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Analysis

Monetary accounting of ecosystem services: A test case for Limburg province, the Netherlands

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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 the 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.

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 the integration of ecosystem services (ES) 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 ES 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; Edens and Hein, 2013; Schröter et al., 2014a; Stoneham et al., 2012; UK NEA, 2011; Weber, 2011).

The SEEA EEA emphasizes the importance of a spatial approach for ecosystem accounting, for both biophysical quantification and monetary valuation of ES (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 (Schröter et al., in press; Sumarga and Hein, 2014). Second, it can help to shed light on spatial interrelationships between ES and dependence of ES on socio-environmental conditions (Schröter et al., 2014a). Third, spatial modelling can offer wall-to-wall coverage of ES 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., 2014a, in press), only few studies apply monetary ecosystem accounting aligned with SNA principles for

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multiple ES in a spatially explicit way (e.g. Campos et al., 2014). Monetary valuation can be a valuable complement to biophysical ES assessments (Schröter et al., 2014b; Troy and Wilson, 2006) and, for instance, be used to quantify and sum ES 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 ES and how resulting values can be represented in accounts for Limburg province, the Netherlands. Valuation is carried out for seven ES, 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 ES 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 ES 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).

2. Methodology

2.1. Case study description

Limburg province is located in the south-east of the Netherlands and covers approximately 2200 km^2 (Fig. 1). Limburg is densely populated (522 inhabitants per km^{-2} in 2010), with a total population of 1.1 million people (Statistics Netherlands, 2013c). 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).

2.2. Biophysical spatial ES 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 ES were modelled for the year 2010. Most biophysical models were developed based on the Dutch 25 × 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).

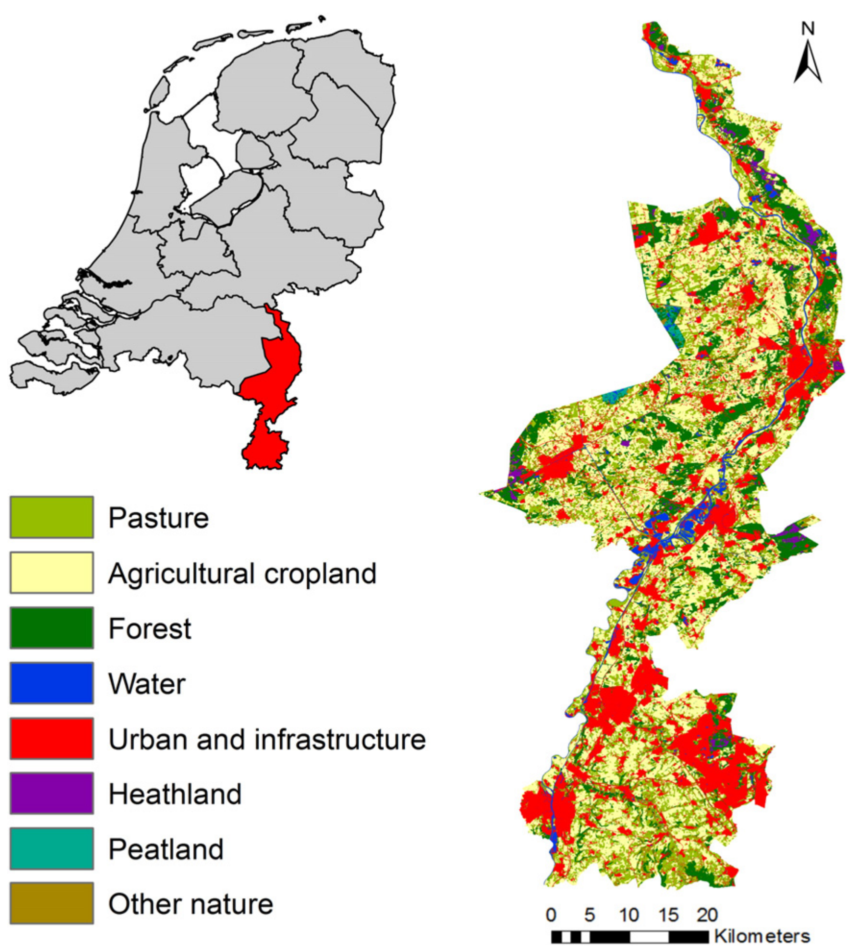

Fig. 1. Location and land cover of Limburg province, the Netherlands. Full colour version of this figure can be found on the journal website.

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 (Statistics Netherlands, 2013d; ZKA Consultants and Planners, 2011). See Appendix I for a model description and underlying assumptions.

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.

2.3. Methodological foundation: ES 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). The 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 ES (UN et al., 2014a). We applied the avoided damage costs (Section 2.3.3).

The SEEA EEA defines ES 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).

2.3.1. Resource rent method

According to the resource rent method, the 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:

$$RR = TR - (IC + LC + FC) \tag{1}$$

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, and 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.[2] 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.

2.3.2. Replacement cost method

The replacement cost method is a cost-based approach to value ES 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 (Bockstael et al., 2000; NRC, 2004; Shabman and Batie, 1978). 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).

2.3.3. 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.

2.4. Monetary ES 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, 2013a). An ES value map was produced

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

for each service. Ecosystem accounting tables were set up based on the model outcomes. Monetary values of the ES 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).

2.4.1. 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, 2013a). 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, 2013c): 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, 2013e). 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[3] and costs per hectare for six arable crops (wheat, barley, seed potatoes, starch potatoes, and potatoes and sugar beets) (LEI, 2013c) was used as input for resource rent calculations. The available intermediate costs items for these crops included costs for planting and energy costs (LEI, 2013c). 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, 2013d). 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 average farm.[4] Labour costs were deducted for each of the crop types (LEI, 2013d,e). The user costs of fixed capital were estimated using information about depreciation from the profit and loss account (LEI, 2013d,e), and information about the stock of fixed capital from the balance sheet of crop farms (LEI, 2013b). 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 1). For arable crops BINternet data was used (LEI, 2013c), for "other crops" information about crop yield per hectare was obtained from Statistics Netherlands (2013e).

Table 1
Revenue, costs and resource rent for the four modelled crop groups, calculated based on data from LEI (2013a).

	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

2.4.2. 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 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, 2013a). 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.

2.4.3. 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.

2.4.4. 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 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

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

[4] 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% in 2010). The absence of information on (net) taxes on production is therefore expected not to have a large effect on our results.

forest cover and the decrease of the PM_{10} concentration in the lower atmosphere can be expressed as:

$$C_p = 0.15 * F_p \tag{2}$$

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 Eq. (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 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 (Oosterbaan et al., 2006; Powe and Willis, 2004), 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).

2.4.5. Carbon sequestration

Carbon 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.

2.4.6. 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, 2013b), of which 23% can be accounted to business trips (ZKA Consultants and 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 and 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 (2013b). 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 and 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

Table 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/m^3))	Treatment costs per case for 2010 (€)	Costs per person per μg PM_{10} (€/person/μg/m^3)
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).

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.

2.4.7. 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.

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 ES 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 ES, because they are all both rival and excludable. Carbon sequestration and air quality regulation, nature tourism and drinking water production were classified as public ES. 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. Results

3.1. ES valuation and maps

3.1.1. ES valuation

For the ES crop production, the resource rent was estimated to be €46 million (Table 3). The specific RRs 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). 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). The value of the ES hunting was estimated to be around €3 million (Table 4).

3.1.2. ES value maps

A monetary value map was produced for each modelled ES (Fig. 2). In Limburg, crop production and fodder production are spatially mutually exclusive, because these ES are located on distinct land cover types. Monetary values of these two ES 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 ES are found in that particular land cover type. For hunting, Fig. 2 shows the results for the median value column in Table 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.

Aggregated value maps of the ES are presented in Fig. 3. Fig. 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 ES flows are generally low. The high value in the south of the province can be explained by overlap between multiple ES 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 (Fig. 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.

ES with a public and private character have a different spatial distribution of monetary value. The value of private ES (Fig. 4a) is predominantly found on agricultural land, whereas the value of the modelled public ES (Fig. 4b) is largely found in areas under some form of protection (e.g. nature areas or drinking water protection zones). The value of public ES 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 ES is highly scattered throughout the province. Private ES attribute 52% of the calculated ES value, while public ES attribute 48% (Table 4), but

Table 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 4
Hunting value per land cover type for the lowest, average and highest indicated values per ha.

Land cover type	Area (×1000 ha)	Value range per ha (€/ha)[a]	Range provincial value (×1000 €)	Median provincial value (×1000 €)
Contiguous forest	22	20–40	433–866	650
Forest patches	84	10–15	1257–1676	1466
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		2058–3217	2637

[a] Source: van Hout (personal communication).

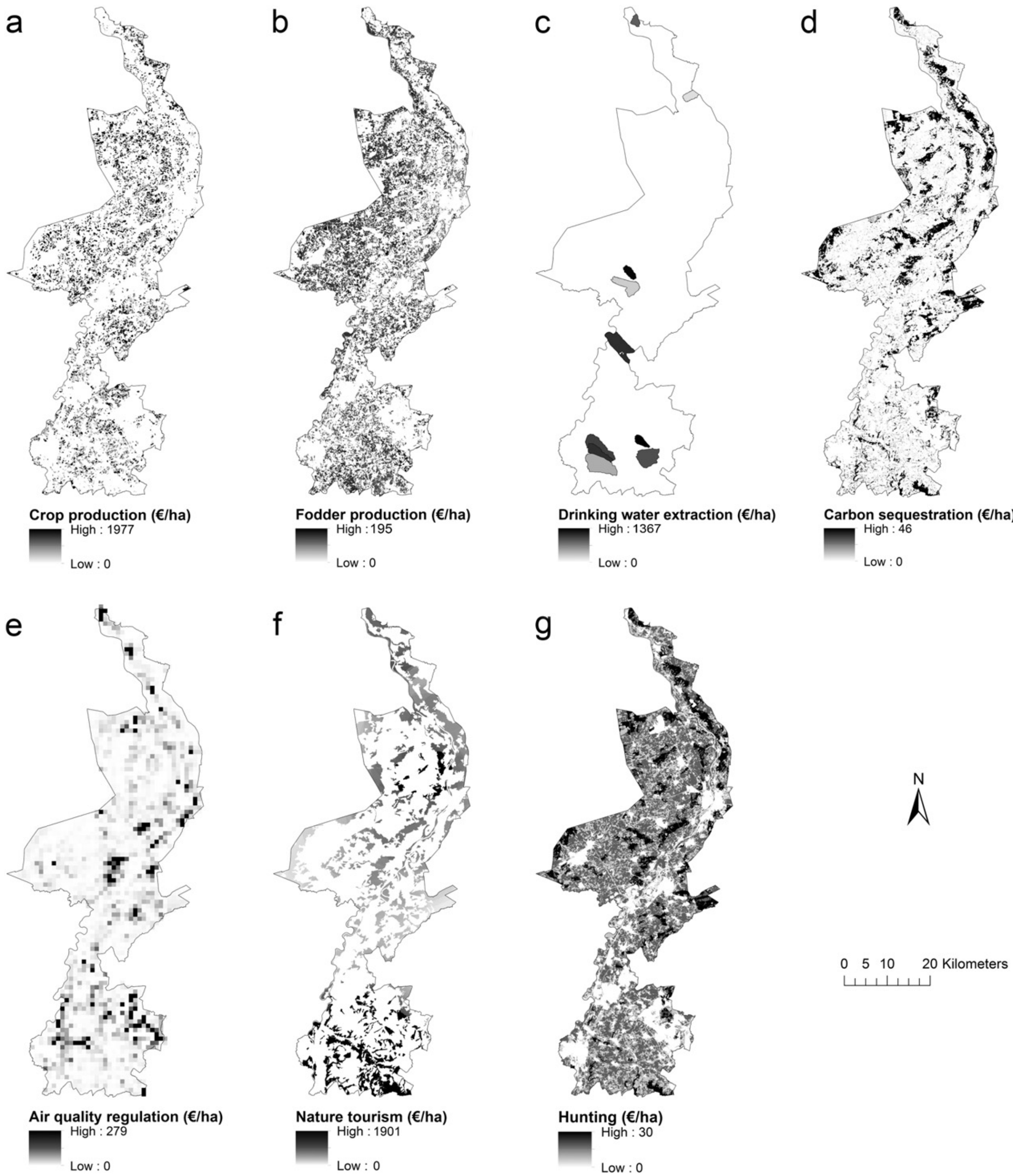

Fig. 2. Monetary value maps of the modelled ecosystem services: (a) crop production, (b) fodder production, (c) drinking water extraction, (d) carbon sequestration, (e) air quality regulation, (f) nature tourism, and (g) hunting. In all maps white indicates the lowest values and black the highest values. All values are in €/ha.

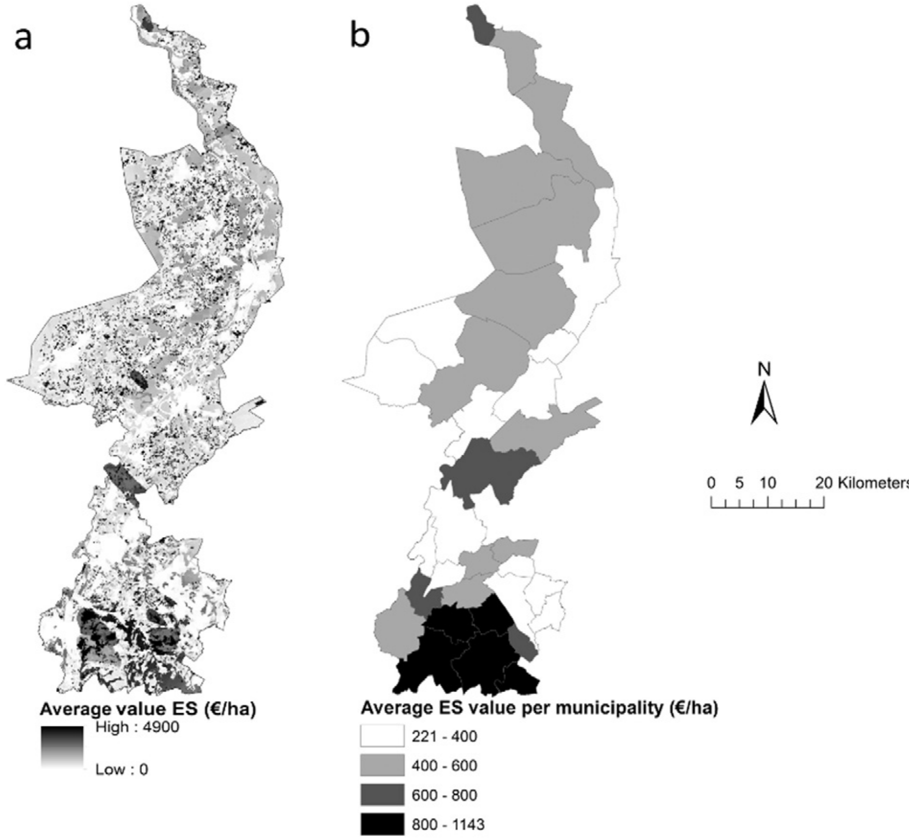

Fig. 3. Aggregated value maps (€/ha) for ecosystem services represented (a) per hectare and (b) per municipality.

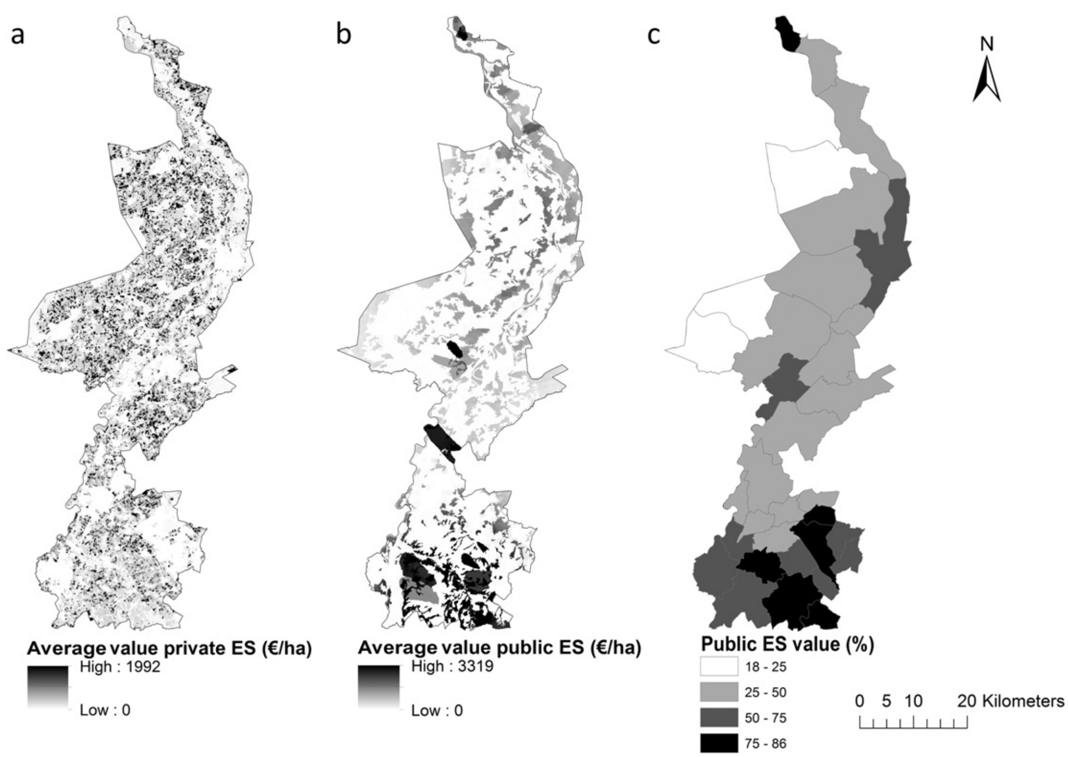

Fig. 4. Aggregated monetary value maps for (a) the private ecosystem services and (b) the public ecosystem services; and (c) the percentage of value generated by public ecosystem services per municipality.

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

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

this relative share of public and private ES is not evenly distributed across the province (Fig. 4c). In the central municipalities the contribution of private ES is generally higher than that of public ES, while in the southern municipalities the contribution of public ES 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 ES to the aggregated value is higher than that of public ES.

3.2. Ecosystem accounting tables

3.2.1. Aggregated value of modelled ES

The total, SNA-aligned monetary value of the modelled ES for Limburg was estimated to be about €112 million in 2010 (Table 5). The average value per hectare was €508 (SD $\pm$655). Crop production and nature tourism constitute the two most important ES in monetary terms. Together, these two ES contribute about 75% of the monetary value of the modelled ES. The two regulating services have the smallest calculated monetary value. For the ES 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%).

3.2.2. Accounting per ecosystem/land cover

Cropland accounts for approximately 55% of the annual value of the modelled ES (Table 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 value from ES per hectare. Public ES are strongly dominant in all land covers except *cropland* and *pasture*. *Cropland* is the only land cover type in which private ES 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.

4. Discussion

4.1. Uncertainty and sensitivity in valuation approaches

4.1.1. 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 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., 2014). In our biophysical models uncertainties are commonly related to a lack of local data on the ES (Table 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.

The monetary valuation methods have additional uncertainties, mostly related to the aggregation level of data (Table 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. Bateman et al., 2011; Chee, 2004; Liu et al., 2010; NRC, 2004; Turner et al., 2010), 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

Table 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	4900	18	82
Pasture	20.2	18.6	412	507	10	3361	61	39
Forest	15.3	19.9	587	473	56	3226	96	4
Urban and infrastructure	23.6	4.8	90	277	0	2900	99	1
Other nature	2.7	4.9	814	687	15	3186	94	6
Water	3.0	1.6	239	313	0	2906	100	0
Heath	1.0	0.9	426	288	20	1923	96	4
Peatland	0.3	0.3	457	135	21	653	97	3
Total province		112	508	655	0	4900	48	52

Table 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) —Information missing about (net) taxes on production —Part of the resource rent will reflect mixed income
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 —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 —Analysis done at a 1 km^2 resolution, coarse compared to other ES	—Little national data on costs of treatments resulting from air pollution —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 —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 —Hunting rights only a partial indicator of hunting as a recreational activity

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.

4.1.2. Sensitivity of results

The sensitivity of outcomes was tested for some ES 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 2×2 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 2×2 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 and 8 million depending on the chosen discount rate. This calculation shows that, in this particular case, applying a different discount rate will 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 4), resulting in a total ES value of between €2.1 and €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.

4.2. Limitations of using exchange value and not welfare

Not all ES 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; Duraiappah and Muñoz, 2012; Mäler et al., 2008). 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 II 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).

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., in press). 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, 2013b), but it is important to keep in mind that we have not valued all ES in this study. Furthermore, exchange values of ES 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 ES according to an ecosystem accounting approach, which is just one of several possible ways to value ES 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 tradeoffs between ES. 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 (Fig. 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 (Plieninger et al., 2013; Raymond et al., 2009) are also of crucial importance here. Spatially explicit monetary accounting can also raise awareness on ES tradeoffs 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.

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 ES 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 ES provide a higher contribution to the aggregated provincial value than public ES. 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 ES, 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 ES 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 ES 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 are 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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Appendices

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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.
Alfa Accountants en Adviseurs, 2011. Cijfers die Spreken 2011. Melkveehouderij, Wageningen, the Netherlands.
Arrow, K.J., Dasgupta, P., Mäler, K.-G., 2003. Evaluating projects and assessing sustainable development in imperfect economies. Environ. Resour. Econ. 26, 647–685.
Bade, T., Enzerink, R., van Middendorp, B., Smid, G., 2010. Wild van de economie. KNNV Uitgeverij, Zeist, the Netherlands.
Banzhaf, H.S., Boyd, J., 2012. The architecture and measurement of an ecosystem services index. Sustainability 4, 430–461.
Bartelmus, P., 2013. Environmental–economic accounting: progress and digression in the SEEA revisions. Rev. Income Wealth 60, 887–904.
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.
Bateman, I.J., Harwood, A.R., Mace, G.M., Watson, R.T., Abson, D.J., Andrews, B., Binner, A., Crowe, A., Day, B.H., Dugdale, S., Fezzi, C., Foden, J., Hadley, D., Haines-Young, R., Hulme, M., Kontoleon, A., Lovett, A.A., Munday, P., Pascual, U., Paterson, J., Perino, G., Sen, A., Siriwardena, G., van Soest, D., Termansen, M., 2013. Ecosystem services: response. Science 342, 421–422. http://dx.doi.org/10.1126/science.342.6157.421-b.
Berendsen, H.J.A., 2005. Landschappelijk Nederland. 3rd ed. Van Gorcum, Assen, the Netherlands.

Bockstael, N.E., Freeman, A.M., Kopp, R.J., Portney, P.R., Smith, V.K., 2000. On measuring economic values for nature. Environ. Sci. Technol. 34, 1384–1389.
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.
Campos, P., Caparrós, A., 2006. Social and private total Hicksian incomes of multiple use forests in Spain. Ecol. Econ. 57, 545–557. http://dx.doi.org/10.1016/j.ecolecon.2005.05.005.
Campos, P., Daly-Hassen, H., Ovando, P., Chebil, A., Oviedo, J.L., 2009. Economics of multiple use cork oak woodlands: two case studies of agroforestry systems. In: Mosquera-Losada, M.R., Rigueiro-Rodriguez, A., McAdam, J. (Eds.), Agroforestry in Europe. Springer, Dordrecht, the Netherlands, pp. 269–295.
Campos, P., Caparrós, A., Beguería, S., Carranza, J., Díaz-Balteiro, L., Díaz, M., Farizo, B.A., Herruzo, C., Martínez, F., Montero, G., Ovando, P., Oviedo, J.L., Soliño, M., Aldea, J., Almazán, E., Álvarez, A., Concepción, E.D., Fernández, C., de Frutos, P., Martínez-Jauregui, M., Mesa, B., Pasalodos, M., Romero, C., Serrano, R., Torres, J., 2014. RECAMAN: Manufactured and Environmental Total Incomes of Andalusian Forest. CSIC, Madrid, Spain.
Cavendish, W., 2002. Quantitative methods for estimating the economic value of resource use to rural households. In: Campbell, B.M., Luckert, M.K. (Eds.), Uncovering the Hidden Harvest. Earthscan Publications Ltd., London, UK, pp. 17–65.
Chan, K.M.A., Satterfield, T., Goldstein, J., 2012. Rethinking ecosystem services to better address and navigate cultural values. Ecol. Econ. 74, 8–18. http://dx.doi.org/10.1016/j.ecolecon.2011.11.011.
Chee, Y.E., 2004. An ecological perspective on the valuation of ecosystem services. Biol. Conserv. 120, 549–565. http://dx.doi.org/10.1016/j.biocon.2004.03.028.
Costanza, R., 2008. Ecosystem services: multiple classification systems are needed. Biol. Conserv. 141, 350–352. http://dx.doi.org/10.1016/j.biocon.2007.12.020.
Daily, G.C., Polasky, S., Goldstein, J., Kareiva, P.M., Mooney, H.A., Pejchar, L., Ricketts, T.H., Salzman, J., Shallenberger, R., 2009. Ecosystem services in decision making: time to deliver. Front. Ecol. Environ. 7, 21–28. http://dx.doi.org/10.1890/080025.
de Groot, R.S., Alkemade, R., Braat, L., Hein, L., Willemen, L., 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol. Complex. 7, 260–272.
de Vries, S., Goossen, M., 2002. Modelling recreational visits to forests and nature areas. Urban For. Urban Green. 1, 5–14. http://dx.doi.org/10.1078/1618-8667-00002.
Duraiappah, A.K., Muñoz, P., 2012. Inclusive wealth: a tool for the United Nations. Environ. Dev. Econ. 17, 362–367.
Edens, B., Graveland, C., 2014. Experimental valuation of Dutch water resources according to SNA and SEEA. Water Resour. Econ. 7, 66–81. http://dx.doi.org/10.1016/j.wre.2014.10.003.
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.
Goldstein, J.H., Caldarone, G., Duarte, T.K., Ennaanay, D., Hannahs, N., Mendoza, G., Polasky, S., Wolny, S., Daily, G.C., 2012. Integrating ecosystem-service tradeoffs into land-use decisions. Proc. Natl. Acad. Sci. 109, 7565–7570. http://dx.doi.org/10.1073/pnas.1201040109.
Gómez-Baggethun, E., Barton, D.N., 2013. Classifying and valuing ecosystem services for urban planning. Ecol. Econ. 86, 235–245.
Gupta, T.R., Foster, J.H., 1975. Economic criteria for freshwater wetland policy in Massachusetts. Am. J. Agric. Econ. 57, 40–45. http://dx.doi.org/10.2307/1238838.
Hazeu, G., 2009. Dataset Landelijk Grondgebruiksbestand Nederland 6, Wageningen, the Netherlands.
Hein, L., 2011. Economic benefits generated by protected areas: the case of the Hoge Veluwe forest, the Netherlands. Ecol. Soc. 16, 13.
IPCC, 2007. Summary for Policymakers. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (Eds.), Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
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). 1st ed. Wageningen Academic Publishers, Wageningen, the Netherlands.
Kemkes, R.J., Farley, J., Koliba, C.J., 2010. Determining when payments are an effective policy approach to ecosystem service provision. Ecol. Econ. 69, 2069–2074.
Konarska, K.M., Sutton, P.C., Castellon, M., 2002. Evaluating scale dependence of ecosystem service valuation: a comparison of NOAA-AVHRR and Landsat TM datasets. Ecol. Econ. 41, 491–507.
LEI, 2013a. BINternet. Accessed on: 20-12-2013. http://www3.lei.wur.nl/binternet_asp/index.aspx.
LEI, 2013b. BINternet table "Balance sheet of arable farms", The Hague, the Netherlands. Accessed on: 20-12-2013. http://www3.lei.wur.nl/binternet_asp/index.aspx.
LEI, 2013c. BINternet table "Gross margin, technical result and price of arable products of arable farms (amounts and prices excluding VAT)", The Hague, the Netherlands. Accessed on: 20-12-2013. http://www3.lei.wur.nl/binternet_asp/index.aspx.
LEI, 2013d. BINternet table "Holding results of arable farms", The Hague, the Netherlands. Accessed on: 20-12-2013. http://www3.lei.wur.nl/binternet_asp/index.aspx.
LEI, 2013e. BINternet table "Holding results of open field vegetable farms", The Hague, the Netherlands. Accessed on: 20-12-2013. http://www3.lei.wur.nl/binternet_asp/index.aspx.
LEI, Statistics Netherlands, 2011. Land- en tuinbouwcijfers 2011 (Agri- and horticulture figs. 2011). (The Hague, the Netherlands).
Liu, S., Costanza, R., Farber, S., Troy, A., 2010. Valuing ecosystem services. Ann. N. Y. Acad. Sci. 1185, 54–78.
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. 105, 9501–9506. http://dx.doi.org/10.1073/pnas.0708856105.
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.
McDonald, A.G., Bealey, W.J., Fowler, D., Dragosits, U., Skiba, U., Smith, R.I., Donovan, R.G., Brett, H.E., Hewitt, C.N., Nemitz, E., 2007. Quantifying the effect of urban tree planting on concentrations and depositions of PM10 in two UK conurbations. Atmos. Environ. 41, 8455–8467.
NRC, 2004. Valuing Ecosystem Services: Toward Better Environmental Decision-making. National Research Council, Washington DC, USA.
Obst, C., Vardon, M., 2014. Recording environmental assets in the national accounts. Oxf. Rev. Econ. Policy 30, 126–144. http://dx.doi.org/10.1093/oxrep/gru003.
Obst, C., Edens, B., Hein, L., 2013. Ecosystem services: accounting standards. Science 342, 420.
Oosterbaan, A., Tonneijck, A.E.G., de Vries, E.A., 2006. Kleine landschapselementen als invangers van fijnstof en ammoniak. Alterra-rapport 1419. Alterra, Wageningen UR, Wageningen, p. 58.
Oviedo, J., Campos, P., Caparrós, A., 2010. Simulated Exchange Value Method: Applying Green National Accounting to Forest Public Recreation. Instituto de Políticas y Bienes Públicos CCHS-CSIC, Madrid, Spain.
Plieninger, T., Dijks, S., Oteros-Rozas, E., Bieling, C., 2013. Assessing, mapping, and quantifying cultural ecosystem services at community level. Land Use Policy 33, 118–129.
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.
Preiss, P., Friedrich, R., Klotz, V., 2008. NEEDS, Report on the Procedure and Data to Generate Averaged/Aggregated Data.
Raymond, C.M., Bryan, B.A., MacDonald, D.H., Cast, A., Strathearn, S., Grandgirard, A., Kalivas, T., 2009. Mapping community values for natural capital and ecosystem services. Ecol. Econ. 68, 1301–1315.
Remme, R.P., Schröter, M., Hein, L., 2014. Developing spatial biophysical accounting for multiple ecosystem services. Ecosyst. Serv. 10, 6–18. http://dx.doi.org/10.1016/j.ecoser.2014.07.006.
RIVM, 2012. Maatschappelijke kosten voor astma, COPD en respiratoire allergie [Societal costs of asthma, COPD and respiratory allergy in the Netherlands]. RIVM - National Institute for Public Health and the Environment. Bilthoven, the Netherlands.
Schägner, J.P., Brander, L., Maes, J., Hartje, V., 2013. Mapping ecosystem services' values: current practice and future prospects. Ecosyst. Serv. 4, 33–46. http://dx.doi.org/10.1016/j.ecoser.2013.02.003.
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., Zanden, E.H., Oudenhoven, A.P., Remme, R.P., Serna-Chavez, H.M., Groot, R.S., Opdam, P., 2014b. Ecosystem services as a contested concept: a synthesis of critique and counter-arguments. Conserv. Lett. 7, 514–523.
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. Ecosyst. Serv. http://dx.doi.org/10.1016/j.ecoser.2014.07.003 (in press).
Schulp, C.J.E., Burkhard, B., Maes, J., Van Vliet, J., Verburg, P.H., 2014. Uncertainties in ecosystem service maps: a comparison on the European scale. PLoS One 9, e109643.
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.
Shabman, L.A., Batie, S.S., 1978. Economic value of natural coastal wetlands: a critique. J. Coast. Zone Manag. 4, 231–247. http://dx.doi.org/10.1080/08920757809361777.
Statistics Netherlands, 2013a. Consumer prices, 5-12-2013, The Hague/Heerlen, the Netherlands. Accessed on: 16-12-2013. http://statline.cbs.nl/StatWeb/publication/?DM=SLNL&PA=71311ned&D1=0,4&D2=0&D3=90,103,116,129,142,155,168,181,194&HDR=G1,T&STB=G2&VW=T.
Statistics Netherlands, 2013b. Regional Accounts; Production Structure, The Hague/Heerlen, the Netherlands. Accessed on: 25-04-2014. http://statline.cbs.nl/Statweb/publication/?DM=SLEN&PA=81355eng&D1=a&D2=a&D3=17&D4=15&LA=EN&HDR=T&STB=G1,G2,G3&VW=T.
Statistics Netherlands, 2013c. Regionale Kerncijfers Nederland (Regional Key Figures the Netherlands). . Accessed on: 31-1-2014, http://statline.cbs.nl/StatWeb/publication/?DM=SLNL&PA=70072ned&D1=0,56-118&D2=16,76-78&D3=15-17&HDR=T&STB=G1,G2&VW=T (The Hague/Heerlen, the Netherlands).
Statistics Netherlands, 2013d. Statline Leisure and tourism, The Hague/Heerlen, the Netherlands. Accessed on: 10-12-2013. http://statline.cbs.nl/statweb/dome/?TH=5400&LA=en.
Statistics Netherlands, 2013e. Vegetables; yield and cultivated area per kind of vegetable, The Hague/Heerlen, the Netherlands. Accessed on: 05-12-2013. http://statline.cbs.nl/StatWeb/publication/?DM=SLEN&PA=37738eng&D1=a&D2=7,9,19,27,37,41&D3=12&LA=EN&HDR=T,G2&STB=G1&VW=T.
Statistics Netherlands, Kadaster, 2009. Wijk- en buurtkaart 2009 (District and neighbourhood map 2009). (Zwolle, the Netherlands).
Stoneham, G., O'Keefe, A., Eigenraam, M., Bain, D., 2012. Creating physical environmental asset accounts from markets for ecosystem conservation. Ecol. Econ. 82, 114–122. http://dx.doi.org/10.1016/j.ecolecon.2012.06.017.
Sumarga, E., Hein, L., 2014. Mapping ecosystem services for land use planning, the case of Central Kalimantan. Environ. Manag. 54, 84–97. http://dx.doi.org/10.1007/s00267-014-0282-2.

TEEB, 2010. The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations. Earthscan, London and Washington.
Thibodeau, F.R., Ostro, B.D., 1981. Economic analysis of wetland protection. J. Environ. Manag. 12, 19–30.
Tianhong, L., Wenkai, L., Zhenghan, Q., 2010. Variations in ecosystem service value in response to land use changes in Shenzhen. Ecol. Econ. 69, 1427–1435. http://dx.doi.org/10.1016/j.ecolecon.2008.05.018.
Troy, A., Wilson, M.A., 2006. Mapping ecosystem services: practical challenges and opportunities in linking GIS and value transfer. Ecol. Econ. 60, 435–449.
Turner, R.K., Morse-Jones, S., Fisher, B., 2010. Ecosystem valuation: a sequential decision support system and quality assessment issues. Ann. N. Y. Acad. Sci. 1185, 79–101. http://dx.doi.org/10.1111/j.1749-6632.2009.05280.x.
UK NEA, 2011. The UK National Ecosystem Assessment: Synthesis of the Key Findings. UNEP-WCMC, Cambridge.
UN, European Commission, International Monetary Fund, Organisation for Economic Co-operation and Development, World Bank, 2009. System of National Accounts 2008, New York.
UN, EC, FAO, OECD, World Bank, 2014a. System of Environmental-Economic Accounting 2012. Experimental Ecosystem Accounting, United Nations, New York, USA.
UN, European Commission, FAO, IMF, OECD, The World Bank, 2014b. System of Environmental-Economic Accounting. Central Framework, New York, USA.
United States Government, 2010. Technical Support Document, Social Cost of Carbon for Regulatory Impact Analysis, Under Executive Order 12866. Interagency Working Group on Social Cost of Carbon, United States.
United States Government, 2013. Technical Support Document, Technical Update of the Social Cost of Carbon for Regulatory Impact Analysis, Under Executive Order 12866. Interagency Working Group on Social Cost of Carbon, United States.
van Hout, G., personal communication. Indicative prices for hunting rights in Limburg of the Royal Dutch Hunters Association (KNJV).
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).
Veldhuizen, E., Graveland, C., Van den Bergen, D., Schenau, S., 2009. Valuation of Oil and Gas Reserves in the Netherlands 1990–2005. Statistics Netherlands, The Hague, the Netherlands.
Vewin, 2013. Water in Zicht 2012, Bedrijfsvergelijking drinkwatersector. Insight in water 2012, Company comparison for the drinking water sector. (The Hague, the Netherlands).
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, 2010. Jaarverslag 2010 / Beheerst ondernemerschap, grip op de toekomst [Annual report 2010 of the drinking water company of Limburg]. WML, Maastricht, the Netherlands.
Weber, J.-L., 2011. An experimental framework for ecosystem capital accounting in Europe. European Environmental Agency, Luxembourg.
ZKA Consultants & Planners, 2011. Toeristische trendrapportage Limburg 2010/2011 (Touristic trend report 2010/2011). (Breda, the Netherlands).