive information while restricting the amount of indicators is likely to differ between different areas.

Finally, the policy purposes of ecosystem accounting need to be further explored. Integrated information from ecosystem accounting on both ecosystem services and biodiversity could be used for decisions on conservation and land management. Moreover, the role of spatial ecosystem service analysis in conservation and spatial relations between ecosystem services and biodiversity require further analysis. Ecosystem services are increasingly considered alongside biodiversity in spatial conservation assessments. The general premise is that inclusion of ecosystem services in spatial conservation assessments will also benefit biodiversity and increase the cost-effectiveness of conservation. These ideas, however, are not conclusively supported by scientific evidence and require more research, addressing issues such as available cost data and target setting for conservation goals. In Chapter 5, I address this challenge by assessing how budget

limitations affect the expansion of a conservation network using a systematic conservation planning approach. This planning approach can integrate both biodiversity and ecosystem services as conservation targets, and hence address the challenge to operationalize ecosystem services as an anthropocentric argument for conservation. I create two conservation scenarios to expand the current conservation network in Limburg. One scenario focuses on biodiversity only and the other integrates biodiversity and ecosystem services. I vary conservation budgets in these scenarios and used the Marxan software to assess differences in the resulting network configurations. In addition, I test the network's cost-effectiveness by allocating a conservation budget either in one or in multiple steps. I include twenty-nine biodiversity aspects and five ecosystem services. The inclusion of ecosystem services to expand Limburg's conservation network only moderately changes prioritized areas, compared to only conserving biodiversity. Network expansion in a single time-step is more efficient in terms of compactness and cost-effectiveness than implementing it in multiple time-steps. Therefore, to cost-effectively plan conservation networks, the full budget should ideally be available before the plans are implemented. I show that including ecosystem services to cost-effectively expand conservation networks can simultaneously encourage biodiversity conservation and stimulate the protection of conservation-compatible ecosystem services.

Relevant data for ecosystem accounting exists on many ecosystem services, environmental, socio-economic variables and biodiversity but this data is very scattered, not always accessible and collected for other purposes. This hampers effective accounting. Therefore, in addition to further investing in the improvement of spatial modelling and valuation techniques, investments are needed in increasing data availability and accessibility for ecosystem accounting. Modern techniques, such as high resolution and up-to-date satellite imagery and big data opportunities through advanced and readily available technology (e.g. smartphones) increase the potential for data collection through citizen science. Such techniques increase the potential for data poor areas, but also increase the effectiveness of data in data rich areas such as the Netherlands.

Ecosystem services are increasingly being included in different types of ecosystem management and biodiversity conservation strategies. This is a positive development according to my results. An operational ecosystem accounting system insures that sufficient information is collected on ecosystem services and biodiversity to monitor not only the conditions, capacity, flow and demand, but also the effectiveness of management and conservation. This thesis contributes to operationalising ecosystem accounting by empirically testing applicable indicators, spatial modelling methods and monetary valuation methods, delineating caveats and assessing uncertainties. Although the studied ecosystem services can easily be expanded and more aspects can be covered, before ecosystem accounting is put into full practice, my study shows that ecosystem accounting withstands multiple empirical tests. Including ecosystem services in the design of a biodiversity conservation network can increase its cost-effectiveness and conserve more aspects of biodiversity and ecosystem services with societal and ecological importance. This thesis provides necessary insights and approaches to develop ecosystem accounting, management and conservation strategies. This helps to ensure the safeguarding of ecosystems for both their own good and those of society.

# Samenvatting

Ecosystemen en biodiversiteit spelen een belangrijke rol in het in stand houden van onze levenskwaliteit. Ecosystemen leveren diverse bijdragen aan ons welzijn en economische activiteiten. Deze bijdragen worden ecosysteemdiensten genoemd. Voorbeelden hiervan zijn natuurlijke zuivering van drinkwater, koolstofvastlegging en een aantrekkelijk landschap voor recreatie en toerisme. Biodiversiteit is de verscheidenheid van alle bronnen van levende organismen, met inbegrip van de diversiteit binnen soorten, tussen soorten en van ecosystemen. Ecosysteemdiensten en biodiversiteit moeten worden behouden om onze planeet goed te laten functioneren. Om dit behoud te monitoren neemt de vraag om ecosysteemdiensten en biodiversiteit systematisch te meten toe (bijvoorbeeld in de Aichi doelen van de Verenigde Naties en de Biodiversiteitsstrategie van de EU) en *ecosystem accounting* is een aanpak die op deze vraag ingaat.

Ecosystem accounting is een systematische aanpak voor het meten en monitoren van de staat van ecosystemen, ecosysteemdiensten en biodiversiteit. Het richt zich expliciet op een ruimtelijke aanpak om de verscheidenheid van ecosystemen en ecosysteemdiensten in kaart te brengen. De basis voor ecosystem accounting is in de afgelopen jaren gelegd. Er bestaat een duidelijke relatie met de nationale economische rekeningen, er is een gedefinieerd doel, er is een geaccepteerde terminologie en er is een analyse gedaan van de benodigde componenten en hun onderlinge relaties. Onder leiding van de Verenigde Naties zijnde eerste internationaal erkende richtlijnen ontwikkeld voor het opstellen van het rekeningstelsel voor ecosystem accounting, de 'System of Environmental-Economic Accounting - Experimental Ecosystem Accounting' (SEEA-EEA). Desondanks zijn er nog meerdere uitdagingen om ecosystem accounting volledig te operationaliseren. Het belangrijkste doel van dit proefschrift is om empirisch te beoordelen hoe ruimtelijke modellen voor ecosysteemdiensten en biodiversiteit kunnen worden toegepast in de context van ecosystem accounting, natuurbescherming en natuurbeheer. Voor dit proefschrift wordt de Nederlandse provincie Limburg gebruikt als studiegebied. Ik onderzoek drie uitdagingen met betrekking tot het operationaliseren van ecosystem accounting en het beheer van biodiversiteit en ecosysteemdiensten: (1) de empirische toetsing van de verschillende soorten rekeningen binnen ecosystem accounting, (2) de rol van biodiversiteit in ecosystem accounting, en (3) de beleidsrelevantie van ecosystem accounting.

Empirische toetsing van de verschillende soorten rekeningen binnen ecosystem accounting is nodig. Hoewel de kennis op het gebied van ruimtelijke analyse van ecosysteemdiensten in de afgelopen jaren snel is toegenomen, is de toepasbaarheid van de verschillende methoden om ecosysteemdiensten te modelleren voor ecosystem accounting nauwelijks onderzocht. In dit proefschrift beoordeel ik een cruciaal onderdeel van het ecosystem accounting systeem, namelijk de *ecosysteemdiensten flow rekening*. Dit is een rekening waarin wordt bijgehouden hoeveel ecosysteemdiensten op jaarbasis worden geleverd in een gebied. Om deze rekening op te kunnen zetten ontwikkel ik in hoofdstuk 2 biofysische modellen voor zeven ecosysteemdiensten in Limburg. De gemodelleerde ecosysteemdiensten zijn jacht, drinkwaterproductie, de productie van voedselgewassen, de productie van veevoedergewassen, regulering van luchtkwaliteit door vegetatie, koolstofvastlegging en fietsrecreatie. In de resulterende ecosysteemdiensten flow rekening worden de bijdragen van landgebruikstypes aan de levering van ecosysteemdiensten geregistreerd. Daarnaast worden ruimtelijke rekeningen voor kleinschalige statistische eenheden ontwikkeld. De modelresultaten laten zien dat de totale jaarlijkse ecosysteemdiensten *flows* over het algemeen het laagst zijn in de meer natuurlijke landgebruikstypes met een kleinere omvang, zoals heide en veen. Maar tegelijkertijd zijn de gemiddelde ecosysteemdienststromen per hectare uit deze landgebruikstypes het hoogst voor meerdere ecosysteemdiensten. In hoofdstuk 3 ontwikkel ik een monetaire ecosysteemdiensten flow rekening voor Limburg. Ecosystem accounting monitort ecosysteemdiensten en meet hun monetaire waarde met behulp van de *exchange value* in overeenstemming met het systeem voor nationale rekeningen (SNA) van de Verenigde Naties. Zeven ecosysteemdiensten worden gemodelleerd en gewaardeerd: de productie van voedselgewassen, de productie van veevoedergewassen, drinkwaterproductie, regulering van luchtkwaliteit door vegetatie, koolstofvastlegging, natuurtoerisme en jacht. Een monetaire ecosysteemdiensten flow rekening wordt ontwikkeld die ecosysteemdienstwaarden per hectare, per gemeente en per landgebruikstype in kaart brengt. De berekende monetaire waarde van gemodelleerd ecosysteemdiensten voor Limburg was ongeveer €112 miljoen in 2010, met een gemiddelde waarde van €508 per hectare. Ecosysteemdiensten met de hoogste waarden zijn productie van voedselgewassen, natuurtoerisme en de productie van veevoedergewassen. Het consumentensurplus wordt in SNA gerelateerde waardering niet meegenomen, waardoor berekende waarden aanzienlijk lager zijn dan waardes die berekend worden met op welzijn gebaseerde waarderingsmethoden. Ik toon aan dat het haalbaar is om ruimtelijke modellen te ontwikkelen voor veel verschillende soorten ecosysteemdiensten en dat ze gewaardeerd kunnen worden met SNA-consistente methoden. De belangrijkste onzekerheden die ten grondslag liggen aan ruimtelijke modellering en monetaire waardering van ecosysteemdiensten flows worden beoordeeld. De belangrijkste onzekerheden in ruimtelijke modellen vloeien voort uit gebrek aan beschikbaarheid en toegankelijkheid van data, het ontbreken van modelvalidatie en de hoge niveaus waarop data geaggregeerd wordt. De belangrijkste onzekerheden in monetaire waardering vloeien voort uit een gebrek aan lokale gegevens en de methodologische tekortkomingen om bijdragen van ecosystemen en mensen volledig te onderscheiden. SNA-consistente methoden om een aantal culturele en regulerende ecosysteemdiensten te waarderen moeten nog verder worden ontwikkeld en getest in een ecosystem accounting context om een betrouwbare waardering te bewerkstelligen.

De rol van biodiversiteit in ecosystem accounting vergt verder onderzoek. De gelaagde relatie tussen biodiversiteit en ecosysteemdiensten zorgt ervoor dat het meten en monitoren van de biodiversiteit nodig is om een uitgebreide ecosystem accounting rekeningstelsel te ontwikkelen. Biodiversiteitsaspecten, zoals soortendiversiteit, worden niet meegenomen in rekeningen die zich op ecosysteemdiensten richten. Een uitdaging voor ecosystem accounting is om te beoordelen welke biodiversiteitsindicatoren kunnen worden opgenomen in een speciale biodiversiteitsrekening binnen het rekeningstelsel. In hoofdstuk 4, onderzoek ik biodiversiteitsindicatoren die de menselijke waardering voor ecosystemen weerspiegelen, en hun geschiktheid om in een biodiversiteitsrekening opgenomen te worden. Ik beoordeel verschillende indicatoren voor soortendiversiteit in Limburg op hun toepasbaarheid in het kader van de SEEA-EEA. Ik beoordeel indicatoren die soortenrijkdom, de aanwezigheid van zeldzame en bedreigde soorten en soortenabundantie weerspiegelen op basis van zes verschillende criteria. Deze criteria zijn (1) of indicatoren kwantificeerbaar zijn, (2) haalbaar om te meten en te monitoren, (3) vergelijkbaar tussen regio's, (4) veelomvattend, (5) geloofwaardig en, ten slotte, (6) begrijpelijk zijn voor een breder publiek. Ik laat zien dat voor Limburg de ruimtelijke variatie tussen verschillende soortengroepen groot is, met als gevolg dat in de ontwikkeling van de biodiversiteitrekening meerdere soortengroepen moet worden meegenomen om deze ruimtelijke variatie te weerspiegelen. Soortenrijkdom is een nuttige indicator om gebieden te identificeren die van bijzonder belang zijn voor het behoud van biodiversiteit, en heeft een sterke ruimtelijke correlatie met soortenrijkdom van bedreigde soorten in Limburg. Zeldzaamheidsindicatoren en soortenabundantie indicatoren hebben een zwakke ruimtelijke correlatie met soortenrijkdom. Die typen indicatoren verstrekken daardoor aanvullende informatie over de verspreiding van soorten. Alle indicatoren hebben verschillende sterke en zwakke punten, waardoor bij de ontwikkeling van de biodiversiteitsrekening meerdere soortengroepen en indicatoren gecombineerd zullen moeten worden. De specifieke maar tegelijkertijd beperkte combinatie van indicatoren die de meest uitgebreide informatie verschaft over de lokale biodiversiteit, zal waarschijnlijk verschillen tussen verschillende gebieden.

De beleidsdoelstellingen van ecosystem accounting zullen verder moeten worden onderzocht. Daarbij moet gekeken naar hoe het kan worden gebruikt voor de beslissingen over natuurbescherming en landschapsbeheer. Bovendien is er meer onderzoek nodig naar de ruimtelijke relaties tussen ecosysteemdiensten en biodiversiteit, en de invloed hiervan op beleid. Ecosysteemdiensten worden in toenemende mate in combinatie met biodiversiteit meegenomen in de ruimtelijke natuurbeschermingsevaluaties. Het algemene uitgangspunt is dat het opnemen van ecosysteemdiensten in ruimtelijke natuurbeschermingsevaluaties ook voordelig zal zijn voor biodiversiteit en de kosteneffectiviteit van natuurbescherming zal verhogen. Er bestaat echter te weinig ondersteunend wetenschappelijk bewijs voor deze ideeën. In hoofdstuk 5, pak ik deze uitdaging aan door te analyseren hoe budgettaire beperkingen van invloed zijn op de verdere ontwikkeling van het natuurnetwerk in Limburg, met behulp van *systematic conservation planning*. Deze planningsbenadering integreert zowel biodiversiteit en ecosysteemdiensten als natuurbeschermingsdoelen. Het haakt daarmee in op de uitdaging om ecosysteemdiensten te operationaliseren als een antropocentrisch argument voor natuurbescherming. Ik ontwikkel twee scenario's om de huidige natuurnetwerk in Limburg uit te breiden. Eén scenario richt zich alleen op biodiversiteit en de andere integreert biodiversiteit en ecosysteemdiensten. In deze scenario's varieer ik de natuurbeschermingsbudgetten en analyseer ik verschillen in netwerkconfiguratie met het softwarepakket Marxan. Daarnaast test ik de kosteneffectiviteit van het netwerk door de toekenning van het budget te spreiden over één of meerdere tijdstappen. In dit onderzoek neem ik negenentwintig biodiversiteitsaspecten en vijf ecosysteemdiensten mee. Ecosysteemdiensten in combinatie met biodiversiteitsaspecten meenemen in het natuurnetwerk van Limburg veroorzaakt slechts kleine veranderingen in de gebieden die gekozen worden voor uitbreiding, in vergelijking met een focus op biodiversiteit alleen. Uitbreiding in een enkele tijdstap is efficiënter voor de compactheid en de kosteneffectiviteit van het netwerk dan uitbrieding in meerdere tijdstappen. Om de kosteneffectief te natuurnetwerken te plannen, zou het volledige budget daarom idealiter beschikbaar moeten zijn voordat de plannen worden uitgevoerd. Ik toon aan dat ecosysteemdiensten meenemen als beleidsdoelen om op een kosteneffectieve manier het natuurnetwerk uit te breiden, zowel biodiversiteitsbescherming kan bevorderen en de bescherming van bepaalde ecosysteemdiensten kan stimuleren.

Er bestaan veel relevante gegevens voor ecosystem accounting op het gebied van ecosysteemdiensten, milieuaspecten, sociaaleconomische variabelen en biodiversiteit, maar deze gegevens zijn zeer verspreid, niet altijd toegankelijk en vaak verzameld voor andere doeleinden, waardoor ze niet direct toepasbaar zijn. Dit belemmert een effectieve ecosystem accounting. Om deze redenen zijn er, in aanvulling op verdere investeringen in de verbetering van de ruimtelijke modellerings- en waarderingstechnieken, investeringen nodig in het verbeteren van de beschikbaarheid en de toegankelijkheid van gegevens voor ecosystem accounting. Moderne technieken, zoals hoge resolutie, up-to-date satellietbeelden en big data mogelijkheden vergroten de mogelijkheden voor het verzamelen van toekomstige data. Dergelijke technieken verhogen de mogelijkheden voor ecosystem accounting in data-arme gebieden, maar ook de doeltreffendheid van data voor accounting in datarijke gebieden, zoals Nederland.

Ecosysteemdiensten worden steeds vaker opgenomen in verschillende soorten biodiversiteits- en natuurbeschermingsstrategieën. Mijn resultaten wijzen uit dat dit een positieve ontwikkeling is. Een operationeel ecosystem accounting systeem zorgt ervoor dat er veel informatie wordt verzameld over ecosysteemdiensten en biodiversiteit, om ze zorgvuldig te monitoren.

Dit proefschrift draagt bij aan het operationaliseren van ecosystem accounting door empirisch toepasselijke indicatoren, ruimtelijke modelleringsmethoden en monetaire waarderingsmethoden te testen, waarbij onzekerheden en ontbrekende kennis worden beoordeeld. Hoewel de set van onderzochte ecosysteemdiensten kan worden uitgebreid en meer aspecten binnen het rekeningstelsel kunnen worden bestudeerd, voordat ecosystem accounting volledig in praktijk wordt gebracht, toont mijn studie aan dat ecosystem accounting meerdere empirische tests heeft weerstaan. Dit proefschrift biedt inzichten en benaderingen die noodzakelijk zijn voor het verder ontwikkelen van ecosystem accounting, en ruimtelijke bescherming- en beheerstrategieën voor ecosysteemdiensten en biodiversiteit. De inzichten zijn daarom zeer belangrijk voor de instandhouding van natuurlijke en gemanagede ecosystemen, en van menselijk welzijn.

## About the author

Roy Remme was born on 28 December 1985 in the same place that he would obtain his PhD 30 years later: Wageningen, the Netherlands. At the age of two Roy was taken on a ten year adventure by his parents through Somalia, Zimbabwe and Ecuador. Together with his younger sister and brother he had the privilege to discover extraordinary parts of the world, full of exotic landscapes, spectacular wildlife, but also with large inequality gaps between rich and poor and tangible environmental issues, such as water scarcity, poaching and smog. The frequent visits to natural parks, alongside the visibly fragile human-nature relationship were important factors for determining Roy's later academic choices.

At the age of twelve Roy returned to the Netherlands with his parents, sister and brother. Roy attended high school in Arnhem, after which he chose to follow the interdisciplinary BSc programme Environmental Sciences at Utrecht University. After four years in Utrecht, he returned to Wageningen to follow the MSc programme Climate Studies at Wageningen University. While finishing his MSc thesis in 2011, he was made aware of the opportunity to start a PhD on ecosystem service modelling and valuation with the Ecospace project at the Environmental Systems Analysis (ESA) group. Always having been interested in a systems approach and interdisciplinary research, he applied and was hired to do research on the Dutch case study, which resulted in this PhD thesis. Besides the thesis research, Roy acquired teaching and supervision experience by, among others, supervising MSc students during their thesis research and teaching the MSc consultancy training course European Workshop. In a project with Statistics Netherlands, Roy further developed ecosystem accounts for Limburg.

Roy now lives in Utrecht and works partly as a postdoctoral researcher on ecosystem service at the ESA group, and partly as a scientific advisor for the Atlas Natural Capital at the Dutch National Institute for Public Health and the Environment (RIVM).

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## SENSE PhD Courses

- Environmental Research in Context (2011)
- Integrated assessment of ecosystem services: from theory to practice, Wageningen University and VU University Amsterdam (2011)
- Research in Context Activity: 'Co-organising SENSE Research Cluster meeting on 'Ecosystem services as a contested concept' and preparing publication with participating PhD candidates' (2012)
- Introduction to R for statistical analysis (2015)

## Other PhD and Advanced MSc Courses

- Techniques for writing and presenting a scientific paper, Wageningen University (2011)
- Cost-benefit analysis and environmental valuation, Wageningen University (2011)
- Geostatistics, Wageningen University (2011)
- Summer School 'Biodiversity and ecosystem services', ALTER-Net: A Long-Term Biodiversity, Ecosystem and Awareness Research Network (2011)

## Management and Didactic Skills Training

- Supervising three MSc students at Wageningen University with thesis entitled 'Spatial models of biodiversity: testing GLOBIO for the province of Limburg, the Netherlands' (2014), 'Assessing ecosystem services to develop 'De Maashorst' nature reserve' (2014), and 'Development of management scenarios for Atewa Range Forest Reserve in Ghana using Integrated Ecosystem Assessment' (2014)
- Co-organising and hosting session 'Approaches to modelling ecosystem services, where are we now and new approaches', Conference Ecosystem Services Partnership (ESP), San José, Costa Rica (2014)

## Oral Presentations

- *Spatial accounting for ecosystem services in Limburg.* Global Land Project (GLP) Open Science Meeting, 19-21 March 2014, Limburg, The Netherlands
- *Spatial correlation between ecosystem services and biodiversity in Limburg, the Netherlands.* Ecosystem Services Partnership (ESP), 8-12 September 2014, San José, Costa Rica
- *Ecosystem accounting developing a spatial approach to account for biophysical and monetary flows of ecosystem services.* Analysis and Experimentation on Ecosystems (AnaEE) Workshop, 14-15 October 2014, Antwerp, Belgium

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