Ecosystem Services 10 (2014) 6–18

Contents lists available at ScienceDirect

Ecosystem Services

journal homepage: www.elsevier.com/locate/ecoser

Developing spatial biophysical accounting for multiple ecosystem services

Roy P. Remme *, Matthias Schröter, Lars Hein

Environmental Systems Analysis Group, Wageningen University, PO Box 47, 6700 AA Wageningen, The Netherlands

ARTICLE INFO

Article history:
Received 11 November 2013
Received in revised form
26 June 2014
Accepted 23 July 2014
Available online 12 September 2014

Keywords:
Cultural landscape
Province
Mapping
Biophysical accounting
SEEA
Ecosystem contribution

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.

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) and there is a high demand from policy makers for sound information on ecosystem services (ES) (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 (United Nations 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) (European Commission et al., 2013).

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 ES can be connected to economic activity, and how flows of ES can be quantified at large spatial scales, with an accuracy sufficient for accounting purposes (Boyd and Banzhaf, 2007; Edens and Hein, 2013; Mäler et al., 2008). Ecosystem accounting takes a spatial approach towards analysing ecosystems and ES. The SEEA EEA guidelines recognise that ecosystems and ES are spatially heterogeneous, and that this spatial variability needs to be captured in ecosystem accounting (European Commission et al., 2013). 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. Burkhard et al., 2012; Maes et al., 2012; Nelson et al., 2009; Raudsepp-Hearne et al., 2010; Schröter et al., 2014a; Serna-Chavez et al., 2014; Willemen et al., 2010). It addresses a wide range of ES at different spatial scales with a

* Corresponding author. Tel.: +31 0 317 48 11 27; fax: +31 0 317 41 90 00.
E-mail addresses: roy.remme@wur.nl, royremme@gmail.com (R.P. Remme), matthias.schroter@wur.nl (M. Schröter), lars.hein@wur.nl (L. Hein).

http://dx.doi.org/10.1016/j.ecoser.2014.07.006
2212-0416/

variety of services modelled with different spatial methods (Crossman et al., 2013b; Martínez-Harms and Balvanera, 2012; 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 (Burkhard et al., 2012; Seppelt et al., 2011) are less suitable for ecosystem accounting. Spatial modelling of ES for ecosystem accounting calls for a definition of ES that is aligned with the national accounting framework (European Commission et al., 2013), 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 ES 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 ES 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 ES typical for North Western Europe. We analysed seven ES: hunting, drinking water extraction, crop production, fodder production, air quality regulation, forest carbon sequestration and recreational cycling.

2. Conceptual framework and definition of ES

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 ES as a "cascade" from ecosystem properties to ES values. In accounting, ES are "the contributions of ecosystems to benefits used in economic and other human activity" (European Commission et al., 2013). 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 (Bateman et al., 2011; Boyd and Banzhaf, 2007; Haines-Young and Potschin, 2010b; TEEB, 2010), and that the processed goods (e.g. milk, processed wood or bread) themselves are not the ES (Schröter et al., 2014b).

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 (i) historic and current management of the ecosystem state and (ii) the extraction or use of the ES (Fig. 1). The magnitude of these human contributions 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 past conversion of a natural ecosystem to cropland. Within an accounting context, past anthropogenic changes to ecosystem properties are reflected in the current state of the ecosystem. Recurrent inputs may be required for generating ES (as in the case of fertilizer inputs required for crop production), and they need to be measured and included in the account as intermediate human input.

For humans to benefit from ES a flow is necessary from the ecosystem to society. For most regulating ES 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 ES, 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 (Fig. 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"

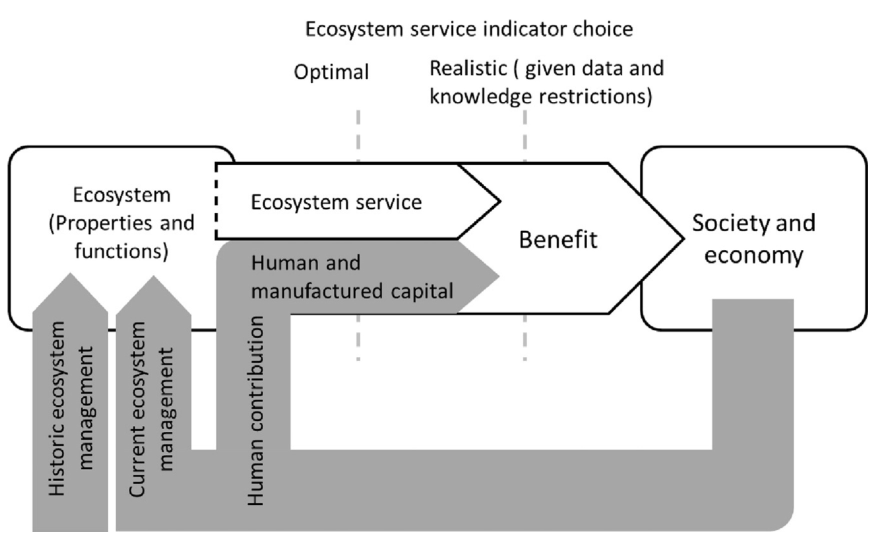

Fig. 1. Framework for conceptualization of human contributions to the emergence of an ecosystem service. Both historic and current management influence ecosystem properties and functions, which in turn has an impact on the ecosystem service. Human and manufactured capital is often needed to realise the 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 1
The modelled ES, 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

through investments, cf. Spangenberg et al. (2014)). In the case of cultural ES a human contribution in the form of an activity is needed. 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 1 we conceptually explain the differences between the ES and the benefit for each ES that was modelled in this study. The notion of ES 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 ES are the result of combined influences of natural processes and human management, contributions of ecosystems are difficult to separate in a meaningful way, given current data and knowledge restrictions.

When measuring ES 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 ES indicates a benefit, which is why a "realistic" choice of indicators (Fig. 1) often does not allow for disentanglement. For more intangible ES 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 1) (cf. Edens and Hein, 2013; Schröter et al., 2012). However, in many cases this was not possible because data for ES indicators were not available. For those ES an indicator which represents the benefit was chosen, as explained in Section 3.3.

3. Methods

3.1. Study area

Limburg province is situated in the south-eastern part of the Netherlands, covering approximately 2200 km^2 (Fig. 2). The province has a varied and fragmented cultural landscape, which has been managed for many centuries (Berendsen, 2005; Jongmans et al., 2013). The area is densely populated (522 people km^{-2} in 2012) (Statistics Netherlands, 2012) and competition for land for agricultural, nature and urban purposes is high (Vogelzang et al., 2010). 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). The province is nationally renowned for the attractive hilly landscape in the southern part of the province.

3.2. Ecosystem accounting units

We used three types of spatial accounting units that are aligned with those proposed in the SEEA EEA (European Commission et al., 2013). The largest unit was the ecosystem accounting unit (EAU), which was delineated by the administrative boundaries of Limburg province. The second unit type was the land cover/ecosystem functional unit (LCEU). Eight types of LCEUs, aligned with the main land cover class types, were distinguished for the analysis of ES flows (Fig. 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 (25×25 m grain) that together make up a LCEU. A BSU is used to assess local variation in ES flows.

3.3. Modelled ES

In this study seven ES have been modelled, chosen to reflect the diversity of services in 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 ES 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). The ES were chosen based on expert judgement and feedback from provincial policy makers, in combination with the criterion of data availability. The ES list is not exhaustive but it does cover key economic sectors (agricultural services), cultural aspects (cycling as a main form of recreation,

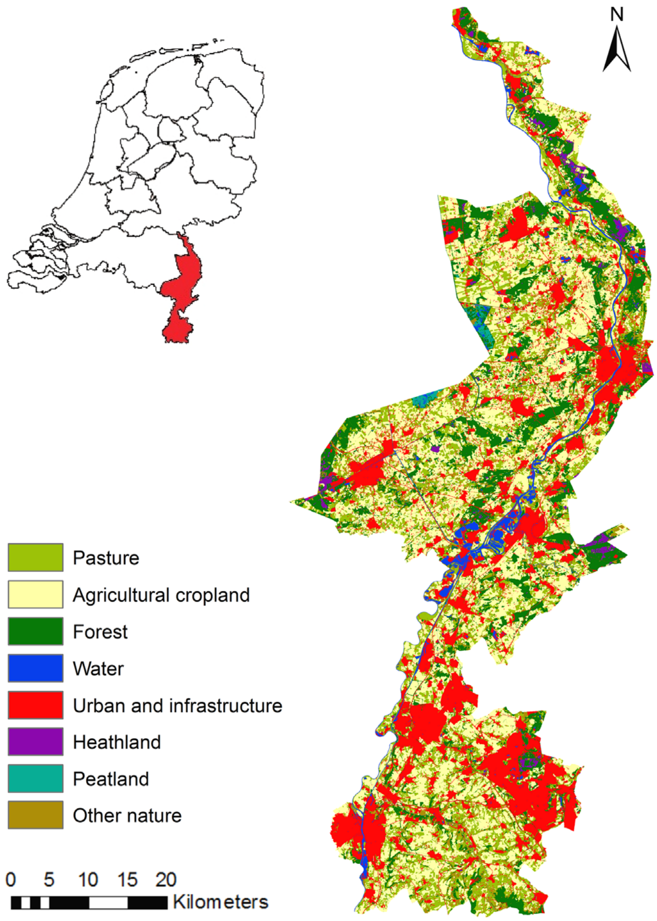

Fig. 2. Location and land cover of Limburg province. Source land cover data: Hazeu (2009).

and to a lesser extent hunting) and human health aspects (air quality regulation and clean drinking water) as well as an ES of international interest (carbon sequestration).

A specific spatial model was developed 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. The models are described below.

3.3.1. 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 the 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.

3.3.2. 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 2386 ha in size. These groundwater 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 groundwater 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.

3.3.3. 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, groundwater 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 (kg ha^{-1} yr^{-1}) as an indicator for the ES, noting that this does not accurately reflect the ecosystem contribution. We express all crops in terms of weight at harvest, but realise that the value of the crops, both on a per kg and on a per ha basis, varies considerably between crops. These differences in values can be made apparent by the monetary component of the ecosystem account (European Commission et al., 2013).

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).

Table 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)

3.3.4. Fodder production

In Limburg cattle rearing for dairy and meat is an important economic activity (Statistics Netherlands, 2013b). 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. Maes et al., 2011; Naidoo et al., 2008; 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 considering 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, 2013a)] 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 intensity 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 based on the average production intensity in the municipality.

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:

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

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 categorised 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.

3.3.5. 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):

$$PM_{10}\ \text{capture} = AV_d tC \tag{2}$$

where A is area, V_d is vertical deposition velocity for specific land covers, t is the time step (one year), and C is the 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 ($\mu g/m^3$) 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.

3.3.6. Carbon sequestration

Carbon sequestration (tC ha^{-1} yr^{-1}) 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. Carbon sequestration rates in different land cover types are based on the literature, as explained in Table 3.

3.3.7. 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 (NBTC-NIPO Research, 2012a, 2012b; Stichting Landelijk Fietsplatform, 2009, 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 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.

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 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.

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.

3.4. Accounting for ES

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 ES at high resolution. For each example BSU a separate account was created, in which land cover was determined and quantities of the seven modelled ES were calculated. Furthermore, at a provincial level the model outcomes were used to account for the quantities of ES 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.

4. Results

4.1. Spatial ES models

Fig. 3 shows the spatial distribution of the annual flows of the modelled ES in Limburg province. The spatial models show substantial

Table 3
Look-up table for carbon sequestration in Limburg.

Land cover category	Carbon sequestration (tC ha^{-1} yr^{-1})	References	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

Table 4
Percentages of cyclists taking recreational cycling trips of different lengths (NBTC-NIPO Research, 2012a; Stichting Landelijk Fietsplatform, 2013).

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

Table 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

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 (Fig. 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 (Fig. 3b). Second, three ES were derived from land cover types using look-up table approaches: crop production, fodder production and carbon sequestration (Fig. 3c, d and f). The third model type couples environmental conditions to land cover types. This model was used for quantifying air quality regulation (Fig. 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 (Fig. 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 the highest in the more densely populated southern part of the province. The highest values for cycling recreation are found in non-urban land covers directly adjacent to large cities.

4.2. Ecosystem accounting at the level of BSUs

Nine adjacent BSUs with a variety of land covers were selected as examples for setting up detailed spatial ecosystem accounts (Fig. 4). The separate accounts for each BSU are shown in Table 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. Table 6 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.

4.3. Ecosystem accounting at the level of LCEUs

LCEUs that have the largest contribution to the total annual flow of an ES do not necessarily have the highest mean annual flow (Table 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 several ES. For instance, *heath* has the highest mean annual flow for hunting and air quality regulation and *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). SDs were relatively high for most modelled ES. The presented SD reflects the spatial variation of BSUs, and the SD is low for ES that use aggregated statistics as input data.

5. Discussion

5.1. ES in cultural landscapes

The definition of ES as stated in the SEEA EEA makes a clear distinction between ES and benefits, recognising that, apart from ecosystem contributions, human contributions are often involved in deriving benefits from ecosystems (European Commission et al., 2013). We argue that in strongly modified cultural landscapes such as Limburg and many landscapes of Europe it is challenging 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, ES 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. Note that, in less intensively managed systems the ecosystem contribution and human contribution may be more straightforward to disentangle. For example, in Telemark county, Norway, sheep are released to graze in natural areas (Schröter et al., 2014a). This system requires little human involvement, and therefore for fodder production there are very few processes that need to be disentangled.

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

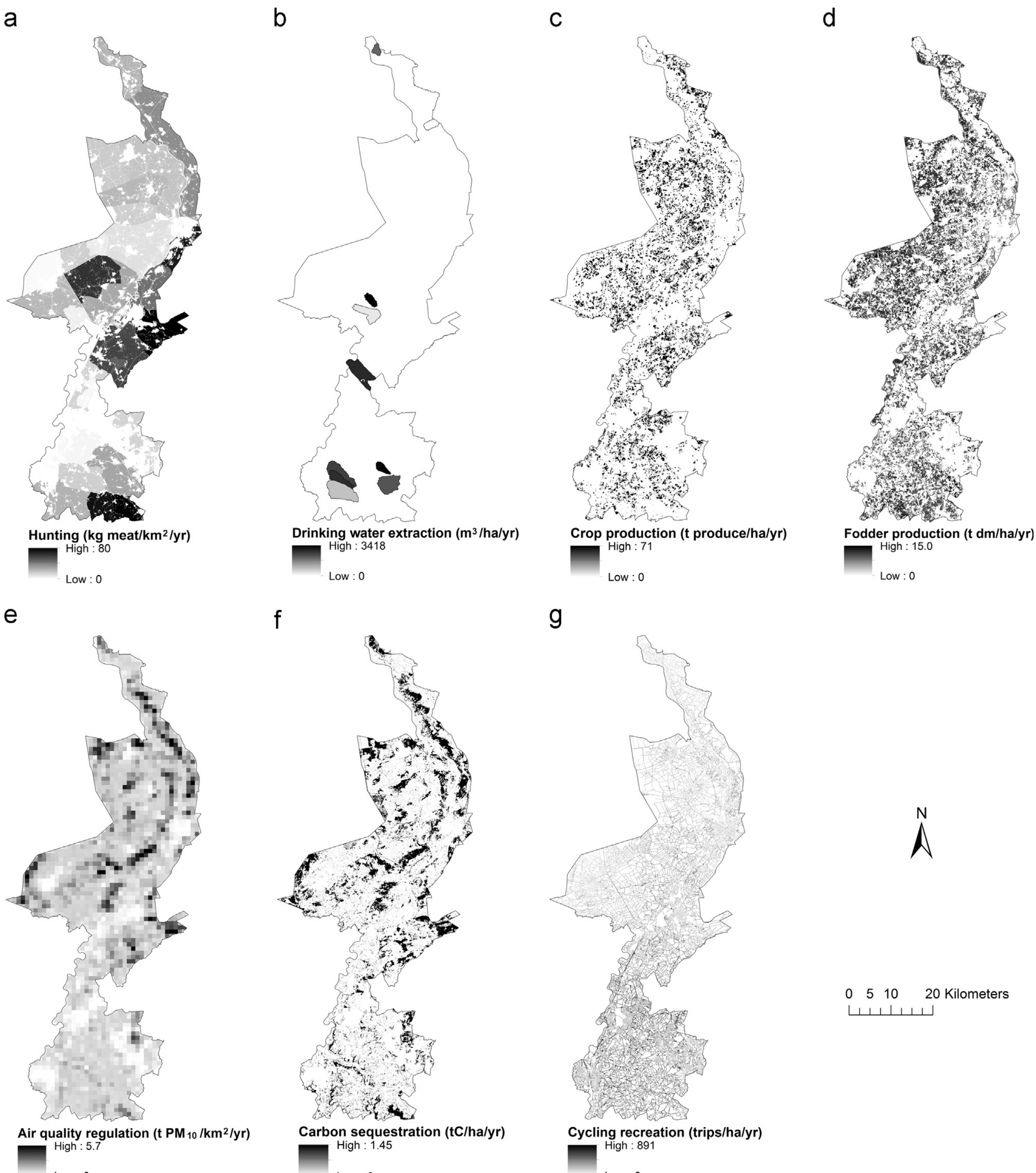

Fig. 3. Model output of ecosystem service flows for 2010 in Limburg. For data sources see Table 2.

ecosystem processes do not contribute significantly anymore. The interpretation of such an "ecosystem accounting boundary" results in a measurable indicator, and is internally consistent with the way other provisioning services are included in ecosystem accounting.

However, defining the last point where ecological processes play a significant role is not always easy, and caution is needed. For example, 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. Hunting is another example where the boundary is vague. 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 – even though populations of all hunted animals are managed by people. These examples show that the last significant ecological contribution can be subject to debate. ES and their indicators need to be well defined if they are to be incorporated in ecosystem accounting.

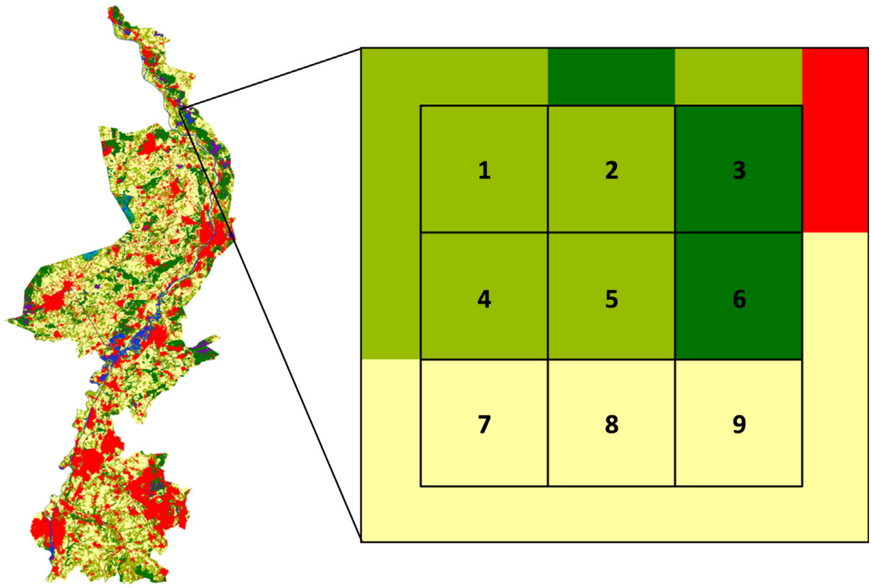

Fig. 4. The nine selected BSUs in northern Limburg. The numbers 1 through 9 correspond with the BSU numbers in Table 5.

Table 6
An ecosystem accounting table for the nine example BSUs (25 m grain) presented in Fig. 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	m^3/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	–	Grassland	Grassland	Forest	Grassland	Grassland	Forest	Cropland	Cropland	Cropland

5.2. Challenges of and uncertainties in spatial ES modelling for accounting

The specific requirements we outlined for spatial ES models in the context of ecosystem accounting included a specific definition of ES 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 ES 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 to verify our results with. 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., 2014; 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 (Crossman et al., 2013b; Martínez-Harms and Balvanera, 2012; Nemec and Raudsepp-Hearne, 2013). These issues also caused uncertainties in our spatial models, which we will discuss below.

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 ES 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.

Table 7
Total and mean annual ES flow per LCEU. Standard deviations (SD) are indicated for mean values over all BSUs. Missing values indicate that ES was not modelled in that LCEU.

LCEU	Ecosystem service													
	Hunting		Drinking water extraction		Crop production		Fodder production		Air quality regulation		Carbon sequestration		Recreational cycling	
	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})	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	9100	21 (*17*)	7026	2389 (*694*)	–	–	533	12,019 (*1571*)	404	909 (*528*)	8019	0.18 (*0*)	1863	103 (*77*)
Cropland	14,732	20 (*17*)	11,227	2329 (*744*)	1868	41,804 (*24,339*)	251	10,894 (*0*)	717	956 (*535*)	273	0.00 (*0.03*)	2611	98 (*72*)
Forest	8100	23 (*20*)	3117	2119 (*811*)	–	–	–	–	700	2001 (*1228*)	50,664	1.45 (*0*)	1565	128 (*95*)
Water	–	–	478	1369 (*621*)	–	–	–	–	40	613 (*558*)	0	0.00 (*0*)	139	109 (*91*)
Urban	–	–	4071	2298 (*788*)	–	–	–	–	272	535 (*546*)	875	0.02 (*0.05*)	2690	70 (*56*)
Heath	678	32 (*25*)	214	1262 (*614*)	–	–	–	–	45	2056 (*1116*)	393	0.18 (*0.03*)	30	84 (*61*)
Peat	70	13 (*3*)	–	–	–	–	–	–	7	968 (*347*)	149	0.18 (*0*)	3	84 (*43*)
Other nature	1513	25 (*20*)	862	2248 (*750*)	–	–	–	–	69	1153 (*705*)	1056	0.20 (*0*)	220	127 (*95*)
Provincial total	34,193		26,995		1868		784		2254		61,429		9122	

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 since we judged this study most consistent with our approach, i.e. analysis at a large spatial scale with temporal steps of one year.

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 (Daniel et al. 2012).

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” (European Commission et al., 2013). 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 (European Commission et al., 2013). 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 (e.g. land cover), 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. The BSU level can provide important accounting information that is relevant for assessing effects of local or sectoral policies, and monitoring trade-offs between ES and ES bundles.

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 flows of ES 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 indicate unsustainable ecosystem use (Schröter et al., 2014a). In the context of ES research closer collaboration among scientists from different disciplines and decision makers is needed (Crossman et al., 2013a). Developing ecosystem accounts requires collaboration between statisticians, policy-makers, land managers, economists and ecologists, as well as the 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., 2014a) 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.

6. Conclusion

This study has shown that spatial modelling of selected ES for ecosystem accounting in line with SEEA EEA is feasible for the data-rich case of Limburg province, the Netherlands. We outlined specific requirements for spatial modelling for the purpose of ecosystem accounting, namely a clear definition of ES, 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 operationalised. 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 ES that have been rarely modelled, in particular cultural services.

Acknowledgements

We would like to thank Bram Edens and Carl Obst and two anonymous reviewers for their constructive comments on the manuscript. We are grateful to Confidence Duku for assistance with the development of the recreational cycling model. This research has been made possible through ERC grant 263027 (Ecospace).

References

Aarts, H.F.M., Daatselaar, C.H.G., Holshof, G., 2005. Bemesting en opbrengst van productiegrasland in Nederland [Fertilization and yield of production grassland in the Netherlands]. Plant Research International, Wageningen, the Netherlands.

Alterra, 2006a. Grondsoortenkaart van Nederland 2006 [Soil map of the Netherlands 2006], Wageningen, the Netherlands.

Alterra, 2006b. Grondwatertrappen 2006 [Groundwater table 2006], Wageningen, the Netherlands.

Bagstad, K.J., Johnson, G.W., Voigt, B., Villa, F., 2013. Spatial dynamics of ecosystem service flows: A comprehensive approach to quantifying actual services. Ecosyst. Serv. 4, 117–125 http://dx.doi.org/10.1016/j.ecoser.2012.07.012.

Barbier, E.B., 2007. Valuing ecosystem services as productive inputs. Econ. Policy 22, 177–229.

Bateman, I., Mace, G., Fezzi, C., Atkinson, G., Turner, K., 2011. Economic analysis for ecosystem service assessments. Environ. Resour. Econ. 48, 177–218 http://dx.doi.org/10.1007/s10640-010-9418-x.

Beckett, K.P., Freer-Smith, P.H., Taylor, G., 2000. Effective tree species for local air quality management. J. Arboricult. 26, 12–19.

Berendsen, H.J.A., 2005. Landschappelijk Nederland, third ed. Van Gorcum, Assen, the Netherlands.

Böhnke-Henrichs, A., Baulcomb, C., Koss, R., Hussain, S.S., de Groot, R.S., 2013. Typology and indicators of ecosystem services for marine spatial planning and management. J. Environ. Manag. 130, 135–145.

Boyd, J., 2007. Nonmarket benefits of nature: What should be counted in green GDP? Ecol. Econ. 61, 716–723 http://dx.doi.org/10.1016/j.ecolecon.2006.06.016.

Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. Ecol. Econ. 63, 616–626 http://dx.doi.org/10.1016/j.ecolecon.2007.01.002.

Burkhard, B., Kroll, F., Nedkov, S., Müller, F., 2012. Mapping ecosystem service supply, demand and budgets. Ecol. Indic. 21, 17–29.

Chen, H., Wood, M.D., Linstead, C., Maltby, E., 2011. Uncertainty analysis in a GIS-based multi-criteria analysis tool for river catchment management. Environ. Model. Softw. 26, 395–405 http://dx.doi.org/10.1016/j.envsoft.2010.09.005.

Coenen, P.W.H.G., van der Maas, C.W.M., Zijlema, P.J., Baas, K., van den Berghe, A.C. W.M., te Biesebeek, J.D., Brandt, A.T., Geilenkirchen, G., van der Hoek, K.W., te Molder, R., Dröge, R., Montfoort, J.A., Peek, C.J., Vonk, J., van den Wyngaert, I., 2012. Greenhouse Gase Emission in the Netherlands 1990–2010, National Inventory Report 2012, RIVM, Bilthoven, the Netherlands.

Crossman, N.D., Bryan, B.A., de Groot, R.S., Lin, Y.-P., Minang, P.A, 2013a. Land science contributions to ecosystem services. Curr. Opin. Environ. Sustain. 5, 509–514 http://dx.doi.org/10.1016/j.cosust.2013.06.003.

Crossman, N.D., Burkhard, B., Nedkov, S., Willemen, L., Petz, K., Palomo, I., Drakou, E.G., Martín-Lopez, B., McPhearson, T., Boyanova, K., Alkemade, R., Egoh, B., Dunbar, M.B., Maes, J., 2013b. A blueprint for mapping and modelling ecosystem services. Ecosyst. Serv. 4, 4–14 http://dx.doi.org/10.1016/j.ecoser.2013.02.001.

Daniel, T.C., Muhar, A., Arnberger, A., Aznar, O., Boyd, J.W., Chan, K.M.A., Costanza, R., Elmqvist, T., Flint, C.G., Gobster, P.H., Grêt-Regamey, A., Lave, R., Muhar, S., Penker, M., Ribe, R.G., Schauppenlehner, T., Sikor, T., Soloviy, I., Spierenburg, M., Taczanowska, K., Tam, J., von der Dunk, A., 2012. Contributions of cultural services to the ecosystem services agenda. Proc. Nat. Acad. Sci. 109, 8812–8819. http://dx.doi.org/10.1073/pnas.1114773109.

Edens, B., Hein, L., 2013. Towards a consistent approach for ecosystem accounting. Ecol. Econ. 90, 41–52 http://dx.doi.org/10.1016/j.ecolecon.2013.03.003.

Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Heinemeyer, A., Gillings, S., Roy, D.B., Thomas, C.D., Gaston, K.J., 2010. The impact of proxy-based methods on mapping the distribution of ecosystem services. J. Appl. Ecol. 47, 377–385 http://dx.doi.org/10.1111/j.1365-2664.2010.01777.x.

Elowson, S., 1999. Willow as a vegetation filter for cleaning of polluted drainage water from agricultural land. Biomass Bioenergy 16, 281–290 http://dx.doi.org/10.1016/S0961-9534(98)00087-7.

European Commission, Organisation for Economic Co-operation and Development, United Nations, World Bank, 2013. System of Environmental-Economic Accounting 2012, Experimental Ecosystem Accounting.

Faunabeheereenheid Groningen, 2012. Rapportage reeënmachtiging 2010–2011 met vaststelling afschot 2012 [Report of roe deer hunting permits 2010–2011 and determination of hunting permits 2012] ⟨http://www.faunabeheereenheid.nl/groningen/beheer_%26_schadebestrijding/rapportages/rapportage%20reeenmachtiging%202010-2011%20met%20vaststelling%20afschot%202012%20gr%20def.doc/⟩.

Faunabeheereenheid Limburg, 2010. Wildbeheereenheden 2010 [Hunting districts 2010].

Faunabeheereenheid Limburg, 2011. Wild boar hunting in Limburg 2009–2011, Roermond, the Netherlands.

Faunabeheereenheid Limburg, 2012a. Afschot reewild 2010–2012 [Hunted roe deer 2010–2012], Roermond, the Netherlands.

Faunabeheereenheid Limburg, 2012b. Jaarrapportage leefgebied wilde zwijnen afschot 2010–2011 en 2011–2012 [Annual report wild boar hunting in national park 2010–2011 and 2011–2012] Roermond, the Netherlands.

Fietsersbond, 2012. Database Fietsersbond Routeplanner, ⟨www.fietsersbond.nl⟩.

Goossen, C.M., 2009. Monitoring recreatiegedrag van Nederlanders in landelijke gebieden, jaar 2006/2007 [Monitoring recreational behaviour of the Dutch in rural areas, year 2006/2007], Wettelijke Onderzoekstaken Natuur & Milieu. Wageningen UR, Wageningen, p. 113.

Grubešić, M., Konjević, D., Severin, K., Hadžiosmanović, M., Tomljanović, K., Mašek, T., Margaletić, J., Slavica, A., 2011. Dressed and undressed weight in naturally bred wild boar (Sus scrofa): The possible influence of crossbreeding. Acta Aliment. 40, 502–508 http://dx.doi.org/10.1556/AAlim.40.2011.4.10.

Haines-Young, R., Potschin, M., 2010a. The links between biodiversity, ecosystem services and human well-being. In: Raffaelli, D., Frid, C. (Eds.), Ecosystem Ecology: A New Synthesis. Cambridge University Press, Cambridge, pp. 110–139.

Haines-Young, R., Potschin, M., 2010b. Proposal for a Common International Classification of Ecosystem Goods and Services (CICES) for Integrated Environmental and Economic Accounting. University of Nottingham, Nottingham, United Kingdom.

Hanegraaf, M., Hoffland, E., Kuikman, P., Brussaard, L., 2009. Trends in soil organic matter contents in Dutch grasslands and maize fields on sandy soils. Eur. J. Soil Sci. 60, 213–222.

Hazeu, G., 2009. Dataset Landelijk Grondgebruiksbestand Nederland, vol. 6. Alterra, Wageningen, the Netherlands.

Hazeu, G., Schuiling, C., Dorland, G.J., Oldengarm, J., Gijsbertse, H.A., 2010. Landelijk Grondgebruiksbestand Nederland versie 6 (LGN6) – vervaardiging, nauwkeurigheid en gebruik [Land-use map of the Netherlands version 6 (LGN6) – production, accuracy and use]. Alterra, Wageningen UR, Wageningen, the Netherlands.

Janssens, I.A., Freibauer, A., Schlamadinger, B., Ceulemans, R., Ciais, P., Dolman, A.J., Heiman, M., Nabuurs, G.J., Smith, P., Valentini, R., Schulze, E.D., 2005. The carbon budget of terrestrial ecosystems at country scale – a European case study. Biogeosciences 2, 15–26 http://dx.doi.org/10.5194/bg-2-15-2005.

Jongman, R.H.G., 2002. Homogenisation and fragmentation of the European landscape: ecological consequences and solutions. Landsc. Urban Plan. 58, 211–221 http://dx.doi.org/10.1016/S0169-2046(01)00222-5.

Jongmans, A.G., van den Berg, M.W., Sonneveld, M.P.W., Peek, G.J.W.C., van den Berg van Saparoea, R.M., 2013. Landschappen van Nederland [Landscapes of the Netherlands], First ed. Wageningen Academic Publishers, Wageningen, the Netherlands.

Kareiva, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S., 2011. Natural Capital: Theory and Practice of Mapping Ecosystem Services. Oxford University Press, USA.

Kuikman, P., de Groot, W., Hendriks, R., Verhagen, J., de Vries, F., 2003. Stocks of C in soils and emissions of CO2 from agricultural soils in the Netherlands. Alterra-rapport, Alterra, Wageningen, the Netherlands.

Künzli, N., Kaiser, R., Medina, S., Studnicka, M., Chanel, O., Filliger, P., Herry, M., Horak Jr, F., Puybonnieux-Texier, V., Quénel, P., Schneider, J., Seethaler, R., Vergnaud, J.C., Sommer, H., 2000. Public-health impact of outdoor and traffic-related air pollution: A European assessment. Lancet 356, 795–801 http://dx.doi.org/10.1016/S0140-6736(00)02653-2.

Larigauderie, A., Prieur-Richard, A.-H., Mace, G.M., Lonsdale, M., Mooney, H.A., Brussaard, L., Cooper, D., Cramer, W., Daszak, P., Díaz, S., Duraiappah, A., Elmqvist, T., Faith, D.P., Jackson, L.E., Krug, C., Leadley, P.W., Le Prestre, P., Matsuda, H., Palmer, M., Perrings, C., Pulleman, M., Reyers, B., Rosa, E.A., Scholes, R.J., Spehn, E., Turner, Ii, B.L., Yahara, T., 2012. Biodiversity and ecosystem services science for a sustainable planet: the DIVERSITAS vision for 2012–20. Curr. Opin. Environ. Sustain. 4, 101–105 http://dx.doi.org/10.1016/j.cosust.2012.01.007.

LEI, Statistics Netherlands, 2011. Land – en tuinbouwcijfers 2011 [Agri – and horticulture figures 2011], The Hague, the Netherlands.

LEI, Statistics Netherlands, 2012. Land – en tuinbouwcijfers 2012 [Agri – and horticulture figures 2012], the Hague, the Netherlands.

Lesschen, J.P., Heesmans, H., Mol-Dijkstra, J., van Doorn, A., Verkaik, E., van den Wyngaert, I., Kuikman, P., 2012. Mogelijkheden voor koolstofvastlegging in de Nederlandse landbouw en natuur. Alterra, Wageningen, the Netherlands.

MA, 2003. Ecosystems and Human Well-Being: A Framework for Assessment. Island Press, Washington D.C..

MA, 2005. Ecosystems and Human Well-Being - Current State & Trends, Volume 1. Island Press, Washington D.C..

Maes, J., Braat, L., Jax, K., Hutchins, M., Furman, E., Termansen, M., Luque, S., Paracchini, M.L., Chauvin, C., Williams, R., Volk, M., Lautenbach, S., Kopperoinen, L., Schelhaas, M.-J., Weinert, J., Goossen, M., Dumont, E., Strauch, M., Görg, C., Dormann, C., Katwinkel, M., Zulian, G., Varjopuro, R., Hauck, J., Forsius, M., Hengeveld, G., Perez-Soba, M., Bouraoui, F., Scholz, M., Schulz-Zunkel, C., Lepistö, A., Polishchuk, Y., Bidoglio, G., 2011. A spatial assessment of ecosystem services in Europe: methods, case studies and policy analysis. European Commission, Joint Research Centre, Institute for Environment and Sustainability, Luxembourg.

Maes, J., Egoh, B., Willemen, L., Liquete, C., Vihervaara, P., Schägner, J.P., Grizzetti, B., Drakou, E.G., Notte, A.L., Zulian, G., Bouraoui, F., Luisa Paracchini, M., Braat, L., Bidoglio, G., 2012. Mapping ecosystem services for policy support and decision making in the European Union. Ecosyst. Serv. 1, 31–39 http://dx.doi.org/10.1016/j.ecoser.2012.06.004.

Mäler, K.-G., Aniyar, S., Jansson, Å., 2008. Accounting for Ecosystem Services as a Way to Understand the Requirements for Sustainable Development. Proc. Natl. Acad. Sci. USA 105, 9501–9506 http://dx.doi.org/10.2307/25463002.

Martínez-Harms, M.J., Balvanera, P., 2012. Methods for mapping ecosystem service supply: a review. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 8, 17–25 http://dx.doi.org/10.1080/21513732.2012.663792.

Nabuurs, G.J., Van Den Wyngaert, I.J.J., Daamen, W., Kramer, H., Kuikman, P., 2008. The Dutch National System for forest sector greenhouse gas reporting under UNFCCC. Mitig. Adapt. Strateg. Glob. Change 13, 267–282.

Naeff, H.S.D., Smidt, R.A., Vos, E.C., 2011. Geactualiseerd GIAB-bestand 2010 voor Nederland [Updated GIAB-database 2010 for the Netherlands]. Alterra, Wageningen, the Netherlands.

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R.E., Lehner, B., Malcolm, T.R., Ricketts, T.H., 2008. Global mapping of ecosystem services and conservation priorities. Proc. Natl. Acad. Sci. 105, 9495–9500 http://dx.doi.org/10.1073/pnas.0707823105.

NASA LP DAAC, 2012. Terra/MODIS Net Primary Production Yearly L4 Global 1 km, Sioux Falls, USA.

NBTC-NIPO Research, 2012a. ContinuVrijeTijdsOnderzoek 2010–2011.

NBTC-NIPO Research, 2012b. Kerncijfers over vrijetijd van Nederlanders [Key figures about leisure of the Dutch].

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D., Chan, K.M.A., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., Shaw, M., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front. Ecol. Environ. 7, 4–11 (10.1890/080023).

Nemec, K.T., Raudsepp-Hearne, C., 2013. The use of geographic information systems to map and assess ecosystem services. Biodivers. Conserv. 22, 1–15 http://dx.doi.org/10.1007/s10531-012-0406-z.

Pelucchi, C., Negri, E., Gallus, S., Boffetta, P., Tramacere, I., La Vecchia, C., 2009. Long-term particulate matter exposure and mortality: a review of European epidemiological studies. BMC Public Health 9, 453.

Petz, K., van Oudenhoven, A.P.E., 2012. Modelling land management effect on ecosystem functions and services: a study in the Netherlands. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 8, 135–155 http://dx.doi.org/10.1080/21513732.2011.642409.

Powe, N.A., Willis, K.G., 2004. Mortality and morbidity benefits of air pollution (SO2 and PM10) absorption attributable to woodland in Britain. J. Environ. Manag. 70, 119–128 http://dx.doi.org/10.1016/j.jenvman.2003.11.003.

Provincie Limburg, 2010a. Grondwaterbeschermingsgebieden [groundwater protection zones], ⟨http://portal.prvlimburg.nl/geo_dataportaal/viewer.do⟩.

Provincie Limburg, 2010b. Groundwater extraction 2010.

Raudsepp-Hearne, C., Peterson, G.D., Bennett, E.M., 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proc. Natl. Acad. Sci. 107, 5242–5247 (10.1073/pnas.0907284107).

Reijneveld, A., van Wensem, J., Oenema, O., 2009. Soil organic carbon contents of agricultural land in the Netherlands between 1984 and 2004. Geoderma 152, 231–238.

Schröter, M., Barton, D.N., Remme, R.P., Hein, L., 2014a. Accounting for capacity and flow of ecosystem services: a conceptual model and a case study for Telemark, Norway. Ecol. Indic. 36, 539–551 http://dx.doi.org/10.1016/j.ecolind.2013.09.018.

Schröter, M., Remme, R.P., Hein, L., 2012. How and where to map supply and demand of ecosystem services for policy-relevant outcomes? Ecol. Indic. 23, 220–221 http://dx.doi.org/10.1016/j.ecolind.2012.03.025.

Schröter, M., Zanden, E.H., Oudenhoven, A.P., Remme, R.P., Serna-Chavez, H.M., Groot, R.S., Opdam, P., 2014. Ecosystem services as a contested concept: a synthesis of critique and counter-arguments. Conserv. Lett. http://dx.doi.org/10.1111/conl.12091.

Schulp, C.J., Alkemade, R., 2011. Consequences of uncertainty in global-scale land cover maps for mapping ecosystem functions: an analysis of pollination efficiency. Remote Sens. 3, 2057–2075.

Schulp, C.J.E., Lautenbach, S., Verburg, P.H., 2014. Quantifying and mapping ecosystem services: demand and supply of pollination in the European Union. Ecol. Indic. 36, 131–141 http://dx.doi.org/10.1016/j.ecolind.2013.07.014.

Schulp, C.J.E., Nabuurs, G.-J., Verburg, P.H., 2008. Future carbon sequestration in Europe–effects of land use change. Agricult. Ecosyst. Environ. 127, 251–264.

Seppelt, R., Dormann, C.F., Eppink, F.V., Lautenbach, S., Schmidt, S., 2011. A quantitative review of ecosystem service studies: approaches, shortcomings and the road ahead. J. Appl. Ecol. 48, 630–636 http://dx.doi.org/10.1111/j.1365-2664.2010.01952.x.

Serna-Chavez, H., Schulp, C., van Bodegom, P., Bouten, W., Verburg, P., Davidson, M., 2014. A quantitative framework for assessing spatial flows of ecosystem services. Ecol. Indic. 39, 24–33.

Spangenberg, J.H., Görg, C., Truong, D.T., Tekken, V., Bustamante, J.V., Settele, J., 2014. Provision of ecosystem services is determined by human agency, not ecosystem functions. Four case studies. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 10, 40–53.

Statistics Netherlands, 2012. Regionale Kernscijfers Nederland [Regional Key Figures the Netherlands], The Hague/Heerlen, the Netherlands ⟨http://statline.cbs.nl/StatWeb/publication/?DM=SLNL&PA=70072ned&D1=0,56-118&D2=16&D3=15-17&HDR=T&STB=G1,G2&VW=T⟩.

Statistics Netherlands, 2013a. Arable crops; production, to region ⟨http://statline.cbs.nl/StatWeb/publication/?DM=SLEN&PA=7100ENG&D1=a&D2=a&D3=l&D4=16&LA=EN&VW=T⟩.

Statistics Netherlands, 2013b. Landbouw; gewassen, dieren, en grondgebruik naar regio [Agriculture; crops, animals, and land use per region] The Hague/Heerlen, the Netherlands ⟨http://statline.cbs.nl/StatWeb/publication/?DM=SLNL&PA=80780ned&D1=0,2-7,13-18,24,50,90,116,156,159,226,321,327,332,364,383-384,388,400-403,406,409,418,427,444,459,504,512,519,526,538&D2=16&D3=0,5,11-12&VW=T⟩.

Statistics Netherlands, Kadaster, 2009. Wijk- en buurtkaart 2009 [District and neighbourhood map 2009], Zwolle, the Netherlands.

Stichting Landelijk Fietsplatform, 2009. Zicht op Nederland Fietsland (Overview of the Netherlands as Cycling Nation), Amersfoort, the Netherlands.
Stichting Landelijk Fietsplatform, 2013. Fietsrecreatiemonitor: Cijfers en trends [Recreational cycling monitor: figures and trends] 〈http://www.fietsplatform.nl/fietsrecreatiemonitor/cijfers〉.
Sumarga, E., Hein, L., 2014. Mapping Ecosystem Services for Land Use Planning, the Case of Central Kalimantan. Environ. Manag., 1–14.
TEEB, 2010. The Economics of Ecosystems and Biodiversity: ecological and economic foundations. Earthscan, London and Washington.
United Nations, 2012. The future we want: Outcome document adopted at Rio +20, Rio de Janeiro, Brazil.
United Nations, European Commission, International Monetary Fund, Organisation for Economic Co-operation and Development, World Bank, 2009. System of National Accounts 2008, New York.
van Oudenhoven, A.P.E., Petz, K., Alkemade, R., Hein, L., de Groot, R.S., 2012. Framework for systematic indicator selection to assess effects of land management on ecosystem services. Ecol. Indic. 21, 110–122 http://dx.doi.org/10.1016/j.ecolind.2012.01.012.
van Zanten, B., Verburg, P., Espinosa, M., Gomez-y-Paloma, S., Galimberti, G., Kantelhardt, J., Kapfer, M., Lefebvre, M., Manrique, R., Piorr, A., Raggi, M., Schaller, L., Targetti, S., Zasada, I., Viaggi, D., 2014. European agricultural landscapes, common agricultural policy and ecosystem services: a review. Agron. Sustain. Dev. 34, 309–325 http://dx.doi.org/10.1007/s13593-013-0183-4.
Velders, G.J.M., Aben, J.M.M., Jimmink, B.A., Geilenkirchen, G.P., van der Swaluw, E., de Vries, W.J., Wesseling, J., van Zanten, M.C., 2012. Grootschalige concentratie- en depositiekaarten Nederland - Rapportage 2012. National Institute for Public Health and the Environment (RIVM).
Vogelzang, T., Gies, E., Michels, R., Wisman, A., Hoefs, R., Smidt, R., 2010. Puzzelen met de ruimte in Limburg: Ruimteclaims in het Limburgs landelijk gebied [Juggling with space in Limburg: Claims to space in rural Limburg]. LEI, Wageningen UR, The Hague, the Netherlands.
Waterleiding Maatschappij Limburg, 2013. Waterproductie 〈http://www.wml.nl/nl-nl/158/5905/waterproductie.aspx〉.
Willemen, L., Hein, L., van Mensvoort, M.E.F., Verburg, P.H., 2010. Space for people, plants, and livestock? Quantifying interactions among multiple landscape functions in a Dutch rural region. Ecol. Indic. 10, 62–73.