Experimental monetary valuation of ecosystem services and assets in the Netherlands

Technical background report

Contributing authors:

Edwin Horlings
Sjoerd Schenau
Lars Hein
Marjolein Lof
Linda de Jongh
Michael Polder

January 2020

CBS Den Haag
Henri Faasdreef 312
2492 JP The Hague
P.O. Box 24500
2490 HA The Hague
+31 70 337 38 00

www.cbs.nl

Wageningen University & Research
Droevendaalsesteeg 4
6708 PB Wageningen
P.O. Box 9101
6700 HB Wageningen
+31 317 480 100

www.wur.nl

Table of Contents

1. Introduction

This background report provides in-depth technical information and discussion on the experimental monetary valuation of ecosystem services and assets in the Netherlands. The aim of the overall study is to provide experimental monetary values for ecosystem services and ecosystem assets in the Netherlands and organize these values into an accounting framework.

This background report consists of two parts. The first part provides a description of the most relevant methods for the valuation of ecosystem services that were used in this study. The second part provides a detailed explanation of the methods and data sources used for the valuation of ten ecosystem services. Some parts of this explanation are also included in the main report.

2. Valuation methods

In this chapter, we describe the most relevant methods for the valuation of ecosystem services that were used in this study. There are basically four categories of valuation methods:

- Market-based methods
- Revealed preference methods
- Cost-based methods
- Stated preference methods

2.1 Market-based methods: Resource Rent

The resource rent is the economic rent that accrues in relation to environmental assets, including natural resources and ecosystems (SNA, 2008). The resource rent can be derived from the national accounts by deducting costs of labour, produced assets and intermediate inputs from market price of outputs (benefits). The use of this approach to pricing is commonly associated with provisioning services like those related to the outputs of agriculture, forestry, and the fishing industry, in particular where there are limited or no possibilities for using land leases and prices as an indicator of the price of ecosystem services (SEEA EEA, 5.79).

Background

David Ricardo defined economic rent as “that compensation, which is paid to the owner of the land for the use of its original and indestructible powers.” He added that “it is evident, that a portion only of the money annually to be paid for the improved farm, would be given for the original and indestructible powers of the soil” (Ricardo 1821, pp. 39-40). A more general formulation of Ricardian rent – applicable to all resources – is that it represents the surplus earned by a factor in excess of the amount needed to induce it do its work (Shepherd, 1970), that is, above the normal return to capital invested (Lange & Motinga, 1997).[1] The surplus is what the resource rent method aims to capture.

The formal analysis of resource rents grew out of the need to regulate the extraction of non-renewable natural resources, such as oil, gas, and coal (Hotelling, 1931; Land, 2008).[2] The ownership of non-renewable natural resources yields extraordinary rents. They are essentially free to the resource owner, having been created millions of years ago and needing only to be extracted. Non-renewable resources also tend to become scarcer as stocks diminish and use increases, and their markets tend towards monopoly or oligopoly.

Table 3.1.1 shows the standard derivation of gross operating surplus based on the SNA using measures of output, intermediate consumption, compensation of employees, and other taxes on and subsidies for production. The resource rent is calculated by deducting consumption of fixed capital, the return to produced assets and labour of self-employed persons from gross operating surplus.

[1] The modern definition defines economic rent as “any payment to a unit of a factor of production in an industry in equilibrium, which is in excess of the minimum amount necessary to keep that factor in its present occupation” (Boulding, 1941).

[2] Hotelling laid the foundations for the economics of non-renewable natural resources (“mine economics”). The challenge was to find to optimum level of exploitation from a private perspective (the mine owner) and a public perspective (consumers but also conservationists), and to find ways to regulate extraction. Hotelling’s work led to a stream of literature on resource rents and taxation of non-renewable resource extraction. He deliberately avoided the problem of renewable natural resources (“semi-replaceable assets”) that present different types of questions.

Table 3.1.1 Derivation of the resource rent

Output
less intermediate consumption
less compensation of employees
less other taxes on production
plus other subsidies on production
Equals gross operating surplus
less consumption of fixed capital (depreciation)
less return on produced assets
less labour of self-employed persons
Equals resource rent
= depletion + net return on environmental assets

Assumptions and limitations

Rent is inextricably linked to scarcity and market structure. Rent is paid because resources are limited in quantity and not uniform in quality (Ricardo, 1821, p. 41). Lange and Motinga (1997) attribute resource rents to the scarcity of the resource relative to demand for the resource on the world market. Under perfect competition rents tend to zero (Harvey, 1994; Torvik, 2002). The SEEA EEA also notes that “if there is open access to the resources and no charge by the owner for access, then the marginal unit resource rent will approach zero, thereby implying that the price of the ecosystem service is zero” (SEEA EEA, 5.82).

Resource rents can be used to calculate the value of ecosystem services under certain conditions. Valuing an ecosystem service using the resource rent method involves the assumption that the resource is “extracted or harvested in a sustainable way and that the owner of the resource seek[s] to maximize his or her resource rent.” (SEEA EEA, 5.81)

Finally, an ecosystem asset may produce a bundle of ecosystem services. The market data or SNA data used to calculate resource rents may only reflect a portion of that bundle. Also, market transactions may not capture the full economic value of ecosystem goods and services (King, Mazzotta & Markowitz, 2004).

2.2 Market-based methods: Rent prices

Rent is the income receivable by the owner of a natural resource (the lessor or landlord) for putting the natural resource at the disposal of another institutional unit (a lessee or tenant) for use of the natural resource in production (SNA, 2018, 7.109). In some cases, rent payments (or imputed rent payments) can be directly related to the provision of certain ecosystem services.

Background

There are several cases where rental prices can be used as a proxy for the value of ecosystem services. One example is the rent associated with agricultural land. Farmers may rent land for crop production or for livestock farming. These payments are directly related to the ecosystem services that are provided by the land. When the farmer owns the land, an imputed rent can be calculated from the value of the land.

Another example is the stumpage value for timber. Stumpage value is the value of standing timber prior to any value added by processing. Stumpage prices represent “the maximum amount potential concessionaires would pay for harvesting rights” with “full knowledge of the resource and competitive bidding” (Repetto et al., 1989). It can be argued that the stumpage value is not really a rent. Rent is usually associated with returns on a resource or factor of production that provides an income over time, whereas stumpage value is the price of a stock of a resource that can be harvested.

Assumptions and limitations
Rent payments take place when the user of the asset and the legal owner are not the same. When the user owns the asset there is no rent payment. When we apply the rent prices we assume that this price is also valid for the assets where no direct rent payment occurs.

2.3 Market-based methods: Production function method

The production function method – also known as the productivity method, the net factor income method, or the derived value method – is applied to estimate the economic value of ecosystems that contribute goods and services to the production of marketed products (King, Mazzotta & Markowitz, 2004, SEEA EEA, 5.98).

Background
Where natural resources are used to produce goods and services, the ecosystems that produce those resources contribute to benefits. Improvements in the quantity or quality of ecosystem services contribute in two ways (King, Mazzotta & Markowitz, 2004; www.openness-project.eu method factsheet on the production function approach):

1. raising the quality and/or lowering the price of the marketed product for consumers, resulting in a higher consumer surplus; and
2. increasing productivity and/or lowering production costs for the producers that use the ecosystem services, resulting in a higher producer surplus.

For example, water quality improvements generate additional producer surplus. They increase the catch of commercial fisheries and the incomes of fishermen (Farber, Costanza, Wilson, 2002), and they raise the productivity of irrigated agricultural crops. Water quality improvements also generate additional consumer surplus by lower the costs of drinking water purification (King, Mazzotta & Markowitz, 2004). Another example concerns insect pollination. Increased pollination leads to higher revenues and higher quality of agricultural crops that depend on pollination.

The production function method can be used to estimate direct use values for provisioning services, such as crop production, and indirect use values for regulating services, such as pollination and flood protection services. The contribution of ecosystem services to the production function can be deduced from observable market data. The production function method is similar to the resource rent method. It requires information on the functioning of the value chain and the production process, including the role of specific ecosystem services (SEEA EEA, 5.98), as well as information on other inputs and production factors in the production process.

Assumptions and limitations
The production function method only captures the contribution of ecosystem services to specific goods and services that are produced to be marketed commercially or that are enjoyed by humans while recreating (www.openness-project.eu; King, Mazzotta & Markowitz, 2004). It cannot capture non-economic values and it may underestimate economic values.

It is often difficult to map the entire value chain and understand the relationships between ecosystem services on the one hand and man-made inputs and produced assets on the other (www.openness-project.eu).

2.4 Cost-based methods: Replacement Costs

The replacement cost method estimates the value of an ecosystem service based on the costs that would be associated with mitigating actions if it would be lost (SEEA EEA, 5.84). it is often used to value regulating services. The core assumption of the replacement cost method is that a service can be replaced, i.e. that a man-made alternative can be developed.

Background

The replacement cost method can be used to value ecosystem services for which market data are not available. It is particularly relevant for regulating services (SEEA EEA, 5.85), such as air filtration and water purification. Replacement costs refer to industrial or natural substitutes that would need to be employed if the ecosystem service that needs be valued should be lost. Examples of such substitutes are:

- the construction of a water purification plant to replace the water filtration service of an ecosystem supplying groundwater to an aquifer used for drinking water (SEEA EEA, 5.84);
- using wetlands to reduce agricultural nitrogen pollution versus alternative methods that do not use wetlands (Byström, 2000);
- industrial treatment systems to replace natural waste treatment (Farber, Costanza, Wilson, 2002); or
- a retaining wall or levee to replace the flood protection services of a wetland (King, Mazzotta & Markowitz, 2004).

In the literature, replacement costs either relate to the ecosystem service or to the asset that provides the service. Replacing an ecosystem asset involves an investment (stock); replacing the ecosystem service involves annual expenditures (flow). Generally, replacement costs are estimated for a single ecosystem service rather than for the basket of ecosystem services that is produced by a given asset (SEEA EEA, 5.87, King, Mazzotta & Markowitz, 2004). Replacing an ecosystem asset consequently has implications for the entire basket of ecosystem services.

Assumptions and limitations

The core assumption is “that, if people incur costs to avoid damages caused by lost ecosystem services, or to replace the services of ecosystems, then those services must be worth at least what people paid to replace them.” (King, Mazzotta & Markowitz, 2004) A similar assumption is that the replacement ecosystem service provides the same utility as the existing ecosystem service and is the least-cost alternative (SEEA EEA, 5.84; EPA, 2009). This may not always be true.

It may be difficult to find perfect substitutes for ecosystems services. Some ecosystem services may be irreplaceable. For these services the replacement cost method cannot be used.

2.5 Cost-based methods: Avoided Damage

The avoided damage method estimate the value of ecosystem services based on the costs of the damages that would occur due to the loss of these services (Farber, Costanza, Wilson, 2002; De Groot et al., 2002). Similar to replacement costs, the focus will generally be on services provided by ecosystems that are lost due to human activity impacting on environmental condition, particularly through pollution. This method is often used to value regulating services.

Background

Ecosystem assets provide services that allow society to avoid costs and production losses. For example, trees filter particulate matter from the air. Particulate matter has an effect on health. By filtering the air, the ecosystem helps to lower air pollution-related health costs (Remme et al., 2015). The replacement cost method is not an option, since there is no least-cost alternative that can reasonably be proposed as a substitute.

The benefits are not provided to specific individuals (compare how croplands provide a direct service to farmers) and there is no exchange between willing sellers and buyers. The avoided damage method presumes that individuals are willing-to-pay to avoid the associated damages (SEEA EEA TR, 5.101). The monetary value of the ecosystem service is equal to the costs associated with the avoided damages (e.g. undesirable health consequences).

Application of the method requires careful consideration of the damages that should be valued (Remme et al. 2015, p. 120). Fisher, Bateman & Turner (2011) examine the damages and benefits of changes in CO_2 concentration to illustrate this point. They distinguish between market and non-market damages (for example, a reduction in forest cover affects timber production as well as recreation). While an increase in CO_2 concentration has a negative effect on agricultural production in some areas (due, for example, to soil moisture loss, pest increases, and a reduction in rainfall), other areas may benefit (for example, from increased precipitation and longer growing seasons). Damages and benefits of climate change are “spatially heterogeneous”.

The avoided damage method is particularly useful for regulating services such as erosion prevention, flood control, sedimentation control, air filtration, and carbon sequestration.

2.6 Revealed preference methods: Travel Costs

The travel cost method is used to calculate the monetary value of recreational ecosystem services. Recreation in nature requires physical access, which may require travel. The amounts consumers spend to visit a recreational site (e.g. transport, fuel, parking fees, bike rentals) are a proxy for their willingness to pay for recreational ecosystem services. In principle, this method reclassifies existing market-based transactions (travel expenditure) to environmental goods (ONS, 2014). Travel time and visiting time can be valued as well, although this value is usually seen as an welfare value (SEEA EEA, 5.103; ONS, 2014). The value of the recreational site is at least equal to what consumers are willing to pay to travel to the site (Farber, Costanza, Wilson, 2002).

Background

There are several types of travel cost method. In a welfare-based valuation estimate, the zonal or individual trvael cost method is used to calculate the consumer surplus of recreational visits to a site. With this approach, a demand curve can be prepared. If the travel cost is applied in a more narrow sense (as done by e.g. ONS), the costs made by visitors are seen as a reflection of their WTP, and this estimate is used to obtain a value estimate for people visiting the site. In this approach, a single-site method is used to estimate the value of one particular recreational site, such as a conservation area or a national park. The value of the ecosystem service is calculated using a recreational demand function based on the number of visitors, the distance they travelled, and the time and costs associated with a trip of that distance. In the multi-site method the value of multiple recreational sites is modelled using information on their properties, such as environmental attributes, recreational facilities, and accessibility (www.openness-project.eu Method factsheet on travel cost method).

Box 2.6.1 Eurostat statistics explained: glossary on tourism

Tourism means the activity of visitors taking a trip to a main destination outside their usual environment, for less than a year, for any main purpose, including business, leisure or other personal purpose, other than to be employed by a resident entity in the place visited.
The usual environment is the geographical area, though not necessarily a contiguous one, within which an individual conducts his regular life routines. Whether or not a visitor is within his or her usual environment is determined based on the crossing of administrative borders or the distance from the place of usual residence, the duration of the visit, the frequency of the visit, and the purpose of the visit.
A traveller is someone who moves between different geographic locations, for any purpose and any duration.
A visitor is a traveller taking a trip to a main destination outside his or her usual environment, for less than a year, for any main purpose (business, leisure or other personal purpose) other than to be employed by a resident entity in the country or place visited. These trips taken by visitors qualify as tourism trips. A visitor is classified as (1) a tourist (or overnight visitor), if his/her trip includes an overnight stay, or (2) a same-day visitor (or excursionist), if his/her trip does not include an overnight stay.

For the valuation of ecosystem services, we need to define nature recreation and tourism as discrete activities. According to Eurostat (Box 2.6.1), the difference between a tourist and a same-day visitor is that a tourist stays overnight. The nature-related value (e.g. rent) of hotels, B&Bs, campings, and other accommodations can be ascribed to tourism. However, in many other ways tourism and recreation overlap. Tourists and same-day visitors have lunch, drink coffee, rent bikes and canoes, take hikes through nature, and so on. Also, while Eurostat defines same-day visitors as travellers who take a trip outside their usual environment, the local population in the environment of an ecosystem can also use that ecosystem for nature recreation. In other words, we need to carefully delineate tourism and recreation – perhaps in the future examine if and how we can merge the two – and avoid double counting.

2.7 Revealed preference methods: Hedonic Pricing

The hedonic pricing method is used to determine the value consumers attach to one particular attribute of a (marketed) product in relation to all the product's other attributes. The most common application is the analysis of variations in housing prices in relation to physical attributes, properties of the neighbourhood, and the proximity to and quality of the natural environment (King, Mazzotta & Markowitz, 2004; SEEA EEA, 5.99).

Background

The hedonic pricing method has its roots in consumer theory.[3] The basic notion is that consumers assign a value to each of the properties of the good or service they purchase. The

[3] The classical references are: Lancaster, K. J. (1966). A new approach to consumer theory, Journal of Political Economy, vol. 74, pp. 132-157. Rosen, S. (1974). Hedonic prices and implicit markets: product differentiation in pure competition. Journal of political economy, 82(1), 34-55.

method captures revealed preferences as it is based on actual transactions and observed values.

The contribution of each property to the value of a house is derived using spatial regression analysis. House prices depend on each house's physical characteristics (e.g. size, number of rooms, energy efficiency, condition), characteristics of the neighbourhood (e.g. crime rate, schools, the value of other houses), and characteristics of the surrounding environment (e.g. proximity to nature, landscape view, water pollution, air pollution) (www.openness-project.eu Method factsheet on hedonic pricing).

The hedonic pricing method can be applied to other natural resources as well. Soil is an example:

> "Suppose we want to value the fertility of soil. Soil fertility is not a good that is bought and sold in a market, so we cannot just look up the price. However, farms are bought and sold, and we could collect data on farm prices, calculate the prices per hectare of the farmland, and then also collect data on the quality of the soil for these farms. Next we would correlate the land price per hectare with the quality of the soil to see how much the fertility of the soil adds to the price of the land. So indirectly we have estimated a price for soil fertility." (Heal, 2000, p. 26)

Assumptions and limitations
Hedonic pricing presumes that house buyers make a rational decision based on complete information on all relevant characteristics of a house. Some observed characteristics may not actually have an effect on the price. Buyers may not be aware of contaminated soil or noisy neighbours. They may not realise that the house is near a splendid nature conservation area. The model defines the house buyers' decision as a function of the parameters included in the model. Outside influences, such as changes in taxation or interest rates, that do affect the price may not have been captured in model parameters.

It is unrealistic to assume that all buyers have information on all relevant positive and negative characteristics of a house. However, few decisions are as well informed as buying a house.

The models link house prices to nearby nature areas to establish the amenity value ecosystems provide to house owners. Yet, ecosystems provide several functions, such as air filtration, flood protection, or recreation, that benefit house owners (www.openness-project.eu Method factsheet on hedonic pricing). If part of the price of a house is determined by a benefit provided by an ecosystem service, are we double counting when we estimate that benefit via hedonic pricing as well as using some other method (e.g. a travel cost approach to recreational services)?

Nature is heterogeneous. Ideally, hedonic pricing models should account for relevant distinctions. For example, different nature areas may have a different impact on house prices. Compare the attractiveness of living near an urban park, in a lane with trees, on a beach, or near a biodiversity hotspot. There is a difference between nearby nature (the value of being *near* nature) and landscape views (the value of *seeing* nature). Both provide a discrete amenity service, which may or may not relate to the same nature area.

2.8 Stated preference methods

Methods such as contingent valuation and choice experiments are applied to measure the stated preferences of a population (SEEA EEA; ONS, 2014). They can be used to identify willingness-to-pay for ecosystem services. These methods will not be used in this report for four reasons:

1. The results are not compatible with the SNA exchange value concept. Stated preference methods do not produce exchange values. They only relate to consumer surplus and not to producer surplus or production costs.
2. In surveys, respondents are asked about individual ecosystem services. However, each ecosystem asset may provide a basket of services. The sum of willingness-to-pay for each individual service is usually higher than the willingness-to-pay for the entire ecosystem asset.
3. Stated preference methods typically ask respondents to assign a value to hypothetical scenarios or options (e.g. Farber, Costanza, Wilson, 2002). The assumption is that respondents have complete information on all options and make a rational decision when choosing between options. However, the willingness-to-pay estimated in stated preference analysis is usually higher than the revealed preference.
4. Stated preference methods rely on detailed and comprehensive surveys and are therefore time consuming and resource intensive to set up and execute.

3. Technical details of the applied valuation techniques

In this chapter, we provide a more detailed explanation of the methods and data sources used for the valuation of the ten ecosystem services included in the main report. The ecosystem services are crop production, grass/fodder production, timber production, water filtration, air filtration, carbon sequestration in biomass, pollination, nature recreation, nature tourism, and amenity services. The annex on the amenity services of ecosystems is a full paper produced by Linda de Jongh, Michiel Daams, and Frans Sijtsma, which has been included in this report with permission of the authors.

3.1 Crop and fodder production

Three different valuation methods have been tested, namely the resource rent method, the rental price method, and the user cost of land (or land price) method.

3.1.1 The resource rent method

The resource rent method is a well-known and often used methodology for provisioning services (see section 2.1). The resource rent is calculated by subtracting all costs and normal returns from the total marketed output. To calculate the resource rent for crop production and fodder production, data were obtained from the SNA production and income accounts for ISIC 1 (agriculture). Within ISIC 1, data are available for five sub-activities, namely (a) arable farming (crop production), (b) livestock farming, (c) horticulture, (d) mixed farming, and (e) support activities to agriculture. Data for user costs of produced assets (consumption of fixed capital and return to produced assets) were obtained from the capital accounts. Also, the incomes of self-employed persons has to be taken into account, which are not recorded under the compensation of employees. For agriculture, this item is particularly important. The remuneration of self-employed has been calculated using data on the average wages in ISIC 1 and the employment (in fte) of self-employed people. No correction for non-agricultural output was made as the share is only around 1 percent of total output.

Table 3.1.1 Resource rent calculation for agriculture (ISIC 1)

million euros	2010	2011	2012	2013	2014	2015	2016	2017
Output	**28053**	**28604**	**29575**	**31091**	**29950**	**29594**	**29972**	**31893**
less intermediate consumption	17089	18726	19142	19611	18634	18117	17944	18721
less compensation of employees	2700	2776	2843	2891	2894	2881	2962	3071
less other taxes on production	390	397	384	346	332	349	357	357
plus other subsidies on production	-1088	-1220	-1171	-1210	-1095	-1070	-902	-916
Equals gross operating surplus	**8962**	**7925**	**8377**	**9453**	**9185**	**9317**	**9611**	**10660**
less consumption of fixed capital (depreciation)	3375	3453	3584	3744	3845	3940	4028	4056
less return on produced assets	1311	1895	1682	1661	1232	1404	1467	1494
less labour of self-employed persons	3312	3415	3468	3546	3544	3395	3492	3638
Equals resource rent	**964**	**-838**	**-357**	**502**	**564**	**578**	**624**	**1473**

On average, the resource rent for total agriculture equaled 160 million euro between 2010 and 2017. Accordingly, the resource rent is only 1.5 % of total agricultural output. The low average resource rent is caused by negative figures in 2011 and 2012. In these years net operating surplus was under pressure as the result of high intermediate costs. Overall, resource rents for agriculture are highly variable, with values fluctuating between -838 and 1,473 million euro. When we look at the different sub-activities within agriculture, the resource rent is low for arable farming (on average 163 million euro) and negative for livestock farming (on average -1,317 million euro). For horticulture the resource rent is positive (on average 1,249 million euro). Horticulture in the Netherlands is primarily done by greenhouse farming, which is much less dependent on land-based ecosystem services than arable and livestock farming.

3.1.2 User cost and rental prices

According to the SNA, land is a non-produced asset that, similar to produced assets, is a source of capital services. The value of land can be used to determine these capital services. The methodology to calculate these capital services, for agricultural land and the results are presented in this section.

The user cost of capital can be viewed as the price that the owner/user of a capital good 'pays to himself' for the service of using his own assets. In a perfect market, and defining away any transaction costs for supplying a rental, the user cost would take the same value as the rental price that the owner of a capital good could achieve if he rented out the asset during one period for use in production (OECD, 2009). Alternatively, the user cost corresponds to the marginal returns generated by the asset during one period of production.

The user costs of agricultural land (excluding any building or structures on the land) can be calculated as the real rate of return times the value of land (OECD, 2009):[4]

$$UC = (r - p)W$$

where r is the nominal rate of return, p is the general price change (inflation), and W is the land price. The rate of return for agricultural land in the Netherlands is annually calculated by Wageningen Research and is based on the return on risk-free fixed-income securities (Wageningen Research, 2018). It is calculated as:

- the three-year moving average of the real long-term capital market interest rate, being the effective return of the 10-year Euro Interest Rate Swap (Euro IRS) of December of each year (for example 0.985% in December 2016)
- minus the three-year moving average of inflation in the eurozone per December of each year (based on the HCIP, the harmonized European consumer price index, for example 0.349% in December 2016)
- plus a surcharge for management costs, taxes and risks of 1.25%.

For this study we applied a long-term average of the real rate of return (0.9%).

When the asset life is assumed to be limited to a maximum of T years, the value of UC should be corrected with a factor $a = 1/(1 - 1/(1 - r)^T)$ to reflect that the investment is to be earned back in a shorter period.[5] In our calculations, the rate of return is assumed to be equal across different types of land.

The value of agricultural land is included in the non-financial asset balance of the Dutch ntional accounts. Its value is based on market prices and the extent of agricultural land. Although the national accounts only include the total agricultural land value for the Netherlands, data for 14

[4] The user cost expression for land has a fairly simple form because there is no depreciation component and holding gains and losses are set to zero (OECD, 2009).

[5] Note that with an unlimited asset life T goes infinity, and $a = 1$. In technical terms, with an unlimited asset life, the net present value calculation is based on a perpetuity, whereas with a limited asset life, the calculation is an annuity.

different agricultural areas are available from Wageningen Economic Research. These agricultural areas are adjacent areas where the type of agriculture is similar. These subnational figures are adjusted to match the published national total in the national accounts. Moreover, using land prices by province, estimates of the land prices by type of land (grass and arable land) were derived.

Table 3.1.2. Value of provisioning services of agricultural land according to the user cost method

Groups of agricultural areas	2010	2011	2012	2013	2014	2015	2016	2017
GRASSLAND								
Bouwhoek en Hogeland	29	29	30	29	31	36	38	39
Veenkoloniën en Oldambt	33	31	36	40	40	47	44	45
Noordelijk Weidegebied	155	157	161	178	184	202	195	198
Oostelijk Veehouderijgebied	183	182	187	195	207	224	224	228
Centraal Veehouderijgebied	47	43	45	47	46	46	41	42
IJsselmeerpolders	22	22	28	29	29	35	34	34
Westelijk Holland	46	45	41	50	51	50	53	54
Waterland en Droogmakerijen	11	15	14	16	18	17	17	17
Hollands/Utrechts Weidegebied	58	63	63	66	65	70	71	72
Rivierengebied	49	49	53	53	53	58	57	58
Zuidwestelijk Akkerbouwgebied	36	36	27	37	36	40	40	42
Zuidwest-Brabant	15	17	18	18	18	18	17	17
Zuidelijk Veehouderijgebied	131	139	157	152	148	157	147	150
Zuid-Limburg	11	11	11	12	12	14	12	12
Netherlands	*825*	*838*	*871*	*923*	*937*	*1015*	*991*	*1008*
ARABLE LAND								
Bouwhoek en Hogeland	31	31	31	28	32	37	41	42
Veenkoloniën en Oldambt	59	57	64	68	71	78	86	91
Noordelijk Weidegebied	11	10	10	11	11	12	12	13
Oostelijk Veehouderijgebied	27	26	25	28	27	28	29	31
Centraal Veehouderijgebied	2	2	2	2	2	2	2	2
IJsselmeerpolders	75	77	88	87	84	95	90	94
Westelijk Holland	21	20	17	19	18	20	20	21
Waterland en Droogmakerijen	1	2	2	2	2	2	2	2
Hollands/Utrechts Weidegebied	1	1	1	1	1	1	1	1
Rivierengebied	10	10	10	11	10	10	10	10
Zuidwestelijk Akkerbouwgebied	129	122	131	139	133	129	149	152
Zuidwest-Brabant	8	9	9	9	9	9	9	10
Zuidelijk Veehouderijgebied	55	54	58	61	55	55	56	60
Zuid-Limburg	10	10	9	12	11	12	12	12
Netherlands	*439*	*432*	*458*	*479*	*466*	*490*	*519*	*538*
GRASSLAND AND ARABLE LAND								
Netherlands	*1265*	*1270*	*1329*	*1402*	*1403*	*1505*	*1510*	*1546*
HORTICULTURE								
Westelijk Holland (LG)	34	34	35	36	21	22	25	25
Other	80	81	86	77	74	76	81	84
Netherlands	*115*	*115*	*121*	*113*	*95*	*98*	*105*	*110*
TOTAL AGRICULTURAL LAND								
Netherlands	*1379*	*1385*	*1451*	*1515*	*1498*	*1603*	*1616*	*1655*

Notes: a) Figures are based on the long-term average of the real rate of return (0.9%) and an asset life of 100 years. b) 2010 and 2017 are based on extrapolations.

The user cost method depends on the rate of return and an assumption on the asset life. Table 3.1.3 presents the results of a sensitivity analysis with respect to these parameters. Putting the asset life to infinity decreases the value of the service up to about 40% for the baseline calculation using the average rate of return, but less so for higher rates of return. Increasing the rate of return increases the value of the services. If the discount rate were 4%, for example, the value of the ecosystem service would increase by a factor between 2 and 3 with respect to our baseline calculation. Finally, using the annual rates of return, rather than the average, shows that the figures become more susceptible to changes in the rate. This causes, for example, a substantial drop in the values towards the end of the sample period.

Table 3.1.3. Alternative calculations for the user cost method

	2010	2011	2012	2013	2014	2015	2016	2017
Asset life is infinite (perpetuity)								
annual rate of return	1164	740	826	1074	1178	985	606	621
average 2010-2017 (~0.9%)	828	832	871	910	900	963	970	994
4%	3594	3609	3780	3947	3903	4177	4210	4314
2%	1797	1804	1890	1974	1951	2088	2105	2157
Asset life 100 years (annuity)								
annual rate of return	1608	1326	1421	1624	1685	1617	1387	1421
average 2010-2017 (~0.9%)	1379	1385	1451	1515	1498	1603	1616	1655
4%	3666	3682	3857	4027	3982	4261	4295	4401
2%	2084	2093	2193	2290	2264	2423	2442	2502

Leases (rents) on land are a form of property income. They consist of the payments made to a land owner by a tenant for the use of the land over a specified period. Currently, around 30 % of agricultural land in the Netherlands is leased.[6] Over the last 30 years the percentage of leased land declined as a number of restrictive provisions in the legislation has made leasing land less attractive.

In the Netherlands rent prices are (partly) regulated by the government. Every year on the 1st of July, the government determines the highest allowable lease prices for agricultural land. These maximum lease prices are based on the five-year average yield of the land and are separately determined for 14 agricultural areas. A tenant and lessor can together arrange the lease price without compulsory intervention of the government, but only if the rent price does not exceed the highest allowable lease price. This regulation rules out the possibility that lease prices are affected by external market effects, for example the state of the property market. Moreover, the fact that maximum lease prices are linked to the actual yield of the land reinforces the suitability to use lease prices to value the contribution of the land to agricultural output.[7]

Based on rent prices and data on the extent of agricultural land the total value was calculated (cropland and grassland). For horticulture separate prices are available. Table 3.1.4 shows the results by agricultural area and over time.

The value of the ecosystem services based on rent prices varies between 1097 and 1452 million euro. Values are quite stable over time, and are only slightly lower than the values for the user cost approach based, when using average rates of return (see Table 3.1.2).[8] The highest values are found in Noordelijk Weidegebied, Oostelijk Veehouderijgebied, and Zuidelijk Veehouderijgebied. This mainly reflects the larger extent of agriculture is these areas.

[6] CBS-Landbouwtelling, Areaal (ha) cultuurgrond naar gebruikstitels, 2012, 2013, 2014, 2015, 2016 en 2017, https://www.cbs.nl/nl-nl/maatwerk/2018/12/areaal-cultuurgrond-naar-gebruikstitels-2012-2017.

[7] Agricultural land on which a lease contract rests has a lower value than land that is free of rent. Tenants can derive a number of rights from a lease contract that make the land on which the contract rests less attractive for a buying party.

[8] Note that the asset life and/or the rate of return in the user cost approach can be set in such way that the resulting service values are exactly equal to the lease prices.

Table 3.1.4 Value of provisioning services of agricultural land according to lease prices

Groups of agricultural areas	2010	2011	2012	2013	2014	2015	2016	2017
GRASSLAND								
Bouwhoek en Hogeland	25	29	30	29	27	31	35	28
Veenkoloniën en Oldambt	25	25	28	33	35	44	45	41
Noordelijk Weidegebied	185	181	176	202	189	216	256	223
Oostelijk Veehouderijgebied	165	159	142	147	137	166	208	191
Centraal Veehouderijgebied	36	34	33	28	26	33	37	31
IJsselmeerpolders	21	22	27	21	25	34	32	29
Westelijk Holland	26	25	29	27	33	39	45	37
Waterland en Droogmakerijen	14	11	10	10	8	11	13	10
Hollands/Utrechts Weidegebied	48	49	52	64	61	70	87	76
Rivierengebied	42	47	44	49	46	53	59	52
Zuidwestelijk Akkerbouwgebied	24	27	23	20	25	30	25	20
Zuidwest-Brabant	11	12	12	11	12	14	14	13
Zuidelijk Veehouderijgebied	116	107	101	104	101	118	133	122
Zuid-Limburg	8	10	11	10	12	13	13	12
Netherlands	*745*	*740*	*720*	*755*	*736*	*872*	*1003*	*885*
ARABLE LAND								
Bouwhoek en Hogeland	27	32	32	28	27	32	38	30
Veenkoloniën en Oldambt	46	46	51	56	62	72	88	83
Noordelijk Weidegebied	13	12	11	13	11	13	15	15
Oostelijk Veehouderijgebied	24	22	19	21	18	21	27	26
Centraal Veehouderijgebied	2	1	1	1	1	1	2	1
IJsselmeerpolders	72	79	85	62	74	91	85	81
Westelijk Holland	12	11	12	10	12	15	17	15
Waterland en Droogmakerijen	2	1	1	1	1	1	2	1
Hollands/Utrechts Weidegebied	1	1	1	1	1	1	1	1
Rivierengebied	8	9	9	10	8	9	10	9
Zuidwestelijk Akkerbouwgebied	85	93	112	75	91	98	93	71
Zuidwest-Brabant	6	6	6	6	6	7	8	8
Zuidelijk Veehouderijgebied	48	42	37	42	38	42	51	48
Zuid-Limburg	7	9	9	10	11	12	12	11
Netherlands	*352*	*366*	*386*	*336*	*361*	*415*	*448*	*399*
GRASSLAND AND ARABLE LAND								
Netherlands	*1097*	*1106*	*1106*	*1091*	*1097*	*1288*	*1452*	*1285*
HORTICULTURE								
Westelijk Holland (LG)	38	28	29	25	34	47	53	54
Other	102	81	64	51	50	57	78	125
Netherlands	*140*	*110*	*93*	*76*	*84*	*104*	*130*	*179*
TOTAL AGRICULTURAL LAND								
Netherlands	*1237*	*1215*	*1199*	*1167*	*1181*	*1392*	*1582*	*1463*

3.1.3 Discussion

Resource rent

The resource rent calculations show that the values for agricultural production (a) are relative low and sometimes negative, and (b) fluctuate significantly over the years. There are several explanations for this.

First, the estimate of the resource rent is highly sensitive to errors and uncertainties in the underlying information. The resource rent method calculates a residual, equal to the difference between total revenues and total costs plus normal returns. Errors and uncertainties pertaining to each individual item in the calculation accumulate and affect the overall estimate of the resource rent. The margin of error may be quite large, particularly if the residual is small compared to total revenues and costs. For agriculture, uncertainties may be particularly high for the estimated labour remuneration of self-employed persons, which is based on the assumption that wages for the self-employed are the same as for employees. The findings of the accounts may reflect that many farmers work long hours for a relatively low wage compared to alternative professions. When the wages of (self-employed) farmers are valued at a market rate, this results in a low resource rent.

Second, the resource rent method is sensitive to price changes. Any difference in price changes between revenues (agricultural products) and costs (wages, energy, materials) will affect the estimated resource rent. This is an important reason why the calculated resource rents for the Netherlands fluctuate significantly over the years. Moreover, such price changes are not necessarily related to the value of the services provided by land.

Third, it is difficult to calculate the resource rent separately for agricultural subsectors, such as crop production and fodder production. Calculations suggest high resource rents for horticulture and low or negative resource rents for arable farming and livestock farming. However, the underlying data probably are not suited for making calculations for individual agricultural activities.

Finally, a practical drawback of the resource rent method is that the national accounts data, needed for the calculations, are not available at a subnational level, thereby precluding any regional breakdowns.

User cost and rental prices
The value of land as recorded in the SNA asset account for the total economy can provide a useful comparison point with respect to the value of ecosystem assets (SEEA EEA, 6.66). The user costs of land (i.e. capital services) that can be derived from the land values may provide a measure for the value of ecosystem services. Similarly, lease prices for land can provide a useful indication of the value of ecosystem services. Here we will discuss whether these alternative methods can be used to determine the value of provisioning services for agriculture.

The value of agricultural land incorporates many ecosystem services, at least with regard to those ecosystem services contributing to benefits that are within the scope of the SNA production boundary. When a farmer buys or leases land to grow crops, the price reflects the potential to grow crops as a function of the ecosystem characteristics of the area, such as acreage, soil fertility, and hydrological properties. Therefore, the price (or lease price) of the land reflects the value of the relevant ecosystem services provided by the land.

However, there are some issues with directly relating the value of agricultural land to the value of the ecosystem asset. First, the value of agricultural land and associated user costs will not include all ecosystem services. Non-marketed ecosystem services, such as carbon sequestration, do not contribute to the production process of the owner of the land, and will therefore not be reflected in the value of the land. Other non-marketed services such as water regulation and pollination are (possibly) partly captured as they also provide benefits to the production process of the owner of the land. Therefore market-based valuation of land provides a lower bound of the value of agricultural land in this respect. Accordingly, the value of these services (and their contribution to the value of the ecosystem asset) is calculated separately in other parts of this report.

Secondly, SNA land values will incorporate, perhaps to a significant extent, the effect of the location of the land (e.g. access to markets, access to water sources), or a potential future destination of the land. Market-based land values may therefore also incorporate elements of value that are not dependent on ecosystems, such as the prospects for property development or the capitalization of farm subsidies (SEEA EEA TR, 6.49). This does not seem to pose a major problem for the Netherlands, as the highest land prices are found not in the most densely populated and urbanized areas, but in the Flevopolders and in Zeeland, which are areas with soil conditions most favourable for agricultural production. These conditions directly relate to the provisioning of ecosystem services. In addition, proximity to urban areas may also truly have an effect on the expected profits of agricultural output, for example due to access to markets. In other words, the higher demand for agricultural products in densely populated areas may also increase the value of the supplied ecosystem services. Moreover, in the Netherlands maximum lease prices are set by the government, based on historical profits (Wageningen Research, 2018). Therefore, lease prices are less vulnerable to market effects that are not related to the use of land for non-agricultural purposes. Finally, to the extent that disturbing market effects still play a role, it should be noted that these effects are likely to be very local. As the valuation in this report is done on a national or regional scale, these effects, as far as they are relevant, are expected to be averaged out.

Thirdly, agricultural land may also be used for the production of non-agricultural output, such as recreational services and tourism, and energy production (e.g. through windmills). In the Netherlands in 2015, the non-agricultural output of crop production and livestock farming was 1% and 1.5% respectively.[9] If non-agricultural production is small it may be neglected or a correction can be made.

In comparison with the large agricultural countries in the European Union, land and rental prices in the Netherlands are exceptionally high. This is the results of scarcity: as the Netherlands is a densely populated country, the total area is limited and the demand for land is high. On average, the Netherlands recorded the most expensive purchase price of agricultural land in the EU in 2016, with 63 thousand euro per hectare, as well as the highest lease price of agricultural land, namely 791 euro per hectare (Eurostat, 2018). Regional variation is quite strong: the highest average lease price is found in Flevoland, with 1536 euro almost twice the national average.
In short, we conclude that the resource rent method is not suitable to value the ecosystem services ‘crop production’ and ‘fodder production’ in the Netherlands. However, the user cost of agricultural land (as calculated from market land values) and rental prices can offer a good approximation for the ecosystem services contributing to crop production and livestock farming. Based on our analysis, and our assessment of advantages and disadvantages of each method (as indicated above) we propose to value crop production with the rental price method. On a per hectare basis, the value reaches the highest provincial maximum in Flevoland, at 791 euro per hectare per year. The value is low compared to that of some other services (e.g. recreation, water infiltration), which reflects that the high productivity of agriculture in the Netherlands is a function of, especially, the knowledge and capital intensity of Dutch farming practices, rather than the extent of the natural capital used by the farmers. Nevertheless, without this natural capital (soils, water), the turn-over of the Dutch farming sector would not be achieved.

---

[9] Output related to renting of buildings and equipment is excluded here as this output clearly is not related to the land.

3.2 Timber production

For timber production we have tested two valuation methods, namely the resource rent method and stumpage prices method.

The resource rent for timber production was calculated using data obtained from the SNA production and income accounts for ISIC 2 (forestry). The resource rent was calculated using the methodology described in section 3.1.1. The total output of the forestry industry in the Netherlands was about 254 million euro in 2015. Around 60 percent of total output is related to timber production.[10] The resource rent for timber production was calculated by multiplying the resource rent of the forestry industry with this percentage.

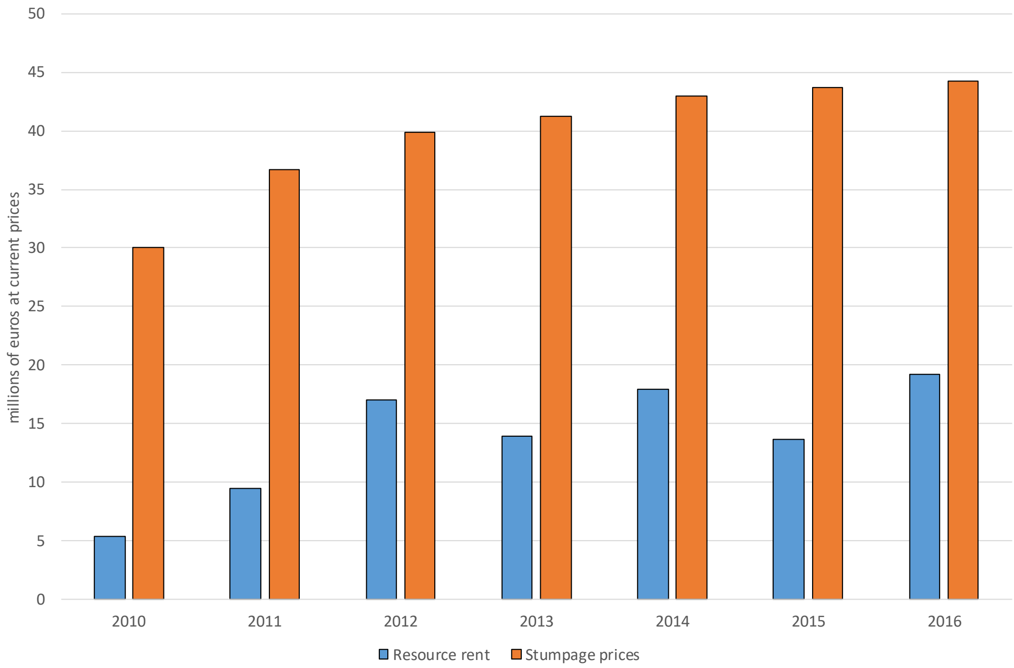
Stumpage prices (in Dutch 'hout op stam') are the prices paid per standing tree, including bark, for the right to harvest from a given land area. These are actual market prices that are paid and thus represent exchange values for the ecosystem service 'provision of timber'. Prices are collected and published by Wageningen Research.[11] Stumpage prices are available for different timber categories (pine, douglas, larix, other coniferous, willow, poplar and other deciduous wood). Here, an average stumpage price was taken. There is no further regionalization of the prices. The value of the ecosystem service timber production is calculated by multiplying the stumpage price (euros/m^3) with the total amount of wood harvested (m^3) as was determined for the physical supply and use tables (Statistics Netherlands, 2018).

The value of ecosystem service 'timber production' calculated using the stumpage prices increased from 30 million euros in 2010 to 44 million euros in 2016. The main reason for the increase is that prices for timber have increased (42 percent). The volume of timber harvested increased by only 4 percent. in 2016, the contribution of the ecosystem service to the total output of the forestry industry and to its value added were 17 and 37 percent respectively.

We conclude that for the ecosystem service timber production, stumpage prices are the preferred methodology to calculate monetary values. The stumpage price most directly reflects the value of the ecosystem service. In addition, the resource rent method is subject to considerable uncertainties on labour costs, costs of equipment, etc. The situation for this service is comparable to that in agriculture. The resource rent method also produces lower estimates than the stumpage price method. In addition, whereas total output of the forestry industry remained more or less constant between 2010 and 2016 (240-267 million euro) the calculated resource rent varies strongly between 5 million and 19 million euro. Stumpage prices are actual market prices paid to harvest wood and thus fully consistent with SNA exchange values.

[10] In addition to timber, the forestry sector also produces forest management services and recreation services.
[11] http://www.agrimatie.nl/Binternet_Bosbouw.aspx?ID=1005&Lang=0

Figure 4.2.1 Value of total provisioning services for timber production (millions of euros, current prices)



3.3 Water filtration

The resource rent method cannot be used to estimate the value of the ecosystem service 'water filtration' in the Netherlands. In the Netherlands, drinking water companies are semi-public (they have been privatised but their shares are owned by municipalities). Each drinking water company has a monopoly within its distribution area. Water prices are regulated by the Human Environment and Transport Inspectorate (*Inspectie voor Leefomgeving en Transport*; ILT) and monitored by the Authority for Consumers and Markets (*Autoriteit Consument en Markt*; ACM). Prices must reflect production costs as closely as possible, with provisions for investments and depreciation. This precludes the presence of economic rents. The resource rent for water companies is indeed negative in the Netherlands (see also Edens & Graveland, 2014).

Information on the total volume of drinking water supplied to households within distribution areas as well as the total volume of water abstracted from groundwater, riverbanks, dunes, and surface water in 2010-2016 was found in the drinking water statistics of VEWIN (the association of drinking water companies in the Netherlands). Total revenues, total costs, and production costs by cost category (taxes, depreciation, capital costs, and operating costs) were taken from the annual monitoring reports of the Authority for Consumers and Markets (*Autoriteit Consument en Markt*; ACM), from the statistical publications of VEWIN, and where necessary from the annual reports of drinking water companies.

The unit value of the ecosystem service that provides clean drinking water through the natural filtration and storage of groundwater is calculated by measuring the difference in unit production costs of companies that mainly extract groundwater and companies that mainly extract surface water. Each drinking water company has been classified as a groundwater, surface water or mixed-type company based on VEWIN (2012). This is the same classification as that of Remme et al. (2015). 'Groundwater companies' are companies that extract water from groundwater reservoirs or riverbank groundwater reservoirs. Production costs concern operating costs, costs of capital, and depreciation; taxes are excluded. Table 3.3.1 shows the average production costs per cubic metre per company. The unit value of the ecosystem service is the difference in production costs weighted by the volume of water supplied to households by each drinking water company (Table 3.3.2). Table 3.3.3 shows the cost difference between surface water and groundwater.

Table 3.3.1 Average production costs (euro per m^3, current prices)

drinking water company	2012	2013	2014	2015	2016
Brabant Water (gw)	0.94	0.97	0.99	0.99	0.95
Dunea (sw)	1.61	1.63	1.65	1.68	1.71
Evides Waterbedrijf (sw)	1.24	1.27	1.26	1.26	1.23
Oasen (gw)	1.53	1.49	1.45	1.41	1.25
PWN (sw)	1.61	1.67	1.82	1.70	1.72
Vitens (gw)	1.08	1.10	1.10	1.09	1.01
Waternet (sw)	1.62	1.62	1.61	1.59	1.61
Waterbedrijf Groningen (gw)	1.04	1.03	1.04	1.07	1.01
WMD Drinkwater (gw)	1.01	1.07	1.08	1.08	1.00
WML (mix)	1.43	1.44	1.47	1.43	1.33

Note: Company type was determined based on Remme et al. (2015) and VEWIN (2013) for the year 2012; (gw) = groundwater company, (sw) = surface water company, (mix) = groundwater company.

Table 3.3.2 Total amount of drinking water delivered (thousand m^3)

drinking water company	2012	2013	2014	2015	2016
Brabant Water (gw)	164928	165550	164000	168000	170000
Dunea (sw)	71281	71405	71660	72862	72894
Evides Waterbedrijf (sw)	155300	154700	154000	154000	156000
Oasen (gw)	45411	46012	45000	46000	47000
PWN (sw)	104288	105603	99000	101000	101000
Vitens (gw)	329800	329700	326000	331000	337000
Waternet (sw)	65143	66000	67000	68000	69000
Waterbedrijf Groningen (gw)	41569	42364	42000	42000	43000
WMD Drinkwater (gw)	28093	28125	27898	28000	29000
WML (mix)	71474	71393	71000	72000	71000

Note: Company type was determined based on Remme et al. (2015) and VEWIN (2013) for the year 2012; (gw) = groundwater company, (sw) = surface water company, (mix) = groundwater company.

Table 3.3.3. Weighted average production costs of drinking water by company type in euro per m^3 at current prices)

	2012	2013	2014	2015	2016
groundwater companies	1.07	1.09	1.09	1.09	1.01
surface water companies	1.47	1.50	1.53	1.51	1.51
cost difference between surface water and groundwater	0.40	0.41	0.44	0.42	0.49

Note: Company type was determined based on Remme et al. (2015) and VEWIN (2013) for the year 2012. The unit value of the ecosystem service is the difference in production costs weighted by the volume of water supplied to households by each drinking water company.

The difference in production costs between groundwater companies and surface water companies is partly the result of technical differences between companies. The drinking water industry identifies three main drivers of production costs: production type (groundwater versus surface water), volume of water supply per connection, and network complexity (VEWIN, 2012). We have tested the relationship between production costs and these three drivers, using different variants and adjusting for multicollinearity. The unstandardised coefficient for the percentage groundwater equals the cost difference attributable to groundwater. Using the sum of phreatic and riverbank groundwater, the unstandardised coefficient was -.293 in 2012, -.340 in 2014, and -.407 in 2016. This compares well with the cost difference calculated in Table 4.3.3: the parameters are lower, as they should be, but still account for 73, 77 and 83 percent of our estimated cost difference.

By classifying each drinking water company as ‘groundwater’, ‘surface water’ or ‘mixed-type’, we ignore the ‘abstraction portfolio’ of each company. Six or seven companies rely entirely on groundwater or on surface water. The other four use a combination of sources. And the importance of different sources changes over time. Classifying drinking water companies is an understandable decision – there is no information on production costs per type of source – but does involve a loss of information.

Boyd and Banzhaf (2007, p. 617) suggest that governments are not capable of properly defining units of trade and compensation for ecosystem services that are public goods and for which there is no market. The drinking water industry in the Netherlands is closely monitored and from a cost perspective, the government has very precise information. For a replacement cost approach, Boyd and Banzhaf’s concern does not apply.

3.4 Air filtration

Particulate pollution covers a broad spectrum of pollutant types that permeate the atmosphere. Particulate matter is commonly referred to by size groupings: coarse and fine. PM_{10} includes particles up to < 10 µm in aerodynamic diameter, whereas $PM_{2.5}$ only represents the smallest particles (<2.5 µm). The health effects of $PM_{2.5}$ particles are larger compared to the fraction between 2.5 and 10 µm because these smaller particles penetrate deeper into the lungs. In recent years it has become clear that $PM_{2.5}$ particles pose a higher health risk because these smaller particles penetrate deeper into the lungs. Data from epidemiological studies indicates that long term exposure to $PM_{2.5}$ can increase both human morbidity and human mortality risks (Kunzli et al., 2000). Therefore, in the monetary account we focus on the smaller particles. Trees and other vegetation play an important role in the reduction of air pollution (Powe and Willis, 2004). To value the ecosystem service air filtration (or air quality regulation) an avoided damage cost approach was used, with $PM_{2.5}$ capture by forests and other vegetation as biophysical indicator.

Particulate matter is captured through deposition on leaf and bark surfaces. The process of deposition depends on tree type and meteorological conditions (Powe and Willis, 2004). Deposition varies depending on density of the foliage and leaf form (the leaf area index, LAI). As $PM_{2.5}$ is a fraction of PM_{10}, capture of $PM_{2.5}$ by vegetation (e.g. forests, natural grasslands, cropland, heath) is modelled using the equations for PM_{10} capture by Powe and Willis (2004) (see Statistics Netherlands & WUR, 2018). Input maps for the capture model are the Ecosystem Type map with a 10m spatial grain, and the yearly average $PM_{2.5}$ in µg m^3 (based on 24 hour daily averages) for 2015 on a 1000 m spatial grain.

Table 3.4.1 Input data

Name dataset	Data type	Source
Ecosystem Type map	Spatial data, raster 10m	Statistics Netherlands
Yearly average $PM_{2.5}$ 2015	Spatial data, raster 1000m	RIVM
Yearly average PM_{10} 2015	Spatial data, raster 1000m	RIVM
Neighbourhood statistics 2015 (CBS buurt)	Spatial data, polygon	Statistics Netherlands
Age-dependent mortality	Statistics	Statline
Life expectancy 2015	Statistics	Statline
PM_{10} capture parameters	Reference values	Powe and Willis (2014)
Tree phenology	Observations	Nature Today
Rain days	Statistics	Environmental Data Compendium

$PM_{2.5}$ capture was estimated using the following equation (as in Powe and Willis, 2004):

ABSORPTION = SURFACE * PERIOD * FLUX

where:

ABSORBTION = dry pollution deposition on vegetation cover ($PM_{2.5}$ *capture* in µg m^{-2})

SURFACE = area of land considered (A in m^2) * surface area index (S in m^2 per m^2 of ground area)

PERIOD = period of analysis (t in s (i.e. 31536000 s)) * proportion of dry days per year (p_{dry})* proportion of in-leaf days per year ($p_{on\text{-}leaf}$)

FLUX = deposition velocity (v_d in m s^{-1}) * ambient $PM_{2.5}$ concentration ($C_{PM2.5}$ in µg m^{-3})

or,

$PM_{2.5}$ $capture_{on\text{-}leaf}$ (in *kg* ha^{-1}) = $A * S_{on\text{-}leaf} * t * p_{dry} * p_{on\text{-}leaf} * v_d * (10^{-9}/10^{-4}) * C_{PM2.5}$
$PM_{2.5}$ $capture_{off\text{-}leaf}$ (in *kg* ha^{-1}) = $A * S_{off\text{-}leaf} * t * p_{dry} * (1 - p_{on\text{-}leaf}) * v_d * (10^{-9}/10^{-4}) * C_{PM2.5}$

We take:

$M_{on\text{-}leaf} = A * S_{on\text{-}leaf} * t * p_{dry} * p_{on\text{-}leaf} * v_d * (10^{-9}/10^{-4})*0.5$

and:

$M_{off\text{-}leaf} = A * S_{off\text{-}leaf} * t * p_{dry} * (1 - p_{on\text{-}leaf}) * v_d * (10^{-9}/10^{-4})*0.5$.

where the factor 0.5 denotes the resuspension rate of particles coming back to the atmosphere (Zinke, 1967).

For each vegetated ecosystem type we add these multiplication factors $M_{year} = M_{on\text{-}leaf} + M_{off\text{-}leaf}$ to calculate $PM_{2.5}$ *capture* in *kg* ha^{-1} based on ambient $PM_{2.5}$ concentration, $C_{PM2.5}$ in µg m^{-3}. The deposition velocities, the surface area index and multiplication factors per ecosystem type with vegetation cover are summarized in table 3.4.2. Values for deposition velocity are based on

Powe and Willis (2004). However, for coniferous forest, we used a similar LAI as for in-leaf deciduous forest based on a meta-analysis by Asner, Scurlock, and Hicke (2003).

Data on phenology of emergence of leaves until the end of leaf fall of trees in the Netherlands (Nature Today, 2017) was used to estimate the proportion of in-leaf days for deciduous forests, on average deciduous trees were on-leaf from mid-April to mid-November (i.e. $p_{on\text{-}eaf}$ = 7/12). Data on average number of rain days with ≥ 1.0 mm precipitation (Environmental Data Compendium, 2017) were used to calculate the proportion of dry days. The average number of rain days in the Netherlands in the period between 1981 and 2010 was 131 (i.e. p_{dry}=234/365).

The above model calculates $PM_{2.5}$ capture in kg per hectare per year. However, the effect of particulate matter on health is mostly derived from epidemiological studies where frequency of the health outcome is related to the level of exposure in $\mu g/m^3$. Therefore, the capture in kg $PM_{2.5}$ per hectare per year needs to be converted to a reduction in annual mean concentration $PM_{2.5}$ in $\mu g/m^3$. Assuming a boundary layer of 2000m with mixing during the day, and converting capture per year to capture per day, results in a conversion factor of 0.137 from kg/hectare/year capture to a reduction of the daily mean ambient $PM_{2.5}$ concentration in $\mu g/m^3$. Some impact categories of the health costs of air pollution are related to $PM_{2.5}$ and others are related to PM_{10}, for the latter the local fraction of $PM_{2.5}$ in PM_{10} is used to calculate the local reduction in PM_{10} concentration.

Table 3.4.2 Deposition velocities (*$m\ s^{-1}$*), the surface area index (*$m^2\ m^{-2}$*) and yearly multiplication factors to calculate yearly PM_{10} capture per ecosystem type with vegetation cover.

	Deposition velocity		Surface area		
Ecosystem type	On-leaf	Off-leaf	On-leaf	Off-leaf	M_{year}
Non-perennial plants	0.0010	0.0010	2	1.5	0.18
Perennial plants	0.0010	0.0010	2	1.5	0.18
Meadows	0.0010	0.0010	2	1.5	0.18
Hedgerows	0.0010	0.0010	2	1.5	0.18
Dunes with permanent vegetation					1.69[1]
Deciduous forest	0.0050	0.0014	6	1.7	1.87
Coniferous forest	0.0050	0.0050	6	6	3.03
Mixed forest					2.45[2]
Heath land	0.0010	0.0010	2	1.5	0.18
Fresh water wetlands	0.0010	0.0010	2	1.5	0.18
Natural grassland	0.0010	0.0010	2	1.5	0.18
Public green space	0.0010	0.0010	2	1.5	0.18
Other unpaved terrain	0.0010	0.0010	2	1.5	0.18
River flood basin	0.0010	0.0010	2	1.5	0.18

[1] Dunes with permanent vegetation is calculated as the average of the factors for coniferous forest, deciduous forest and other vegetation.
[2] Mixed forest is calculated as the average of the factors for coniferous forest and deciduous forest.

Several studies use a 1km² resolution to calculate the effect of vegetation on air pollution reduction (Remme et al., 2015; Powe and Willis, 2004; Oosterbaan et al., 2006). Furthermore, the yearly average $PM_{2.5}$ concentration in $\mu g\ m^3$ (based on 24 hour daily averages) was also available at a 1km² spatial resolution (RIVM, 2013). The reduction in $PM_{2.5}$ concentration due to vegetation was first calculated at a 10m spatial resolution based on the Ecosystem Types map and based on that map an average reduction in $PM_{2.5}$ concentration per km² was calculated. This was combined with population distribution data at a local level (CBS buurt 2015, polygon map), that included population density, age distribution and number of females and males per neighbourhood. These data were aggregated to a km2 raster. Based on these data 6 maps for population density per km^2 were generated. One for the total population, which was used to calculate the avoided health costs. Five maps were generated for the population density in the age categories 0-14, 15-24, 25-44, 45-64 and 65 and older, as used to calculate avoided mortality costs. Furthermore, one map for the fraction of females per km^2 was generated.

To value air filtration, we compare three measures, all of which represent a measure for avoided damage. The first involves valuing the avoided health costs, similar to Remme et al. (2015) for the Dutch province Limburg. The second approach involves valuing avoided health costs and avoided costs of mortality, using the value of a statistical life year (VOLY). Strictly speaking, this is a welfare measure which is not compatible with SEEA ecosystem accounting, but it is added to obtain a first indication of how exchange values and welfare-based values differ. There are different approaches to estimate the VOLY from a survey on mean willingness-to-pay (WTP) asking people for their WTP to increase their (statistical) life expectancy by a given period. Here we have used WTP for an increasing life expectancy with 3 months based on a study of Desaigues et al., 2011. This study is often used in cost-benefit analysis (CE Delft, 2017). The third approach also values both avoided health effects and avoided mortality, but mortality is valued with the maximum societal revenue VOLY (MSR-VOLY) as proposed by Hein, Roberts and Gonzalez (2016). This is a potentially relevant indicator to capture the benefit of clean air in a natural capital accounting approach. The MSR-VOLY represents the VOLY that would theoretically apply in case there was 'market' for clean air, based on the demand curve for clean air and assuming that there are no costs related to supplying the ecosystem service. It corresponds with the Simulated Exchange Value proposed by Caparros et al. 2015, and is a type of posited exchange value as stipulated in the UK SEEA accounting work (White et al., 2015).

Health costs

In the first approach, the avoided increase in $PM_{2.5}$ concentration was valued based on avoided air pollution related health costs. Similar to Remme et al. (2015) health impact categories were used that were identified in a study by Preiss et al. (2008) on health costs of air pollution in the European Union. In line with the SEEA-EEA approach, categories that were based on direct costs were included while categories that include components of consumer surplus were excluded. Damage costs for a person due to an increase of 1 $\mu g/m^3$ $PM_{2.5}$ was estimated at about 7.36 euros per person (2015 €) and damage cost for a person due to an increase of 1 $\mu g/m^3$ PM_{10} was estimated at 2.68 euros per person (2015 €) (Table 3.4.3). For the costs related to PM_{10}, we correct the reduction in $PM_{2.5}$ concentration with the fraction of $PM_{2.5}$ in PM_{10}. In 2015, this fraction ranges from 0.30 to 0.75, with a mean of 0.58. The value of avoided exposure to 1 $\mu g/m^3$ $PM_{2.5}$ per person is in this case about 12.00 euros per person (2015 €).

MSR – VOLY and mean VOLY

In the second approach, the avoided increase in $PM_{2.5}$ concentration was not only valued based on air pollution related health costs (as above) but in addition air pollution mortality was taken into account and valued based on the maximum societal revenue. This approach is potentially also aligned with the natural capital accounting approach. It represents the point where the multiplication of a WTP and the number of people expressing at least this WTP is at its maximum (Hein et al., 2016). We used an estimate for the MSR-based on the mean and median value of a WTP survey in several EU countries and Switzerland by Desaigues et al. (2011) in which people were asked to value a three-month increase in life expectancy.

Table 3.4.3 Health impact categories resulting from $PM_{2.5}$ and PM_{10} concentration change, risk group, age group, concentration response functions, physical impact on a person, monetary value per unit and external costs in €(2015) per person per µg/m^3. Risk group, age group and concentration response function are adapted from Preiss et al. (2008), unless stated otherwise. Age group factor is adjusted for the Netherlands (2015) and monetary value is corrected for inflation.

Health impact categories	Risk group	Risk group factor	Age group	Age group factor	Concentration response function	Physical impact per person per µg/m³	Unit	Monetary value per unit (2015€)	External cost € per person per µg/m³
Nett restricted activity days	all	1	total	1	9.59×10^{-3}	9.59×10^{-3}	days	145.48	1.40
Work loss days	all	1	15-65	0.652	2.07×10^{-2}	1.35×10^{-2}	days	330.13	4.46
Minor restricted activity days	all	1	18-65	0.617	5.77×10^{-2}	3.56×10^{-2}	days	42.52	1.51
Total in € per person per µg/m³ $PM_{2.5}$									7.36
New cases of chronic bronchitis	all	1	≥27	0.684	2.65×10^{-5}	1.81×10^{-5}	cases	24840[a]	0.45
Respiratory hospital admissions	all	1	total	1	7.03×10^{-6}	7.03×10^{-6}	cases	2845[b]	0.02
Cardiac hospital admissions	all	1	total	1	4.34×10^{-6}	4.34×10^{-6}	cases	2845[b]	0.0123
Medication use/brochodilator use child	children meeting PEACE criteria – EU average	0.2	5 - 14	0.114	1.80×10^{-2}	4.10×10^{-4}	cases	1.12	0.000459
Medication use/brochodilator use adult	astmathics	0.045	≥20	0.775	9.12×10^{-2}	3.18×10^{-3}	cases	1.12	0.00356
Lower respiratory symptoms (adult)	symptomatic adults	0.3	adults	0.779	1.30×10^{-1}	3.04×10^{-2}	days	42.52	1.29
Lower respiratory symptoms (child)	all	1	5 - 14	0.114	1.86×10^{-1}	2.12×10^{-2}	days	42.52	0.90
Total in € per person per µg/m³ PM_{10}									2.68

[a] Adapted from Remme et al. (2015).
[b] Adapted from "passantenprijzen DBC zorgproducten".

The damage costs based on the MSR for a statistical life year lost due to an increase in $PM_{2.5}$ are estimated at 16,270 euros (2015 €). The mean value of an avoided exposure to 1 µg/m3 per person is in this case about 10.10 (2015 €). This value, however, depends on the spatial distribution of the reduction in $PM_{2.5}$ and the spatial distribution of the population and the spatial age distribution.

The mean VOLY is based on a WTP survey and thus is subject to a consumer surplus. However, as it is the preferred metric in cost-benefit analysis to analyse and inform on monetary benefits resulting from improvements in air quality (Hein et al., 2016; Desaigues et al., 2011) we include it in the analysis to determine the band-with of the avoided damage costs associated with air filtration. The damage costs based on the mean VOLY for a statistical life year lost due to an increase in $PM_{2.5}$ are estimated at 49,607 euros (2015 €). The value of an avoided exposure to 1 µg/m3 per person is in this case about 30.90 (2015 €). This value, however, depends on the spatial distribution of the reduction in $PM_{2.5}$ and the spatial distribution of the population and the spatial age distribution.

Both mean VOLY and MSR-VOLY value a statistical life year. Data from epidemiological studies indicate that long-term exposure to $PM_{2.5}$ can increase age-specific mortality by about 6% per 10 µg/m^3 (Carey et al., 2013). Avoided statistical life years lost were modelled based on spatial data per neighbourhood regarding population density, age and gender supplemented with statistics on age-dependant mortality and life expectancy. The spatial maps contained spatial data on population size in the age categories 0-14, 15-24, 25-44, 45-64 and 65 and older and number of females per neighbourhood (maps: based on CBS buurt, Statistics Netherland). The density was first sampled at a 10x10m raster and then aggregated to a 1kmx1km raster, using the mean. Age-specific mortality for the above age categories is calculated based on mean mortality rates per 5-year age class and relative abundance of the 5-year age class in the above age categories (data: adapted from Statline A and B, table 3.4.4). Furthermore, age-dependent life expectancy of males and females was available up to an age of 80 year, age-specific mortality rates were used to estimate life expectancy up to 100 years (data: adapted from statline C, table 3.4.4).

Table 3.4.4 Age specific mortality rate and life expectancy of females and males, as used in the calculations for 2011, 2013 and 2015.

	Mortality rate			Life expectancy (months)					
				Female			Male		
Age category	2011	2013	2015	2011	2013	2015	2011	2013	2015
0 – 14	0.00013	0.00013	0.00013	910.0	912.6	913.2	866.9	869.5	873.0
15 – 24	0.00024	0.00024	0.00024	773.0	766.3	766.4	721.4	723.7	726.7
25 – 44	0.00059	0.00059	0.00059	587.4	589.8	590.5	546.9	548.9	552.7
45 – 64	0.00401	0.00401	0.00401	364.8	366.2	366.0	326.1	327.9	330.9
65+	0.03644	0.03600	0.03542	166.4	167.7	168.3	140.1	141.8	144.1

3.5 Carbon sequestration in biomass

There are two approaches to estimating the economic value of carbon sequestration, both of which represent a measure for avoided damage. The first approach involves the social cost of carbon; the second approach concerns the carbon price of policy targets.

Social cost of carbon
The social cost of carbon (SCC) represents the monetary value in the present of damages that occur in the future as a result of an additional ton of carbon emissions in a given year. The SCC is derived from estimates of the total economic effects of climate change (Tol, 2009). The value of the ecosystem service measures the marginal benefits associated with avoided damages owing to the sequestration of carbon emissions in a given year (IPCC, 2007; IAWG, 2016).[12] Remme et al. (2015) used the SCC to estimate the value of carbon sequestration in the Dutch province of Limburg.

Damages and benefits are modelled using increasingly complex models (see, e.g., Nordhaus, 2017). Estimates of the SCC intend to capture all future benefits and damages associated with climate change, including, for example, changes in agricultural productivity, property damage from increased flood risk, effects on human health, reduced costs for heating, increased costs for air conditioning, and so on (EPA, 2016).

Following Remme et al., we first use the American SC-CO2 to estimate the value of carbon sequestration. The American SCC estimates are based on integrated assessment models that project future damages until the year 2300.[13] The SC-CO2 represents the net present value of damages that occur in the future – between the year of emission and the end of the model run (the year 2300) – as a result of one ton of carbon dioxide emissions in a given year.[14] Conversely, the SC-CO2 can also be seen to represent the future damage *avoided* as a result of one ton of carbon sequestration in a given year (IAWG, 2016).

Figure 3.5.1 presents the social cost of carbon per ton of carbon in current prices and at different discount rates. Starting point was the most recent update of the SC-CO2, which concerns the social cost of carbon in US dollars of 2007 per metric ton of CO_2 (US Government 2016). The Interagency Working Group on Social Cost of Greenhouse Gases produces five-yearly estimates of the SC-CO2. Values for the intervening years have been interpolated for the purposes of this report. The time series of SC-CO2s at constant prices of 2007 have been converted to current prices using the GDP price index for the Netherlands.[15] Values in US dollars have been converted to euros using purchasing power parities.[16] Values per metric ton of CO_2 have been converted to values per ton of carbon using the ratio between the molecular weights of CO_2 and C.[17]

---

[12] Global warming can be associated with localised benefits, such as lower energy consumption for heating and higher agricultural productivity. Most of the future effects of CO_2 emissions in the present are, however, damaging.
[13] The SC-CO2 is based on three integrated assessment models (IAMs): the Climate Framework for Uncertainty, Negotiation, and Distribution (FUND) model, developed by Richard Tol; the Policy Analysis of the Greenhouse Effect (PAGE) model, developed by Chris Hope; and the Dynamic Integrated Climate and Economy (DICE) model, developed by William Nordhaus.
[14] “The SC-CO2 is meant to be a comprehensive estimate of climate change damages and includes, among other things, changes in net agricultural productivity, human health, property damages from increased flood risk and changes in energy system costs, such as reduced costs for heating and increased costs for air conditioning. However, it does not currently include all important damages. The IPCC Fifth Assessment report observed that SC-CO2 estimates omit various impacts that would likely increase damages.” (EPA, 2016)
[15] http://statline.cbs.nl/Statweb/publication/?DM=SLNL&PA=84087ned&D1=63-83,114-123,169-183&D2=5-22&VW=T
[16] OECD, Purchasing Power Parities for GDP and related indicators.
[17] The molecular weight of CO_2 is 44,0095. The molecular weight of C is 12,0107. The conversion rate is 3,6642.

Figure 3.5.1. The social cost of carbon in 2010-2017 at different discount rates

	5%	3%	2.5%
2010	32	100	162
2011	32	101	162
2012	33	105	166
2013	33	106	167
2014	34	111	173
2015	35	116	180
2016	36	120	185
2017	37	125	191

Sources: US Government (2016). OECD Purchasing Power Parities for GDP and related indicators. CBS Statline GDP price index.

Carbon price

The second approach is to calculate the costs of achieving a policy-defined target of reduction in CO_2 emissions. This calculation produces a carbon price. By valuing carbon sequestration in biomass at this carbon price, we estimate in monetary terms the contribution of ecosystems to achieving the policy target.

The carbon price is used in ecosystem accounting in the UK.[18] This carbon price is specific to the UK policy environment (Department of Energy & Climate Change, 2011) and is not relevant to the Netherlands. The Netherlands Environmental Assessment Agency (PBL) and the Netherlands Bureau for Economic Policy Analysis (CPB) have calculated the Dutch carbon price.

Aalbers, Renes & Romijn (2017) argue that the SCC does not adequately measure willingness-to-pay for a unit reduction in CO_2 emissions or the costs of preventing damage due to CO_2 emissions. In addition, the SCC is estimated based on complex models with very long time horizons and high uncertainty. The result is that estimates of the SCC vary widely (Tol, 2009; IPCC, 2007).[19] A pragmatic reason is that willingness-to-pay and the SCC are unknown, whereas marginal prevention costs can be known and are therefore the best alternative.

The efficient price of CO_2 is the price at which the necessary cumulative reduction in CO_2 emissions is achieved at the lowest costs (PBL, 2018). There are different scenarios for what will be necessary. PBL and CPB distinguish three scenarios: a high-reduction scenario, a low-reduction scenario, and a two-degree temperature increase scenario.

According to PBL (2018), by 2050 the efficient price is equal to the ETS price of a ton of CO_2 emissions, as all economic actors fall under the ETS. In the high-reduction scenario, the efficient price is 160 euros per ton of CO_2 in 2050; in the low-reduction scenario it is 40 euros per ton; and in the two-degree policy target it ranges from 200 to 1000 euros per ton. The discounted net present value is calculated using a discount rate of 3.5%.[20] For the year 2015, the corresponding figures are 48 euros for the high-reduction scenario, 12 euros for the low-reduction scenario, and 60 to 300 euros for the two-degree policy target. Figure 3.5.2 presents the net present value per ton of carbon (C) in 2010 thru 2017.

[18] The UK carbon price is a simulated exchange value based on “the marginal abatement cost (supply) of meeting UK policy targets (demand).” (ONS, 2018) The carbon price relates to the non-trade sector (i.e. emissions outside the EU Emission Trading System); the traded carbon price (within the EU ETS) should be used to value changes in emissions in the traded sector. The UK has reduction targets for both sectors.

[19] See Tol (2009) for a review of uncertainty and variation in SCC estimates. “Studies with a lower discount rate have higher estimates and much greater uncertainties.” (Tol, 2005)

[20] Normally, this discount rate is 3%. PBL/CPB argue that a higher discount rate is warranted because the growth potential of economies in Southern and Eastern Europe is higher (Aalbers, Renes & Romijn, 2017, p. 10).

Figure 3.5.2. The efficient carbon price for the Netherlands: net present value per ton of carbon in 2010-2017

	high-reduction scenario	low-reduction scenario	2°-scenario lower boundary	2°-scenario upper boundary
2010	148	37	185	925
2011	153	38	192	958
2012	159	40	198	991
2013	164	41	205	1026
2014	170	42	212	1062
2015	176	44	220	1099
2016	182	46	228	1138
2017	188	47	235	1177

Source: PBL (2018).

The estimates of the value of the ecosystem service carbon sequestration in biomass is presented in Figures 3.5.3 and 3.5.4. In Figure 3.5.3 the amounts of carbon sequestered are valued at the social cost of carbon. In Figure 3.5.4 they are valued at the efficient carbon price for the Netherlands.

Figure 4.5.3. The value of the ecosystem service carbon sequestration in biomass at the social cost of carbon, 2010-2017

	Total carbon sequestration (kton C/yr)	SCC at 5% discount rate, mln euro	SCC at 3% discount rate, mln euro	SCC at 2,5% discount rate, mln euro
2010	975	31,6	97,9	157,8
2011	975	31,6	98,9	158,4
2012	975	32,2	101,9	162,1
2013	975	32,2	103,0	162,6
2014	975	33,3	107,8	168,9
2015	975	34,5	112,8	175,4
2016	975	35,3	117,1	180,3
2017	975	36,3	121,9	185,7

Table 4.5.4. The value of the ecosystem service carbon sequestration in biomass at the carbon price, 2010-2017

	high-reduction scenario, mln euro	low-reduction scenario, mln euro	2°-scenario lower boundary, mln euro	2°-scenario upper boundary, mln euro
2010	144,4	36,1	180,5	902,3
2011	149,4	37,4	186,8	933,9
2012	154,7	38,7	193,3	966,6
2013	160,1	40,0	200,1	1000,4
2014	165,7	41,4	207,1	1035,5
2015	171,5	42,9	214,3	1071,7
2016	177,5	44,4	221,8	1109,2
2017	183,7	45,9	229,6	1148,0

The carbon price seems preferable over the social cost of carbon:

- The social cost of carbon is less dependent on policy than the efficient carbon price. The efficient carbon price is dependent on the level of political ambition. Higher ambitions raise the carbon price. The efficient carbon price is the optimum where marginal willingness-to-pay for CO_2 emission reduction is equal to marginal unit prevention costs. However, the SCC also depends upon policies: it is likely that the marginal costs of carbon vary with the amount of carbon being emitted as a function of the implementation of climate policies.

- The efficient carbon price has lower uncertainty than the SCC. The SCC is estimated based on complex models that predict CO_2 emissions, climate change, and output into the far future. These models are incomplete and highly uncertain – in particular how they deal with low probability-high impact events (CE Delft,, 2017; Tol, 2009; IPCC, 2007). The result is an enormous variation in estimates and no instrument for prioritisation. The efficient carbon price is calculated based on a present-day assessment of the costs of reduction measures (i.e. prevention costs).
- Aalbers, Renes and Romijn (2017) argue that the SCC does not adequately measure willingness-to-pay for a unit reduction in CO_2 emissions or the costs of preventing damage due to CO_2 emissions. Willingness-to-pay and the SCC are unknown. On the other hand, marginal prevention costs (used to calculate the carbon price) can be known.
- The efficient carbon price is more viable and more relevant than the SCC. The American SC-CO2 that was used by Remme et al. is produced by a government working group that was recently disbanded by president Trump. It remains to be seen if the SC-CO2 will be estimated again. The Dutch carbon price is calculated specifically for the Netherlands and will most likely be updated.

Which scenario is most relevant? Remme et al. opted for the SCC at a 5 percent discount rate, producing a conservative (i.e. low) net present value. CE Delft take the high-reduction scenario as the central scenario (between low and high). This puts the value of the ecosystem service of carbon sequestration in biomass at 177.5 million euros in 2015.

Figures 3.5.3 and 3.5.4 show that neither method produces a single estimate. The results vary by an order of magnitude, from tens of millions to hundreds of millions. The main reason for this variation is that calculations of the SCC and the carbon price involve wide-ranging assumptions about the future – about the development of CO_2 emissions, the introduction of technologies that reduce emissions, and the effects on climate change and economic growth.

The SCC and the carbon price were never intended to be used in ecosystem accounting. National (ecosystem) accounting calls for a single estimate for each variable. The carbon price is used to inform cost-benefit analyses for public policy. The American SC-CO2 "should not be thought of as a single number or even a range of numbers; rather it should be thought of more broadly as a process that yields updated estimates of those numbers and ranges. The ultimate goal of the process is scientific credibility, public acceptance, and political and legal viability." (Metcalf & Stock, 2017).

3.6 Pollination

About 75% of the leading global food crops species depend on animal pollination (Klein et al., 2007). Together these crop species produce 35% of the global production volume. Without animal pollination the production of these crops will be up to 90% lower. The majority of crops are most effectively pollinated by bees (Klein et al., 2007; Ricketts et al., 2008). Pollinator visits not only move outcross pollen among individuals but also increase the total amount of pollen deposited on flower stigmas, both of which are known to increase quantity and quality of crops. Animal pollination reduces production loss, thereby increasing production.

Most studies on natural pollination have focussed on wild bees and bumble bees. Historically, pollination demand was fulfilled by wild pollinators that live in the agricultural landscape. Nowadays, beekeepers often place hives with cultivated honey bees, *Apis mellifera*, close to pollination demanding crops. Many crops, however, are also effectively pollinated by wild bees. A field study carried out by Alterra (de Groot et al., 2015) showed that for Elstar, a common apple species in the Netherlands, up to 60% of the flowers were pollinated by wild bees despite a lower occurrence of wild bees in comparison to honey bees. In the pollination service provided by ecosystems in our model, pollination by wild organisms such as wild bees, bumble bees, butterflies and hoverflies was considered. Managed honey bees were excluded. Wild pollinators require sufficient resources in the agricultural landscape. These resources include suitable nesting habitats (e.g. tree cavities, or suitable soil substrate) as well as sufficient floral resources (i.e. pollen and nectar). Bees are central place-foragers, meaning that they return to their nest site after foraging. The availability of nesting habitats close to agricultural fields is critical for bee-pollinated crops (Ricketts et al., 2006).

Table 3.6.1. Input data

Name dataset	Data type	Source
Ecosystem Type map	Spatial data	Statistics Netherlands
Basisregistratie Gewaspercelen 2015, 2016 and 2017	Spatial data	RVO.nl
Pollination requirements	Table	Klein et al. (2007)
Habitat suitability for pollinators	Table	Kennedy et al. (2013)
Standard yield	Table	Wageningen Economic Research
Yield apples and pears	Table	Statline

Crops differ in pollination requirements. Klein et al. (2007) divided crops into five classes, depending on their degree of production dependence (Table 3.6.2). These are used to assign pollination demand to crops in the Netherlands based on the spatial location of crops in 2015, 2016 and 2017 ("Basisregistratie Gewaspercelen", RVO.nl, 2017) (Table 3.6.3).

Table 3.6.2 Classes for dependence of crops on pollination, based on yield loss in absence of pollinators. Between brackets the class mean that is used to generate maps of pollination demand of crops.

Degree of dependence	Production reduction in absence of pollinators	Crops
Essential	> 90%	Courgette, pumpkin
Great	40% - 90% (65)	Raspberries, blackberries, other berries, annual fruit cultivation, perennial fruit cultivation (e.g. pear, apple, cherry) and summer rapeseed, and winter rapeseed
Modest	10% - 40% (25)	Strawberries, eggplant, redcurrants, blackcurrants, summer oilseed rape, winter oilseed rape, and sunflower
Little	0% -10% (5)	Other beans and other oilseeds
No increase	no reduction (0)	Other crops

Source: Klein et al. (2007).

Table 3.6.3 Look-up table for pollination demand of pollination dependent crops and crop production per hectare, given per pollination dependent crop in the basic registration of crops in the Netherlands (*Basisregistratie Gewaspercelen*)

Crop code	Description	Pollination demand (%)	Region	Production euro/ha 2015	Production euro/ha 2016
242	Beans (bruine bonen)	5	NL	2,805*	2,256*
311	Field beans	25	NL	895	895
258	Alfalfa	5	NL	900	900
515	Sunflower	25	NL	1,440	1,440
663	Lupine	5	NL	1,260	1,260
664	Rapeseed	65	NL	905*	590*
665	Soybeans	5	NL	1,270	1,270
666	Linseed	5	NL	1,270	1,270
853	Broad beans (tuinbonen, droog)	5	NL	2,380	2,380
854	Broad beans (tuinbonen, groen)	5	NL	2,980	2,980
1922	Oilseed rape, winter	25	NL	905*	590*
1923	Oilseed rape, summer	25	NL	905*	590*
2700	Strawberry, open field, multiplication	25	NL	102,500	102,500
2701	Strawberry, open field, waiting bed	25	NL	41,100	41,100
2702	Strawberry, open field, production	25	NL	55,900	55,900
2703	Strawberry, open field, seed	25	NL	102,500	102,500
2704	Strawberry, rack, multiplication	25	NL	133,500	133,500
2705	Strawberry, rack, waiting bed	25	NL	53,400	53,400
2706	Strawberry, rack, production	25	NL	72,700	72,700
2707	Strawberry, rack, seed	25	NL	133,500	133,500
2731	Gherkin, production	65	NL	19,100	19,100
2732	Gherkin, seed	65	NL	5,380	5,380
2723	Courgette, production	95	NL	28,800	28,800
2724	Courgette, seed	95	NL	5,380	5,380
2729	Cucumber, production	65	NL	13,700	13,700
2730	Cucumber, seed	65	NL	5,380	5,380
2733	Melon, production	95	NL	13,700	13,700
2734	Melon, seed	95	NL	5,380	5,380
2735	Pumpkin, production	95	NL	6,340	6,340
2736	Pumpkin, seed	95	NL	5,380	5,380
2779	Stem green bean, production	5	NL	2,320	2,320
2780	Stem green bean, seed	5	NL	5,380	5,380
2781	String beans, production	5	NL	13,700	13,700
2782	String beans, seed	5	NL	5,380	5,380
1095	Apple, new	65	Mid	15,105*	14,690*
1095	Apple, new	65	North	14,940*	16,110*
1095	Apple, new	65	West	15,840*	16,525*
1095	Apple, new	65	South	12,935*	11,505*
1096	Apple	65	Mid	15,105*	14,690*
1096	Apple	65	North	14,940*	16,110*
1096	Apple	65	West	15,840*	16,525*
1096	Apple	65	South	12,935*	11,505*
1097	Pear, new	65	Mid	33,830*	29,625*
1097	Pear, new	65	North	36,555*	37,385*
1097	Pear, new	65	West	32,165*	31,530*
1097	Pear, new	65	South	17,345*	23,035*
1098	Pear	65	Mid	33,830*	29,625*
1098	Pear	65	North	36,555*	37,385*
1098	Pear	65	West	32,165*	31,530*
1098	Pear	65	South	17,345*	23,035*

Crop code	Description	Pollination demand (%)	Region	Production euro/ha 2015	Production euro/ha 2016
1100	Stone fruits (including peach)	65	NL	35,000	35,000
1869	Blueberry	65	NL	58,700	58,700
1870	Plum	65	NL	17,100	17,100
1872	Cherry, sour	65	NL	6,340	6,340
1873	Blackberry	25	NL	3,160	3,160
1874	Other small fruits	25	NL	35,200	35,200
2325	Red berry	25	NL	62,500	62,500
2326	Raspberries	65	NL	130,000	130,000
2327	Blackberries	65	NL	177,000	177,000
2328	Cherry, sweet	65	NL	35,000	35,000

Notes: Pollination demand is based on the classification used for the pollination requirements of Klein et al. (2007). Crop production is based on production statistics produced by Statline per year when available (marked with *). Remaining data is based on the standard production as calculated by the LEI (Everdingen and Wisman, 2017) based on average production in 5 consecutive years (i.e. 2011 - 1015). For apples and pears Statline produces statistics on production per region of the Netherlands.

Table 3.6.4 Look-up table for an indicator of combined nesting suitability and floral resource availability for ecosystem types in the Netherlands (0 to 100 scale, with 100 indicating most suitable and 0 unsuitable)

Code	Ecosystem type	Total nesting and floral suitability
1	Non-perennial plants	(30)
2	Perennial plants	(58)
3	Greenhouses	0
4	Meadows / pasture	26*
5	Hedgerows	80
6	Farmyard and barns	0
11	Dunes with permanent vegetation	80
12	Active coastal dunes	26
21	Deciduous forest	89
22	Coniferous forest	44
23	Mixed forest	66
24	Heath lands	100
25	Inland dunes	26
26	Fresh water wetlands	48
27	Natural grassland	80
28	Public green space	41
29	Other unpaved terrain	41
31	River flood basin	48
32	Tidal salt marshes	36
41-48	Paved and built-up area	0
51	Sea	0
52	Lakes and ponds	0
53	Rivers and streams	0

*lower than indicated for the EU, because Dutch meadows are relatively species poor.
Source: (based on Kennedy et al., 2013).

Compared to the ecosystem service that we calculated for 2013, we no longer include: 175 flower cultivation, 176 flower bulbs and tubers. These groups were also not included in Klein et al., 2007. Furthermore, in 2015, 2016 and 2017 the classification of the crops changed and adistinction was made between fruit types and open field vegetables. For the latter, this made it possible to make a distinction between vegetables that depend on pollination and vegetables that do not depend on pollination.

Ecosystems differ in the suitability for pollinators, because there are differences in the presence of tree cavities or suitable substrates for nesting, and differences in the availability and suitability of floral resources (Kennedy et al., 2013). We used indicators for total nesting and floral resource availability for the suitability of the ecosystem types (Table 3.6.4). These indicators were based on a meta-analysis of 39 studies that was conducted by Kennedy et al. (2013). Note that private gardens, whether in rural (farmyards and barns) or in urban areas (residential areas), are set to zero suitability due to the lack of information and the spatial heterogeneity of all 'paved and built- up areas'.

The maps for the pollination account are generated based on the spatial location of crops that require pollination (Basisregistratie Gewaspercelen 2015, 2016 and 2017; RVO.nl, 2017) and the spatial location of ecosystems that are suitable for pollinators on the Ecosystem Type map 2013. Different species of pollinators move across different distances. Large pollinators such as bumble bees forage over long distances (up to 1750 m; Walther-Hellwig and Frankl, 2000), while small pollinators such as solitary bees, forage over shorter distances (up to several hundred meter). We generate the suitability and demand maps for all natural pollinators.

Ricketts et al. (2006) found in their meta-analysis on 13 studies in temperate biomes that visitation rates of pollinators declined to half its maximum at 1308 m distance between the nesting sites and the crop. The optimal model for visitation rate (scaled 0-1, with 1 being the maximum visitation rate) in temperate bioms is exp(-0.00053*d*) where *d*, is distance between the nesting sites and the crop in meters. This model includes both species that forage over long distances and species that remain close to their nesting site. In the model, pollination service is assigned to the nearest suitable habitat. Pollinators leave their nesting sites to forage in the surrounding landscape. We assume that pollinators from all suitable habitats in the local landscape contribute to pollination. To obtain the relative visitation rate (scaled 0-100) in a crop in map unit *c* (Lonsdorf et al., 2009) we calculate

$$v_c = \sum_{h=1}^{H} S_h \frac{e^{-0.00053 d_{hc}}}{\sum e^{-0.00053 d}}$$

where S_h represents the relative pollinator abundance (scaled 0-100, where 100 marks maximum suitability) in map unit *h* (based on the suitability for nesting and foraging for pollinators of the habitat in map unit *h*), d_{hc} is the distance between map unit *h* and the crop in map unit *c*. Pollination is then a function of the relative visitation rate:

$$P_c = f(v_c)$$

Rader et al. (2016) find a relationship between visitation variation and fruit set variation, based on 39 studies. Variation in fruit set was measured in 14 crops. They found that both bees (not including honey bees) and non-bee pollinators had a positive relationship between fruit set and pollination. Furthermore, studies show that often more pollen are deposited than needed for successful fruit set, 10 to 40 times more pollen have been reported in Sáez et al. (2014) and Pfister et al. (2017). Therefore, we model the function of pollination based on visitation rate as $P_c = 5v_c$, v_c between 0 and 20 and 100 for $v_c \geq 20$. This is a starting assumption, there can be differences between crops, but we do not take that into account here.

To calculate avoided reduction in crop production due to the presence of pollinators, we combine the pollination map that is based on the Ecosystem Type map and spatial relationships of visitation rates by pollinators with the potential production reduction map (based on standard yield in euro per hectare for each pollination dependent crop, table 3.6.3), using the following equation:

"Avoided production reduction" = "potential production reduction" * ("pollination")/100

The avoided production reduction represents the use of the pollination service by the crops. Next, we calculate the contribution (supply) of the ecosystems to the avoided production reduction, APR_h,

$$APR_h = \sum_{c=1}^{C} APR_c \frac{\sum_{h=1}^{H} S_h \frac{e^{-0.00053 d_{ch}}}{\sum e^{-0.00053 d}}}{\sum_{h=1}^{H} S_h}$$

where APR_c is the avoided production loss in the crop in map unit *c* (in euro/hectare), d_{ch} is the distance between the crop in map unit *c* and the ecosystem in map unit *h*. The relative contribution of all ecosystems in a 6 km square around the crop is weighted by the sum of the relative pollinator abundances, S_h. Contribution to avoided production loss in crop fields by the ecosystem in map unit *h* is based on all crop fields that require pollination in a 6 km square around map unit *h*. This is calculated for all map units that contain an ecosystem that provides pollination.

Discussion
Expert knowledge on habitat suitability was used to approximate pollinator potential. Local data on pollinator presence and abundance is needed to accurately calculate pollination potential. These data are not available for the Netherlands. As a result, the pollination model only takes into account effects of land use on pollination potential.

The resource rent calculations for crop production show that the values for agricultural production are relative low and often negative, and fluctuate significantly over the years (see Table 3.1.1). If the resource rent method would be used, the contribution of the pollination service would therefore be very small or negative, while it is generally acknowledged that pollination is an important service for agriculture.

The ecosystem service crop production is also calculated based on rent prices of agricultural land. The user cost of agricultural land (as calculated from market land values) and rental prices can offer a good approximation for the ecosystem services contributing to crop production. However, using this method would require additional assumptions on the fraction of the rental price that is based on the contribution of pollination from the surrounding landscape. As a result of scarcity, land and rent prices in the Netherlands are very high. Therefore, the level of land and rent prices might not reflect the effects of the local landscape, but rather the high demand of a limited area.

3.7 Nature recreation and nature tourism

We have tested two methods to value the ecosystem services nature recreation and nature tourism: the resource rent method and the consumption expenditure-based method.

3.7.1 Resource rent method

The first step for applying the resource rent method to nature related tourism is to delineate the tourism sector. Using the definition from the tourism satellite accounts (TSA), tourism characteristic activities are those that typically produce tourism characteristic products (TSA, 2008). Note that the tourism sector as defined by the TSA covers both tourism activities (i.e. including overnight stays) and recreational activities (i.e. excluding overnight stays). First, we selected from the Dutch TSA data for (a) internal tourism (i.e. tourism/recreational activities taking place in the Netherlands by residents and non-residents) and (b) tourism for leisure (i.e. excluding business travel). The tourism sector covers many different industries (see table 3.7.1). In the Netherlands, the most important industries are Accommodation (ISIC 55), Food and beverages service activities (ISIC 56), Retail trade (ISIC 47), and Sports, amusement and recreation activities (ISIC 93). Note that (internal) tourism related activities usually cover only part of the total of an ISIC category.

Table 3.7.1 Dutch tourism sector (internal tourism only, excluding business travel), 2017

ISIC	Industry	Production million euro	Value added million euro	Employment 1000 FTE	Percentage share in total output
19	Refineries	1,381	92	0	6
47	Retail trade	5,697	3,357	74	14
49	Land transport	2,523	1,045	19	9
55	Accommodation	4,858	2,500	44	66
56	Food and beverages service activities	13,150	6,414	136	64
90	Creative, arts and entertainment activities	993	490	8	21
92	Gambling and betting activities	1,013	693	7	49
93	Sports activities and amusement and recreation activities	3,696	1,538	28	56
94	Activities of membership organizations	739	412	6	11
	Other	3,113	1,397	23	
	TOTAL	37,163	17,939	344	

Next, we selected only those industries that were most important (with regard to total output) and that were potentially related to nature to calculate the resource rent for nature related tourism. Retail trade was excluded because ´shopping´ during the holiday or recreational trip is probably not directly induced by nature. Also, transport related activities (refineries, land transport) were omitted. Accordingly, we selected the following three industries: Accommodation (ISIC 55), Food and beverages service activities (ISIC 56) and Sports, amusement and recreation activities (ISIC 93). All these activities may directly profit from being located in a nature area. For the selected industries data were obtained from the SNA production and income accounts. The resource rent was calculated using the methodology described in section 3.1 (Table 3.7.2).

The calculated resource rent equals ca. 1.4 billion euro between 2010 and 2014, and increased rapidly to 4.1 billion euro in 2017. When looking at the different industries, resource rents were relative low for Accommodation and Sports/recreation services and high for Food/beverages services activities (Table 3.7.3).

Table 3.7.2 Resource rent calculation for ISIC 55, ISIC 56 and ISIC 93

million euros	2010	2011	2012	2013	2014	2015	2016	2017
Output	**24773**	**25754**	**26145**	**26467**	**27610**	**30520**	**32369**	**34345**
Intermediate consumption	13507	13935	14119	14264	14587	16088	16859	17864
Compensation of employees	6522	6746	6883	7013	7310	7977	8435	8901
Other taxes on production	231	253	253	257	275	317	314	327
Other subsidies on production	-107	-93	-76	-59	-36	-200	-202	-262
Equals Gross operating surplus	**4620**	**4913**	**4966**	**4992**	**5474**	**6338**	**6963**	**7515**
Less consumption of fixed capital	1167	1172	1186	1195	1202	1162	1184	1228
Less return to produced assets	276	355	328	361	286	284	279	255
less compensation self emlpoyed	1871	1917	1991	1993	2126	1697	1784	1896
Equals Resource rent	**1306**	**1470**	**1461**	**1443**	**1860**	**3195**	**3716**	**4135**

Table 3.7.3 Resource rent calculation for different industries, 2015

million euros	Accomodation	Food/ beverages	Sports/ recreation services
Output	**6311**	**17975**	**6234**
Intermediate consumption	3240	9218	3630
Compensation of employees	1606	4698	1673
Other taxes on production	210	80	27
Other subsidies on production	-11	-23	-166
Equals Gross operating surplus	**1266**	**4002**	**1070**
Less consumption of fixed capital	414	457	291
Less return to produced assets	120	91	73
less compensation self emlpoyed	140	1139	418
Equals Resource rent	**592**	**2315**	**288**

In the final step, we have to determine what part of the resource rent is related to nature. For this we used the expenditure data from the tourism statistics (see next section). By relating total expenditure for nature related recreational/tourism activities to total expenditures we calculated the share for the different industries (Table 3.7.4). The resource rent related to nature tourism/recreation was c. 300 million euro in 2015.

Table 3.7.4 Calculation of the resource rent activities for tourism related to nature, 2015

	Accomodation	Food/ beverages	Sports/ recreation services	Total
Resource rent: total ISIC (million euros)	592	2315	288	3195
Percentage nature-related	30%	4%	10%	
Resource rent: related to nature (million euros)	178	93	29	299

3.7.2 Consumption expenditure related to nature tourism and recreation

In this section we discuss the methodology to calculate consumer expenditure related to (a) nature recreation, (b) nature tourism by residents, and (c) nature tourism by non-residents.

A. Nature recreation

Recreational activities include all leisure related activities for which one is away from home for two hours or longer, but that do not include an overnight stay. Data on recreational expenditures were obtained from the Dutch recreation statistics, which in turn are based on surveys. These statistics provide information on the different kinds of expenditures and the types of recreational activities. In order to delineate nature related recreation, we selected the following types of recreational activities:

1. *Outdoor recreation*, which includes hiking for pleasure, cycling for pleasure, outdoor recreation near water, outdoor recreation in nature, touring around in the countryside by car or motor, and trips by tour boats.
2. *Water sports*, which include outdoor swimming, canoeing, rowing, surfing, fishing, sailing, and boat trips.
3. *Outdoor sports* (excluding water sports), which include jogging/running, mountain biking, horse riding, hiking (as a sport), and cycle racing.

These are all activities that take place outdoors and are thus fully depend on the outdoor environment.

With respect to expenditure categories we included admission fees, travel costs, costs for food and drinks and other related costs.

Figure 3.7.5 Expenditure on nature related recreational activities, 2015

	Admission fees etcetera	Travel costs	Food, drinks, etcetera	Other costs	Total
Hiking	30.8	506.6	176.2	30.8	744.4
Cycling	25.1	65.7	185.5	25.1	301.4
Other outdoor recreation	87.8	821.5	333.2	300.5	1543.0
Water sports	80.3	78.5	34.2	122.0	315.0
Outdoor sports	119.1	196.9	152.0	501.5	969.5
Total	343.2	1669.1	881.1	980.0	3873.4

Total expenditure on nature related recreational activities was 3.9 billion euro in 2015. This is only 11% of total recreational expenditure. This relative low percentage is because expenditure in shops (c. 11 billion in total) is not considered part of nature related expenditure. Most nature related recreational expenditure is related to hiking and other outdoor recreation. Travel costs are half of the total expenditure related to nature.

Data on expenditures by region (province) are not yet available. Accordingly, in order to allocate expenditure to provinces the regional distribution of the number of activities was used. So, for example, the 744 million euro expenditure related to hiking activities was distributed based on the number of hiking activities per province.

B. Nature tourism by residents

Tourism is defined as all activities for leisure that include at least one overnight stay. Data on the expenditure by residents were obtained from the Dutch tourism statistics for provinces, which in turn are based on survey results (the 'continuous holiday survey'). These statistics provide information on the different kinds of expenditures by residents, the types of holidays and the different regions (provinces) where the holidays take place. In order to delineate nature related tourism, we selected the following holiday types: nature holidays, active holidays (which include hiking and cycling holidays), beach holidays and water sports holidays. Expenditure include costs for (1) accommodation, (2) food and drinks, (3) travel costs, and (4) other costs (entry fees, etcetera). Expenditure in shops is excluded.

Table 3.7.6 Expenditure on nature related tourism activities by region, 2017

	Active tourism	Nature tourism	Water sports tourism	Beach tourism	Other	Total	Percentage of total tourism expenditure
	million euro						%
Groningen	7.7	7.6	0.0	0.6	4.2	20.1	27.9
Friesland	85.0	35.5	20.0	45.5	11.1	197.1	60.8
Drenthe	68.9	33.2	1.3	2.3	12.7	118.4	41.4
Overijssel	69.6	27.6	3.5	1.2	12.8	114.6	40.3
Flevoland	15.2	6.6	1.5	1.2	12.7	37.1	36.7
Gelderland	153.2	79.3	3.6	1.5	27.8	265.3	48.6
Utrecht	14.8	6.7	0.0	0.6	3.9	26.0	33.5
Noord-Holland	88.1	28.2	1.7	64.1	14.1	196.2	52.8
Zuid-Holland	18.7	8.5	0.2	29.5	11.7	68.5	30.6
Zeeland	47.5	8.6	1.1	94.4	12.0	163.7	63.7
Noord-Brabant	63.5	18.8	2.5	2.2	19.9	107.0	32.0
Limburg	114.2	38.6	0.1	3.1	10.3	166.3	33.5
Total	748.0	301.1	36.0	247.1	155.6	1,487.8	43.7

Total nature related expenditure of residents within the Netherlands was 1.5 billion euro in 2017 (Table 3.7.6). This is c. 44 % of total tourism related expenditure. Most expenditure is related to active tourism, followed by nature tourism and beach tourism. Expenditure on nature related tourism is highest in the provinces Gelderland, Friesland and Noord Holland.

Table 3.7.7 Expenditure on nature related tourism activities by type of surrounding, 2017

	Active tourism	Nature tourism	Water sports tourism	Beach tourism	Other	Total
	million euro					
Agricultural areas	94.7	35.9	1.6	1.6	27.7	161.6
Forest and heath	316.9	172.0	3.1	7.1	85.9	585.1
Seaside, beach and dunes	147.3	51.8	4.3	227.6	23.5	454.7
Inland water	73.9	16.6	26.9	5.6	13.9	136.8
Hills	71.1	24.3	0.0	1.6	11.2	108.2
Other areas	46.5	5.5	0.0	6.2	26.8	84.9

Based on information from the surveys, nature related expenditures by residents could also be allocated to the main type of surrounding (Table 3.7.7). Here we see that total expenditures are highest for forests and heath areas, closely followed by seaside, beach and dunes.

C. Nature tourism by non-residents

Data for tourism expenditures by non-residents (inbound tourism) were directly obtained from the Dutch TSA. Total expenditure by inbound tourism (excluding business travel) equals 6.5 billion euro (2016). We excluded expenditures related to shopping. Most inbound tourism in the Netherlands takes place in the large urban areas (i.e. Amsterdam, The Hague, etc.). No information is available on the main motive of the inbound tourists. Therefore, as an approximation, we took the location where these tourists stay overnight to delineate nature related tourism by non-residents. We selected the following tourism areas: coast, water sport areas, forest and heath areas.

Table 3.7.8 Nature related expenditure by non-residents

million euros	2015	2016	2017
Coast	1485	1561	1742
Watersport	539	559	618
Forest, Heath, Northeast Netherlands	226	238	302
Forest, Heath, Middle Netherlands	233	273	341
Forest, Heath, South Netherlands	858	941	997
Total	3341	3572	4000

Total nature related expenditure by non-residents was 3.3 billion euro in 2015 (Table 3.7.8).

D. Other related expenditure (mainly consumer durables)
Data on total other consumer expenditure related to tourism was obtained from the tourism satellite accounts (TSA). This mainly concerns expenditure on goods and services that households need for their recreational activities, such as camping equipment, walking boots, etcetera. Here we assumed the same percentage as for nature related tourism to calculate the nature related expenditure.

Overview
Table 3.7.9 provides an overview of all nature recreation and tourism related expenditure for 2015-2017. In this period total expenditure increased from 9.8 billion to 10.8 billion euro.

Table 3.7.9 Overview of nature related expenditure for tourism and recreation activities

million euros	2015	2016	2017
Nature-related recreation			
Hiking	744	750	798
Cycling	301	315	333
Other outdoor recreation	1543	1560	1650
Water sports	315	320	334
Outdoor sports	970	989	1031
Nature tourism: residents			
Active tourism	741	741	784
Beach tourism	234	234	247
Nature tourism	285	285	301
Other	147	147	156
Nature tourism: non-residents			
Coast	1485	1561	1742
Watersport areas	539	559	618
Other nature areas	1317	1452	1640
Other expenditure			
Total	1146	1160	1194
TOTAL	**9766**	**10072**	**10827**

3.7.3 Discussion

Resource rent method
Applying the resource rent method for nature tourism and recreation is problematic for a number of reasons:

- The tourism sector consists of many different ISIC categories. It is difficult to delineate the sector, particularly to determine which industries may directly benefit from nature.

- The resource rent is low for accommodation (ISIC) but high for food and beverage services. This may be an indication that we are not calculating the resource rent related with nature but something else.
- It is difficult to make a good estimate for the labour input of self-employed persons, which is important in the tourism sector. This will have a significant impact on the calculated results.
- It is difficult to determine the percentage of nature-related tourism and recreation for different industries. There is a high degree of uncertainty for the calculated percentages.

There is, however, also a more fundamental issue in applying the resource rent method for nature-based tourism and recreation. Businesses that profit from being located in an attractive natural environment for visitors do not directly pay extra for using these ecosystem assets. Basically, nature provides a ‘free service’ that does not contribute value to the net operating surplus of these businesses. Consequently, it can be expected that the resource rent related to nature is very low or near zero. There is consequently little sense in trying to calculate a resource rent.

Consumer expenditure
Nature provides all kinds of opportunities for recreational and tourism activities for people. This leads to all kinds of expenditures by households. First of all, people have to travel to the recreation site by car, train, and other means, which involves costs. Sometimes admission fees have to be paid to gain access to the site. During the activity, food and drinks are bought from on-site facilities. The activity may also include overnight stays, for example in hotels or camping sites. Finally, people need all kinds of products that they will use during these activities, such as hiking boots, bikes, and camper vans.

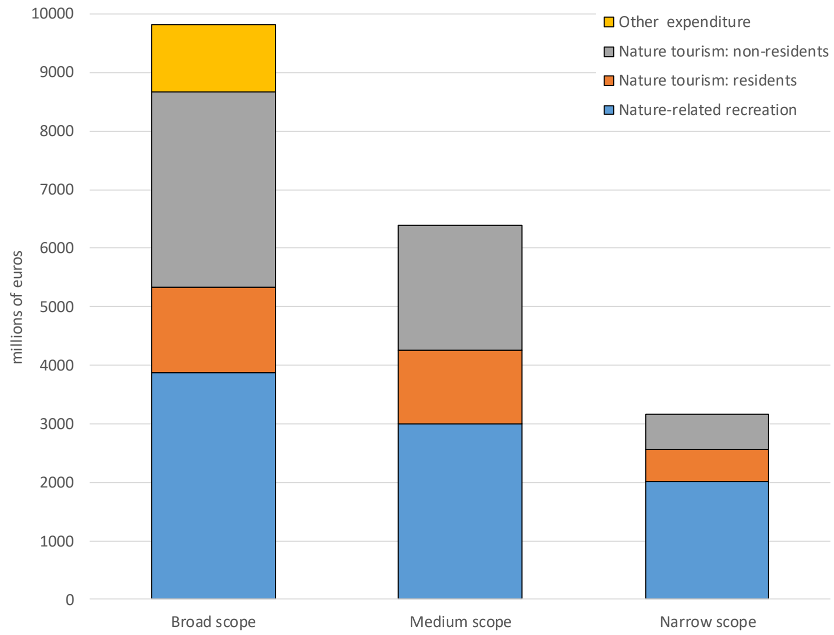
The consumer expenditure approach as applied in this study in principle uses the same approach as the travel cost method that is often used to value recreational services (e.g. Barton and Obst, 2019). The travel cost method assumes that travel costs of tourists and recreationists can be taken as an indication for their willingness to pay for the services of nature. The consumer expenditure approach as presented here is very similar to the ‘simple’ travel cost method applied in the United Kingdom to value these ecosystem services (ONS, 2016). A difference is that we consider not only travel costs and admission fees, but also expenditure on foods and drinks and (for tourism) accommodation. We consider all expenditures induced by nature as a measure for the value of the service.

Because the results of this method are dependent on the scope of the expenditure, we have tested three scenarios and calculated the associated nature-related expenditures:

1) *Limited scope*: travel costs, admissions fees.
2) *Medium scope*: travel costs, admissions fees, accommodation costs, other costs.
3) *Broad scope*: travel costs, admissions fees, accommodation costs, other costs, expenditure on food and drinks, other related expenditure (mainly consumer durables).

The results show that the range is quite considerable (from 3.2 billion to 9.8 billion euro; see figure 3.7.10). More discussion is needed, also on an international level, about what would be the most appropriate scope of the expenditure to include.

Figure 3.7.10 Nature related expenditure for tourism and recreation activities calculated according to three scenarios



Survey results show that many recreational activities do not involve costs at all. Respondents indicated that 55% of all recreational activities involve no costs. This is even higher for nature-related activities, namely 76% for outdoor sports and 82% for outdoor recreation (including hiking and cycling). This reflects that most outdoor activities take place in the direct neighbourhood which incur no travel or admission costs. When the consumer expenditure method is used, these activities are assigned zero value. One approach that has been applied to include a value for these kind of activities is to value the opportunity costs of time spent during the outdoor recreation activities. Using data on the total number of activities/holidays, the average time spent on each activity, a median wage, and a factor to reflect assumed opportunity cost, a monetary value can be calculated (ONS, 2016). A quick calculation for the Netherlands results in values of 10 billion euro for nature related tourism (by residents) and 68 billion euro for nature related recreation.[21] However, the valuation of time in ecosystem accounts is particularly controversial as it is inconsistent with SNA approaches (SEEA EEA; ONS, 2016). These values will therefore not be used for the calculation of the asset values.

What is the preferred method to use?
To determine what is the most appropriate method to value nature-related tourism and recreation we have to address two issues:

(1) What is the nature of this ecosystem service?
As discussed above, these ecosystem services could be interpreted as a service that contributes to production activities (businesses) or consumption activities (households and non-residents). Here we argue that this service should be interpreted as the latter, namely a direct supply of an ecosystem service to households and non-residents. The main argument is that cultural services, which are defined as '*giving rise to intellectual and symbolic benefits obtained by people from ecosystems through recreation, knowledge development, relaxation and spiritual*

[21] using an average wage of 33 euro/hour (2015) and an opportunity cost factor of 0.75.

reflection (SEEA EEA par. 3.2), are by definition supplied to people. People (households) are thus the direct beneficiaries from the opportunities that nature provides for recreational activities. Businesses active in the tourism sector are only indirect beneficiaries as they benefit from the increased demand of goods and services related to the activities by households. Accordingly, the resource rent method is by definition not suitable to calculate the contribution of nature-related tourism.

(2) Is the consumer expenditure method an appropriate approach to value this ecosystem service? To address this issue we have to answer three questions:

1. *Does this method provide exchange values?* Expenditures by households are key examples of market transactions and consequently represent exchanges values, so the answer is affirmative.
2. *Do the values provided by this method represent a contribution to an economic benefit?* Expenditures by households on accommodation, travel, consumer durables and so on are, in SNA terms, part of final household consumption. Final consumption by households plus consumption by government plus gross capital formation plus exports less imports equals GDP. So, these values indeed represent a contribution to an economic benefit.
3. *Do the values provided by this method represent a contribution by ecosystems?* The tourism and leisure activities under consideration can only take place, resulting in the (extra) spending, because of the presence of nature areas. The argument is that without the ecosystems and the cultural services they provide this expenditure would not occur and GDP would be lower. Thus, the expenditure can be taken as a measure for the value of the ecosystem service.

Assigning expenditure values to ecosystems reattributes consumer expenditure values that are already recorded elsewhere in the SNA. In section 6.2 it is shown how these values can be integrated into the accounting framework while avoiding double counting.

We conclude that, at this moment, the consumer expenditure method provides the best approximation for valuing nature-related tourism and recreation. The advantage of this method is that it provides a pragmatic approach: it draws upon existing statistical data, is relatively straightforward to understand and easy to undertake. Furthermore, it incorporates the direct economic benefits provided by nature for recreation and tourism and is therefore fully consistent with SNA exchange values.

The values obtained by consumer expenditure only capture part of the economic benefits provided by these ecosystem services. Recreational activities in nature provide all kinds of (positive) health effects for people. This will provide economic benefits in the form of reduced healthcare costs. These values are not yet included in the SNA and thus will increase GDP. The exact health effects are often difficult to quantify, so further research is needed to find out whether this value component can be added for a future update of the monetary accounts. Furthermore, nature based tourism and recreation also provide welfare values that are probably much higher than the exchange values presented here. Consumers are willing to pay much more to enjoy nature than they are actually spending on travel costs or admission fees. In a future update, it may be worthwhile to present welfare values for tourism and recreation alongside the exchange values.

3.8 Amenity services: The value of living near nature: an analysis of Dutch house values

Authors: Linda de Jongh (CBS), dr. Michiel N. Daams (RUG), dr. Frans J. Sijtsma (RUG)

1. Introduction

This study considers the largest economic sector, in terms of material wealth, in which ecosystem services add value: the real estate sector. Real estate that provides housing represents close to 41 percent of the capital stock in the Netherlands – and part of this capital can be indirectly attributed to ecosystems (Statistics Netherlands, 2019). The reason for this is that the value of houses does not come from the provision of living space alone, but also from the quality of the local environment surrounding a house. The quality of a house's local environment is known to be determined by access to jobs and so-called 'amenities' such as shops, schools, restaurants – or nature. Nature offers 'ecosystem services', such as space recreation, visual aesthetics, or air and noise pollution reduction. Such services are valued by people. As a result, when nature in certain locations provides ecosystem services that are sought after, this may cause houses with relative good access to these locations to sell at higher prices. In other words, house buyers may be willing to pay for a house and associated land in order to enjoy the benefits from the services that green and blue surroundings may provide.

This analysis is aimed to reveal the economic value of nature's ecosystem services that is reflected in the value of houses in the Netherlands. Specific attention is paid to so-called cultural ecosystem services. These reflect "nonmaterial benefits people obtain from ecosystems" which may relate in particular to recreational activities and aesthetic enjoyment (MEA 2005, 40). The presence of such cultural services is approximated using information from an innovative online map-based survey, Greenmapper. Greenmapper measures which natural places in the Netherlands and beyond are highly appreciated. Highly appreciated nature implies locations where cultural ecosystem services are considered by potential house buyers and therefore may, on average, increase house values. Besides this approach, which has been found to explain the impact of nature on nearby house prices beyond the reach of existing models of house prices, see Daams et al. (2016), this study also considers more generally how nature's ecosystems may influence house values.

This study's methodology is based on the hedonic pricing model. This model disentangles the value of houses into implicit values for house attributes using regression analysis. Such analysis provides estimates of the value of a marginal change in, for example, square meters of floor space or the number of rooms - but also in terms of whether a house is located close to nature or not.. Hence, results from this model will indicate the difference in value between houses close to nature or houses farther away from nature, everything else held constant.
This then identifies part of the total economic benefits provided by nature. This estimation does for instance not include economic benefits from nature which relate to non-local tourism or overnight stays. Nor does it necessarily include the value of nature as being revealed through daytrip destinations by, for example, bike-car/train combinations, because the involved areas may be relative far away from the house location. The analysis captures the benefits from nature that are provided within people's daily living environment. Hence, this study considers the nature that is likely to have considerable impact on people's well-being and, by consequence of increased housing values, the capital stock.

2. Literature

The hedonic pricing method

The hedonic method of housing value is often traced back to Lancaster's (1966) theoretical notion that the utility that people derive from a heterogeneous good stems not from the good as a whole, but from its distinct characteristics (Lancaster, 1966). Hence, as demand and supply of certain house characteristics shape home values, the implicit value of individual characteristics of homes can be estimated (Rosen, 1974). This is what hedonic pricing models do, using regression analysis. Home values can in this way be disentangled into implicit values for the structural and locational characteristics of homes. The implicit value of proximity to nature, for example, can be interpreted as an indication of how much home buyers are willing to pay for living nearby nature as compared to living farther away from nature. Of course, not every potential buyer may be willing to pay for nature. However, home prices are driven by the interactions between all potential buyers – and therefore reflect how this house characteristic is evaluated on average – and not just by the buyer who made the winning bid (Palmquist, 2005).

Advantages and disadvantages

The hedonic pricing method's major strength is that it considers actual market behavior – unlike laboratory-like study designs that investigate how people respond to hypothetical changes in their living environment. Moreover, the value of homes represents the lion's share of material wealth in developed economies – on average, approximately 85% in OECD countries (OECD, 2017).

However, to appropriately reveal how home characteristics add to people's well-being, as people value them, requires that some main limitations be accounted for (Palmquist, 2005). The first of two main limitations that are relevant in the context of environmental amenity valuation is omitted variable bias (Palmquist, 2005). Such type of bias may arise when not all the variables that are correlated with the variable of interest are included in a model – which is commonly the case, given the heterogeneity of houses and their being bound to locations which inherently differ (De Haan and Diewert, 2013). Therefore, variables in a hedonic model should be carefully selected, as omission may otherwise confound the estimation. However, omitted processes may also be dealt with indirectly using spatial controls (e.g., location dummies). Spatial controls account for local similarity in house prices (e.g., within a zip code area). Such spatial structure in house values can also be accounted for through the model's error term (De Haan and Diewert, 2013). The second main limitation is the hedonic method's vulnerability to multicollinearity (Palmquist, 2005). For example, in an environmental valuation context, distinct types of nature may be located nearby each other (e.g. forest and agricultural land); then, when homes' proximity to each of these types of nature is included in a hedonic pricing model, the model may not be able to separate their implicit contributions to home value due to their high coherence, implying that this should be assessed. It has been shown that a useful approach is to aggregate natural amenities to prevent issues of multicollinearity between environmental amenities that are co-located (Pendleton and Shonkwiler 2001).

Despite some limitations, the hedonic pricing model has a clear overall benefit since it is essentially the sole model that can contribute to ecosystem valuation from a house value perspective. Hedonic house value analysis, moreover, offers a clear and powerful means of assessing economic value of nature and may do so beyond the scope of other methods, including the replacement cost method and the travel cost methods, as indicated by De Groot et al. (2002).

Hedonic valuation of nature areas

The hedonic method is firmly established in the environmental valuation literature that considers nature (Anderson and West, 2006; Czembrowski et al., 2016; Daams et al., 2016; Gibbons et al., 2014; Liebelt et al., 2018; McConnell and Walls, 2005; Melichar and Kaprová, 2013; Panduro and Veie, 2013; Schläpfer et al., 2015; Von Graevenitz and Panduro, 2015; Waltert and Schläpfer, 2010; Xiao et al., 2016). The impact of nature on house prices reported in such studies tends to be within the 2 to 5 percent range. A partial reason for this is that impacts are typically identified for physical land use types, without further differentiating in terms of amenity quality, although some studies do (e.g. based on the greenness of vegetation, recreational infrastructure, or noise levels – see Bark et al., 2009 or Ham et al., 2012). In contrast to those approaches, Luttik (2000) find that when nature nearby homes is attractive this translates to relative high impact up to 16 percent. Daams et al. (2016) expand on that study by analyzing a reproducible perceptual measure, based on the Greenmapper survey, with which they enrich land use data for the entire Netherlands. Their study finds that the impact of highly attractive nature on house prices falls with distance smoothly: from 16.0 percent for houses within 500 meters, to 1.6 percent for houses up to 7 kilometers away. This indicates that the economic benefits of living nearby nature extends over a larger distance than earlier studies suggest, which is key to the understanding of the aggregate housing value impact of nature areas. Indeed, over which effects extend determines how many houses' values are impacted. Hence, the precise estimation of the distance interval across which nature impacts on house prices is an important analytical challenge.

3. Method and data

This research makes use of two classifications of nature. Perceived attractive nature as defined through clustering of national markers in the Greenmapper (also called Hotspotmonitor, following Daams et al., 2016). In the remainder of this study we refer to these clusters of (nationally) attractive nature area as CANA. These areas are combined with other nature areas (ONA).

> Two types of nature areas are distinguished in this study:
> **CANA:** Clusters of attractive nature areas; attractive on a national scale.
> **ONA:** Other nature areas

Greenmapper is a value mapping survey in which people are asked to identify nature areas that they perceive as attractive, valuable or important (Sijtsma et al., 2012, De Vries et al., 2013; Sijtsma et al., 2013; Davis et al., 2016; Bijker and Sijtsma, 2017; Scholte et al., 2018). The Greenmapper survey has different spatial levels. The Greenmapper has a location-based design with participant's residence as the starting point. Participants are asked to identify natural places they find attractive at four spatial levels. They can mark one place at each spatial level. Local: a circle with a range of two kilometers from home. Regional: a circle with a range of 20 kilometers from home. National: the whole of the country. World level: anywhere in the world. Every individual respondent marks point-shaped places (Bijker and Sijtsma, 2017). Using GIS-based clustering techniques these points could be grouped together to form highly-appreciated polygons, allowing various types of analysis (De Vries et al., 2013; Daams et al., 2016). In this study, only clusters of national markers are used (CANA). The ONA nature will in many instances be the recipient of regional or local markers: that is, they can be locally or regionally attractive and important but not on a national level. The available local and regional Greenmapper data are not used in this study because they are insufficient in number. The Greenmapper database needs to be extended from (around) 8000 respondents now, to around 200,000 respondents in the future to accommodate analysis based on attractiveness on all levels.

Using the cluster areas of national attractiveness (CANA), Daams et al. (2016) show that it is attractiveness of nature that result in larger distance effects than other house prices had substantiated before. This means that even on large distances between nature and a dwelling, nature brings positive economic benefits to the dwelling. Our research confirms this conclusion. However, it is also shown that nature areas not perceived as attractive can still produce economic benefits, although in a much smaller amount. Therefore, CANA is combined with ONA to account for this. ONA are established based on the Ecosystem Units map (Van Leeuwen et al., 2017; Statistics Netherlands, 2017). Some ecosystem types are classified as ‘nature’ (see table 1 for the list). Then a grid of 100x100 meter is used; when at least 80 percent of the area is defined as nature as in table 1, this grid is defined as ONA. When an area is both a CANA and an ONA, the area is classified as CANA. Therefore, CANA is prioritized and ONA can be seen as *other* nature area (see also figure 1). Figure 2 shows the two classifications on a map for the Netherlands.

Table 1: Classification of the Ecosystem Units map for the definition of ONA

Nature	No nature
Hedgerows	(Non-)perennial plants
Forest (deciduous, coniferous, mixed)	Meadows
Heath land	Greenhouses
Inland dunes	Farmyards and barns
Fresh water wetlands	Other unpaved terrain
(Semi-) natural grassland	Residential areas and other built-up areas
Public green space	
Water (Sea, lakes and ponds, rivers and streams)	

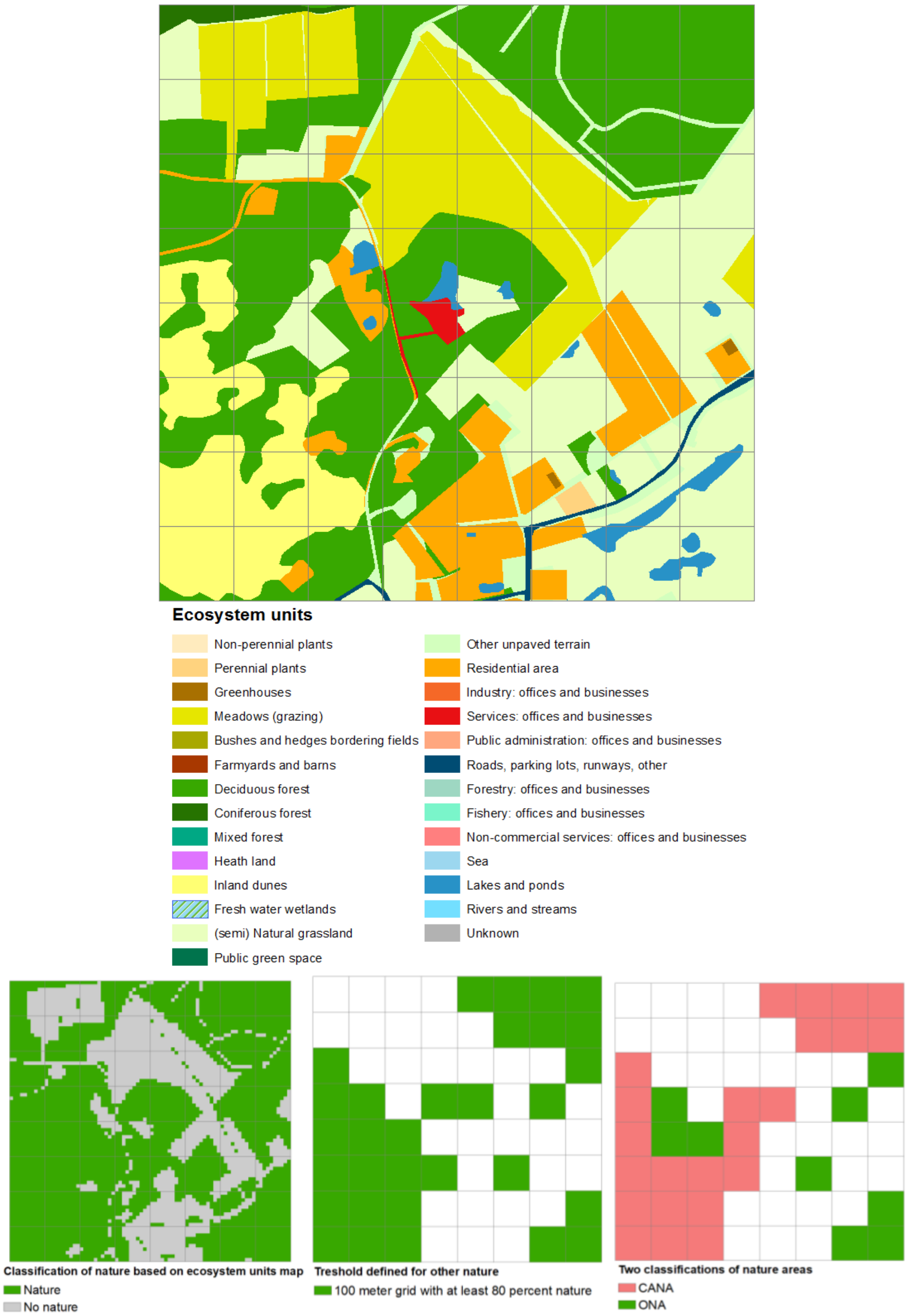


Figure 1: Two classifications of nature areas: clusters of attractive nature areas (CANA) and other nature areas (ONA)

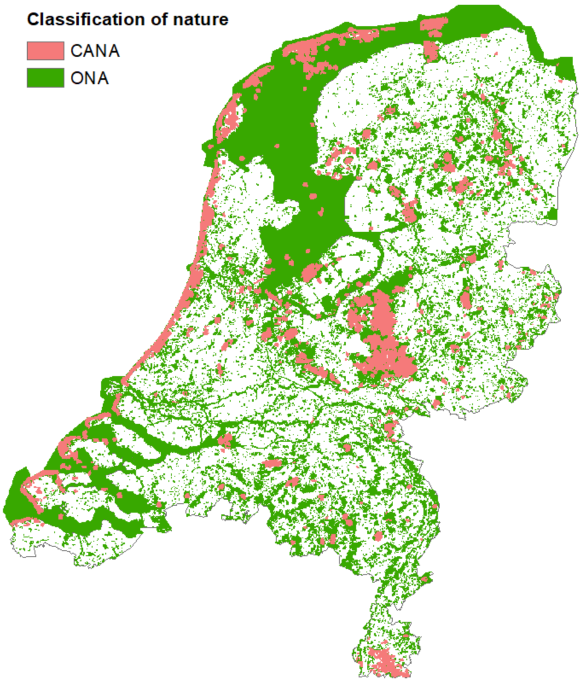


Figure 2: Classification of nature (CANA and ONA) for the Netherlands

From the housing stock registry (Statistics Netherlands, 2018), a selection is made of single-family dwellings with solely a residential function for the year 2013. This excludes apartments, as they have a different relationship between closeness of nature to the value of the dwelling (e.g. because of stock level and view). This gives a dataset consisting of 4.5 million observations of which there is information on assessed property value (WOZ-value), characteristics of the dwelling (such as size of the dwelling and parcel, building year, dwelling type) and ownership (owner-occupier of leased by a tenant). From the Building and Addresses Registry (BAG) the location of each dwelling in the dataset is known. This is used to calculate the Euclidean distance between each building and the nearest CANA and ONA. To account for outliers the dataset is cleaned by deleting observations for which one of the variables is missing or for which WOZ, size of the dwelling, size of the parcel or the ratio WOZ to size of dwelling or WOZ to size of parcel is in the very far tails of the distribution (0.25 percent).

Table 2 shows descriptive statistics of the variables of interest. The dependent variable of interest in this research is the value of the dwelling as captured in the assessed property value. WOZ is used as this is known for all the dwellings in the dataset whereas market transaction prices are only known for houses sold in a specific period. This comes with the advantages that this study can observe values across the entire stock of single-family homes. A disadvantage, however, is that WOZ value might represent the preferences of people less accurately than market values – since it represents an assessment of value and not a value that arises directly from interactions between buyers and sellers of houses. The mean WOZ is €248,754. The independent variable of interest is the distance to nearest nature area, both CANA and ONA. The average distance to CANA is 4,906 meter and to ONA 392 meter. It is expected that there is a decreasing relationship between value of the dwelling and closeness to nature, meaning that the further away from nature the less the influence of nature on the value of the dwelling. To measure this, Daams et al. (2016) is followed by discretizing CANA and similarly for ONA. The distance intervals used are shown in table 2.

Table 2: Descriptive statistics (*N = 4.568.200*)

Variable	Mean	Std. Dev	Variable	Mean	Std. Dev
WOZ-value (euros)	248.754	125.208	Distance (m) to CANA	4.906	3.872
Living area (m2)	131,43	50,53	0-500 m	0,05	
Parcel size (m2)	991,99	2125,78	500-1000 m	0,06	
Type of dwelling			1000-2000 m	0,15	
Detached	0,18		2000-3000 m	0,13	
Semidetached	0,14		3000-4000 m	0,10	
End-of-terrace	0,21		4000-5000 m	0,10	
Terraced	0,47		5000-6000 m	0,09	
Rent	0,27		6000-7000 m	0,07	
Construction year			> 7000 m	0,25	
Before 1905	0,03		Distance (m) to ONA	392	280
Between 1906-1930	0,09		0-50 m	0,02	
Between 1931-1944	0,06		50-100 m	0,06	
Between 1945-1959	0,10		100-150 m	0,08	
Between 1960-1970	0,16		150-200 m	0,09	
Between 1971-1980	0,20		200-250 m	0,09	
Between 1981-1990	0,16		250 -300 m	0,09	
Between 1991-2000	0,13		300-350 m	0,09	
Between 2001-2010	0,08		350-400 m	0,08	
After 2010	0,01		400-450 m	0,07	
			> 450 m	0,33	

Control variables in the regression analysis are year of construction (discretized to intervals shown in table 2), size of the dwelling and parcel (in square meters; mean is, respectively, 131 and 992 meters[22]), whether the house is rented out (27 percent) or in use by the owner and type of dwelling (detached (18 percent), semidetached (14 percent), end-of-terrace (21 percent) or terraced (47 percent).

4. Hedonic price model

<u>Basic model specification explained</u>

A common specification of the hedonic price model for dwelling of interest *i (i* = 1,…, *n)* may be given by

$$\ln(WOZ_i) = \alpha + \sum_{j=1}^{m} \beta_j \boldsymbol{X}_{i,j} + \varepsilon_i \tag{1}$$

where α is the constant; $ln(WOZ_i)$ is the natural logarithm of the assessed property value (WOZ) of dwelling *i*; $\boldsymbol{X}_{ij}$ is the *jth* characteristic *(j* = 1,…, *m)*; and ε_i denotes standard errors that are spatially clustered to mop up remaining local correlations below street-level (PC6 level). The functional form is a semilog, since WOZ is skewed to the right, to mitigate heteroskedasticity (Diewert, 2003).

[22] Size of parcel is higher than expected because buildings belonging to corporations are included. Parcel size is based on cadastral map of ownership and these building blocks are based on one, large, cadastral parcel. This does not influence the final regression results.

Model specification used in this study

The main model specification in this study considers the proximity of homes to CANA and ONA. CANA is a holistic proxy of nature areas that are perceived attractive by residents in general. Our approach addresses both of the main limitations that are associated with hedonic pricing models. First, the holistic character of this measure mitigates possible issues with regard to multicollinearity that might arise if CANA were split by land use type (Pendleton and Shonkwiler, 2001). Indeed, attractive forest might be similarly close to a home as attractive grasslands if they constitute the same CANA. Second, to account for omitted variable bias from structural and locational house characteristics that are constant on a local level, for example safety in the neighborhood or housing market effects, the regression is estimated in spatial first differences. These first differences are taken on zip code (PC4) level since this is the lowest spatial scale at which one-unit changes in the dummies for proximity to CANA remain plausible (Daams et al., 2016). This implies that variance of the data that might otherwise lead the model to reveal a higher true impact of CANA is removed from the estimation (Abbott and Klaiber, 2011; Daams et al., 2016). The benefit of this approach, however, is that it is stricter than common so-called spatial fixed effect models (see Von Graevenitz and Panduro, 2015) as it accounts not only for spatial structure in prices but also in terms of house characteristics: within-pair similarity in both observed and unobserved characteristics is cancelled out. Noteworthy is that the estimation data are first-differenced for pairs of homes that have been matched based on having location within the same PC4 area, following the techniques described in detail in Guo et al. (2015). This gives the following hedonic price model to be estimated:

$$ln(WOZ_{iz}) - ln\left(WOZ_{jz}\right) = \sum_{k=1}^{K} \beta_k(\boldsymbol{X}_{izk} - \boldsymbol{X}_{jzk}) + \sum_{c=1}^{C} \beta_c(\boldsymbol{CANA}_{izc} - \boldsymbol{CANA}_{jzc}) + \sum_{d=1}^{D} \beta_d(\boldsymbol{ONA}_{izd} - \boldsymbol{ONA}_{jzd}) + \varepsilon_{ijz} \quad (2)$$

where $ln(WOZ_{iz})$-$ln(WOZ_{jz})$ is the difference in assessed property value of paired houses *i* and *j*, both located in the same 4-digit zip code area *z*; ***X*** a vector of control variables including year of construction, type of dwelling, size of dwelling, parcel size, and leased status; $CANA_{i,z,c}$ is the vector of dummy variables for dwelling *i* indicating the distance to the closest CANA within interval *c* (*c* = 0-500 m, 500-1,000 m, 1,000-2,000 m, 2,000-3,000 m, 3,000-4,000 m, 4,000-5,000 m, 5,000-6,000 m, 6,000-7,000 m); and $ONA_{i,d}$ is the vector of dummy variables for dwelling *i* indicating the distance to closest ONA within interval *d* (*d* = 0-50 m, 50-100 m, 100-150 m, 150-200 m, 200-250 m, 250-300 m, 300-350 m, 350-400 m).

5. Empirical estimates

Model (2) is the main specification estimated using the data described above with more than 4.5 million observations. The results are shown in table 3, column (1). The control variables behave as expected: larger size of dwelling and parcel means a higher WOZ; dwellings that are rented have on average a lower WOZ; dwellings build before the year 2000 have on average a lower WOZ compared to dwellings build after thereafter (base variable is built after 2010). Additionally, type of dwelling matters as detached dwellings (base variable) are on average the most expensive, followed by semidetached, end-of-terrace and terraced dwellings.

In terms of the variables of interest, closeness to nature, both CANA and ONA have an effect on WOZ. The closer a dwelling to CANA or ONA the larger the WOZ of that dwelling. However, the effect for CANA is much larger and exists over a much larger distance. If the distance between the dwelling and CANA is below 500 meters 6.7[23] percent of WOZ can be attributed to CANA. This effect becomes smaller the further away CANA, but stretches up to 7000 meters, where the effect is reduced to 0.7 percent, but still significant at the 1 percent level.

If the dwelling is closer than 50 meters from ONA WOZ is on average 4.9 percent higher. This effect decreases in size over to a distance of 350 meter, where the effect is 0.1 percent, but significant at the 1 percent level and then becomes negligible. Therefore, the effect of ONA is both smaller and restricted to a much shorter distance from the dwelling compared to CANA.

6. Regional variation

Daams et al. (2016) show that there are variations between regions, where the price effect is higher and stretches further in urbanized regions compared to nonurban regions. Therefore, model (2) will be separately estimated based on urbanization level of the smaller spatial scale of municipalities; that is, the municipality where the dwelling is located.

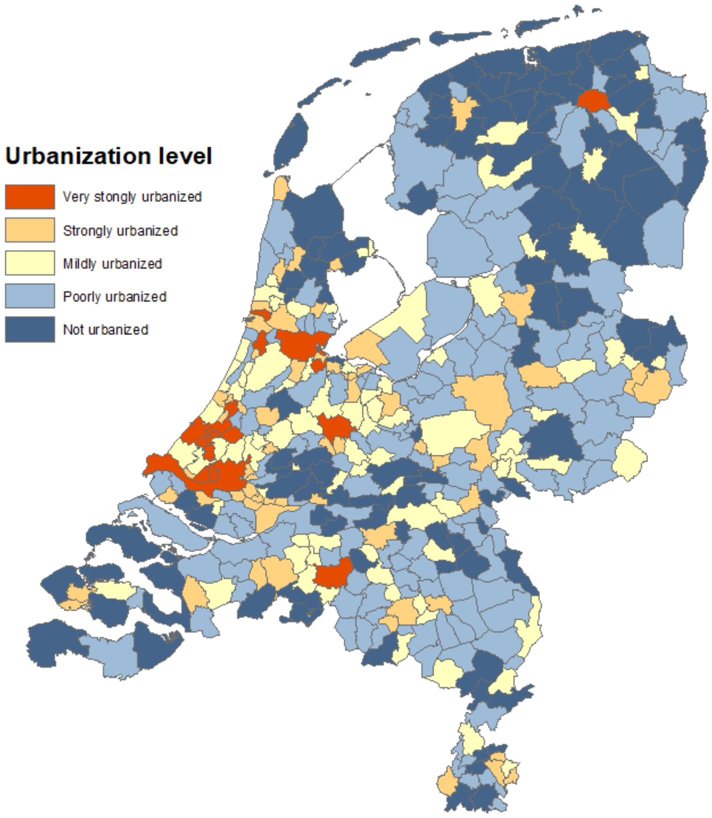


Figure 3: Urbanization levels across municipalities in 2013 (source: Statistics Netherlands, 2013)

For this, the urbanization classification from Statistics Netherlands (2013) is used (see figure 3). This classification is based on the average amount of addresses per square km in 2013 per municipality. There are five different urbanization levels: 1 (very strongly urbanized); 2 (strongly

[23] In order to denote the effect as a percentage, the regression coefficient (β) needs to be corrected because of the log-level model: percentage effect = $e^{\beta}-1 = e^{0.0644}-1 = 0.0665$.

urbanized); 3 (mildly urbanized); 4 (poorly urbanized); and 5 (not urbanized). Figure 2 shows the urbanization levels on a map.

Table 4 shows the estimators corrected for the log-level regression. It indicates that large regional variations exist. Most notable are the large variations in distance to CANA across urbanization levels: in very strong urbanized regions 21.9 percent of WOZ can be attributed to CANA within 500 meter from the dwelling. Secondly, the effect stretches up to 6000 meters. Contrarily, in not-urbanized regions there is only an effect of 2.2 percent if the dwelling is within 500 meters from highly appreciated nature on a national scale.

In all regions, the remaining ONA have only a limited range of the effect: it reaches to 400 meter maximum. In very strongly urbanized regions ONA seems less valued by home buyers, however from urbanization level 2 onwards the effect is larger for urbanized regions and stretches further compared to lower levels of urbanization.

In other words, from a housing perspective, the aggregate value of nature in urban areas is much higher than the value of nature in rural areas for three main reasons: 1) a higher percentage of the value of a house is attributed to nature areas; 2) the density of houses is higher, so that more houses contribute a percentage of their value to each nature area in the vicinity; and 3) the average price per m2 is higher. An explanation based on the mechanism which determines how nature is valued by home owners or users, is that in urban areas nature is relative scarce.

Table 3: Regression results of model (2) for the Netherlands and different urbanization levels

Variables	(1) Netherlands	(2) Urbanization level				
		1	2	3	4	5
Living area (ln)	0.579***	0.677***	0.670***	0.605***	0.526***	0.491***
	(0.000715)	(0.00261)	(0.00156)	(0.00150)	(0.00120)	(0.00150)
Parcel size (ln)	0.0416***	0.0132***	0.0265***	0.0425***	0.0557***	0.0691***
	(0.000213)	(0.000562)	(0.000384)	(0.000438)	(0.000391)	(0.000593)
Rent	-0.125***	-0.0788***	-0.1000***	-0.120***	-0.149***	-0.166***
	(0.000370)	(0.000977)	(0.000633)	(0.000767)	(0.000738)	(0.00109)
Construction year						
Before 1905	-0.154***	-0.0950***	-0.181***	-0.121***	-0.169***	-0.157***
	(0.00362)	(0.0104)	(0.00726)	(0.0108)	(0.00598)	(0.00730)
Between 1906-1930	-0.156***	-0.0952***	-0.180***	-0.125***	-0.172***	-0.164***
	(0.00356)	(0.0102)	(0.00700)	(0.0107)	(0.00586)	(0.00720)
Between 1931-1944	-0.126***	-0.0783***	-0.136***	-0.0892***	-0.150***	-0.157***
	(0.00359)	(0.0103)	(0.00704)	(0.0107)	(0.00591)	(0.00727)
Between 1945-1959	-0.152***	-0.125***	-0.175***	-0.127***	-0.158***	-0.154***
	(0.00356)	(0.0105)	(0.00697)	(0.0107)	(0.00587)	(0.00723)
Between 1960-1970	-0.148***	-0.154***	-0.176***	-0.123***	-0.155***	-0.148***
	(0.00354)	(0.0104)	(0.00698)	(0.0106)	(0.00583)	(0.00718)
Between 1971-1980	-0.125***	-0.138***	-0.167***	-0.103***	-0.125***	-0.112***
	(0.00355)	(0.0104)	(0.00699)	(0.0106)	(0.00583)	(0.00719)
Between 1981-1990	-0.0818***	-0.0890***	-0.110***	-0.0570***	-0.0818***	-0.0775***
	(0.00355)	(0.0102)	(0.00698)	(0.0106)	(0.00584)	(0.00722)
Between 1991-2000	-0.00650*	-0.0345***	-0.0259***	0.0247**	-0.00887	-0.0118
	(0.00356)	(0.00994)	(0.00699)	(0.0107)	(0.00584)	(0.00722)
Between 2001-2010	0.0126***	-0.000622	-0.0149**	0.0407***	0.00968*	0.0112
	(0.00349)	(0.00947)	(0.00678)	(0.0105)	(0.00574)	(0.00710)
End-of-terrace house	-0.273***	-0.293***	-0.275***	-0.269***	-0.268***	-0.264***
	(0.000588)	(0.00315)	(0.00147)	(0.00119)	(0.000960)	(0.00136)

Variables	(1) Netherlands	(2) Urbanization level				
		1	2	3	4	5
Semidetached house	-0.176***	-0.173***	-0.176***	-0.168***	-0.178***	-0.184***
	(0.000519)	(0.00313)	(0.00140)	(0.00109)	(0.000820)	(0.00110)
Terraced house	-0.324***	-0.331***	-0.322***	-0.324***	-0.321***	-0.315***
	(0.000615)	(0.00319)	(0.00150)	(0.00124)	(0.00102)	(0.00148)
Distance to nearest CANA						
Within 0-500 m	0.0644***	0.198***	0.0665***	0.0460***	0.0282***	0.0216***
	(0.00372)	(0.0118)	(0.00659)	(0.00396)	(0.00385)	(0.00429)
Within 500-1000 m	0.0411***	0.154***	0.0481***	0.0209***	0.0140***	
	(0.00348)	(0.0110)	(0.00597)	(0.00309)	(0.00321)	
Within 1000-2000 m	0.0344***	0.113***	0.0543***		0.00939***	
	(0.00322)	(0.0101)	(0.00539)		(0.00232)	
Within 2000-3000 m	0.0294***	0.0900***	0.0533***			
	(0.00300)	(0.00897)	(0.00487)			
Within 3000-4000 m	0.0206***	0.0909***	0.0268***			
	(0.00274)	(0.00793)	(0.00388)			
Within 4000-5000 m	0.0114***	0.0652***	0.0204***			
	(0.00244)	(0.00678)	(0.00267)			
Within 5000-6000 m	0.0112***	0.0504***				
	(0.00217)	(0.00510)				
Within 6000-7000 m	0.00743***					
	(0.00173)					
Distance to nearest ONA						
Within 0-50 m	0.0482***	0.0427***	0.0585***	0.0538***	0.0407***	0.0274***
	(0.00120)	(0.00341)	(0.00235)	(0.00245)	(0.00231)	(0.00307)
Within 50-100 m	0.0342***	0.0349***	0.0407***	0.0377***	0.0313***	0.0171***
	(0.000802)	(0.00229)	(0.00162)	(0.00166)	(0.00147)	(0.00195)
Within 100-150 m	0.0204***	0.0272***	0.0265***	0.0243***	0.0154***	0.00592***
	(0.000689)	(0.00195)	(0.00142)	(0.00145)	(0.00120)	(0.00160)
Within 150-200 m	0.0135***	0.0217***	0.0216***	0.0162***	0.00670***	0.00287**
	(0.000635)	(0.00177)	(0.00136)	(0.00133)	(0.00105)	(0.00133)
Within 200-250 m	0.00779***	0.0117***	0.0145***	0.0118***	0.00291***	
	(0.000596)	(0.00165)	(0.00128)	(0.00126)	(0.000911)	
Within 250-300 m	0.00421***	0.00612***	0.00872***	0.00915***		
	(0.000559)	(0.00141)	(0.00122)	(0.00122)		
Within 300-350 m	0.00127**		0.00538***	0.00507***		
	(0.000498)		(0.00116)	(0.00115)		
Within 350-400 m			0.00327***	0.00323***		
			(0.00101)	(0.00104)		
Constant	-6.48e-07	-3.49e-05	-6.62e-06	-8.08e-06	-6.46e-07	4.17e-05
	(0.000125)	(0.000389)	(0.000240)	(0.000272)	(0.000230)	(0.000313)
Observations	4,564,195	467,905	1,256,424	1,039,062	1,204,331	596,473
R-squared	0.709	0.679	0.716	0.729	0.718	0.705
Adjusted R-squared	0.709	0.679	0.716	0.729	0.718	0.705
F statistic	72146	4281	14883	22810	34085	21740
Root MSE	0.176	0.150	0.153	0.172	0.188	0.205

Note: Dependent variable is the natural log (ln) of WOZ-value. All variables are differenced with a random house within the same zip code area (PC4). The reference categories include construction year after 2010; detached house; not rented. Specification (1) includes all observations in the dataset; Specification (2) are the results of the regression analysis using data for each urbanization level seperately. Urbanization levels: 1) very stronly urbanized; 2) strongly urbanized; 3) mildly urbanized; 4) poorly urbanized; 5) not urbanized. Clustered standard errors are in parentheses. *** p<0.01, ** p<0.05, * p<0.1.

Table 4: Results of the hedonic pricing model for the Netherlands and different urbanization levels

	(1) Netherlands	(2) Urbanization level 1	2	3	4	5
Distance to nearest CANA						
Within 0-500 m	6,7	21,9	6,9	4,7	2,9	2,2
Within 500-1000 m	4,2	16,6	4,9	2,1	1,4	
Within 1000-2000 m	3,5	12,0	5,6		0,9	
Within 2000-3000 m	3,0	9,4	5,5			
Within 3000-4000 m	2,1	9,5	2,7			
Within 4000-5000 m	1,1	6,7	2,1			
Within 5000-6000 m	1,1	5,2				
Within 6000-7000 m	0,7					
Distance to nearest ONA						
Within 0-50 m	4,9	4,4	6,0	5,5	4,2	2,8
Within 50-100 m	3,5	3,6	4,2	3,8	3,2	1,7
Within 100-150 m	2,1	2,8	2,7	2,5	1,6	0,6
Within 150-200 m	1,4	2,2	2,2	1,6	0,7	0,3
Within 200-250 m	0,8	1,2	1,5	1,2	0,3	
Within 250-300 m	0,4	0,6	0,9	0,9		
Within 300-350 m	0,1		0,5	0,5		
Within 350-400 m			0,3	0,3		

7. Main results

Taking the results from table 4, the value that can be attributed to nearby nature of each dwelling can be calculated using information of WOZ-value, urbanity and distance to nearest CANA and ONA. This value is then allocated to the nearby nature areas. For CANA, the value is evenly distributed over all CANA areas within seven kilometers from the dwelling. For ONA, the radius is 500 meter as the results from previous section show a much smaller sketch of the visible effect.

The sum of the added value of CANA and ONA across the observed 4.5 million houses in our dataset is EUR 31.8 billion. Of this amount, 71 percent (EUR 22.5 billion) is attributed to CANA and the remaining 29 percent (EUR 9.3 billion) is attributed to ONA (see table 5). Table 6 shows the result per ecosystem type. Especially water areas (26.0 percent), forests (22.7 percent), public green space (20.1 percent), and dunes and beaches (10.8 percent) contribute to the amenity value of nature. Figure 4 shows the results on a map for the Netherlands.

Table 5: Value of CANA and ONA

Type of nature	million euro	percentage
Value CANA	22.526	71%
Value ONA	9.270	29%
TOTAL	31.796	

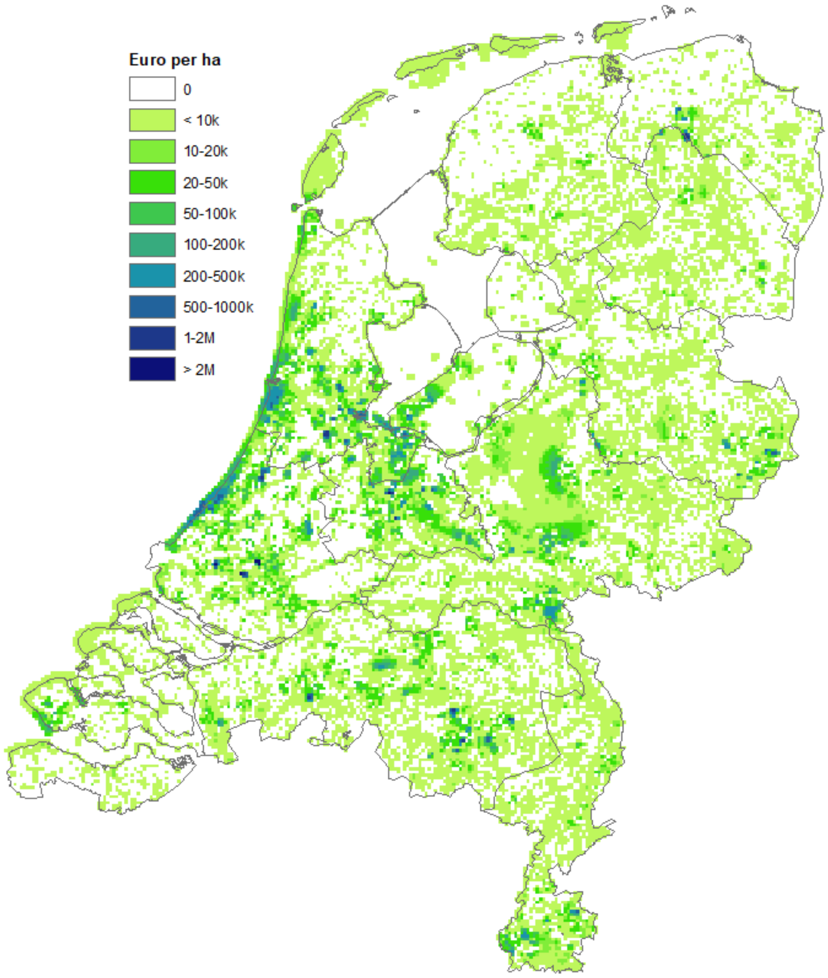


Figure 4: Spatial visualization of the housing value impacts of CANA and ONA

Table 6: Contribution of ecosystem type to house values

Ecosystem type	million euro	%
(Non)-perennial plants	856	3
Meadows	1576	5
Hedgerows	199	1
Greenhouses, farmyards and barns	31	0
Forest	7233	23
Heath land and inland dunes	745	2
Dunes and beaches	3503	11
Fresh water wetlands	284	1
(semi)Natural grassland	729	2
Public green space	6252	20
Other unpaved terrain	1059	3
River flood basin and salt marshes	413	1
Built up areas	579	2
Water (sea, lakes, rivers)	8336	26
Other	2	0
TOTAL	31.796	100

8. Comparison with earlier study

This study is to a large extent based on the method used in the paper by Daams et al. (2016). From that paper a popular internet version is presented at www.woningwaarde.natuur.nl (from here on ‘WWN’). How do these studies compare, and what explains the difference?
WWN indicates a total housing value of CANA of EUR 75 billion whereas the current study suggests 22.5 billion.

The studies use a similar hedonic pricing model but differ in a few important aspects, while it should be noted that, despite differences in methodologies and results, both studies yield estimates that are legitimate. Table 7 shows the main differences in methodologies.

Table 7: Main differences in methodologies between current study and Daams et al. (2016)

		Current study	Daams et al. 2016 study
1	Model	Hedonic model with km rings around property	Hedonic model with km rings around property
2	Level of spatial controls (spatial fixed effects)	Postal Code 4-digit	Housing submarket areas (Woningmarktregio’s NVM) and Postal Code 4-digit (in distinct models)
3	Type of house price data	WOZ (Assessed property values) *N* = 4,568,200 Years = 2013	Transaction data *N* = 203,344 Years = 2009-2012
4	Split models according to urbanity	5 levels of urbanity (CBS classification)	3 levels of urbanity (OECD classification)
5	Types of housing properties	Single family homes	Single family homes

Ad 1: The first things to compare are the similarity of the underlying models and the distance decay curve of impacts estimated in the current study and in Daams et al. (2016).

Ad 2&3: In table 8 and figure 5, we show the results for the current study (column 1 on the left) and the earlier study by Daams et al. (2016) using a similar type of model as in the current study (column 2a in the middle), and the main model in Daams et al. (2016) which is used in WWNhttp://www.woningwaardenatuur.nl/ (column 2b on the right).

The overall results of the current study appear to be lower here than in the Daams et al. study, but the distance effect is sustained. The main possible explanation is the use of assessed property values (in this study) instead of market transaction prices (Daams et al. study); assessed property values may not fully reflect the intangible benefits from CANA – which were moreover not quantified appropriately by studies before the Daams et al. (2016) study. The difference between the current study and the Daams et al. (2016) study results in column 2a are smaller than the difference with their results in column 2b, since 2a uses similarly fine-scale spatial controls as the current study, which may partially absorb the housing value that could otherwise be associated with proximity to CANA (Abbott and Klaiber, 2010; Daams et al., 2016).

Table 8: Percentage price impacts by distance interval, per study

	(1) current study	(2a) Daams et al.	(2b) Daams et al.
Spatial controls scale	PC4 (*N* = 4,005)	PC4 (*N* = 3,255)	Housing markets (*N*=76)
Distance (m) to CANA			
Within 0-500 m	6,7	12,9	16,0
Within 500-1000 m	4,2	8,5	10,7
Within 1000-2000 m	3,5	6,5	8,8
Within 2000-3000 m	3,0	6,1	6,3
Within 3000-4000 m	2,1	5,0	3,7
Within 4000-5000 m	1,1	4,1	3,5
Within 5000-6000 m	1,1	2,8	3,0
Within 6000-7000 m	0,7	2,6	1,6

Notes: the percentages in 2a are based on a first-differencing model similar to the main model in the current study; the percentages in 2b are those that are used in www.woningwaardenatuur.nl.

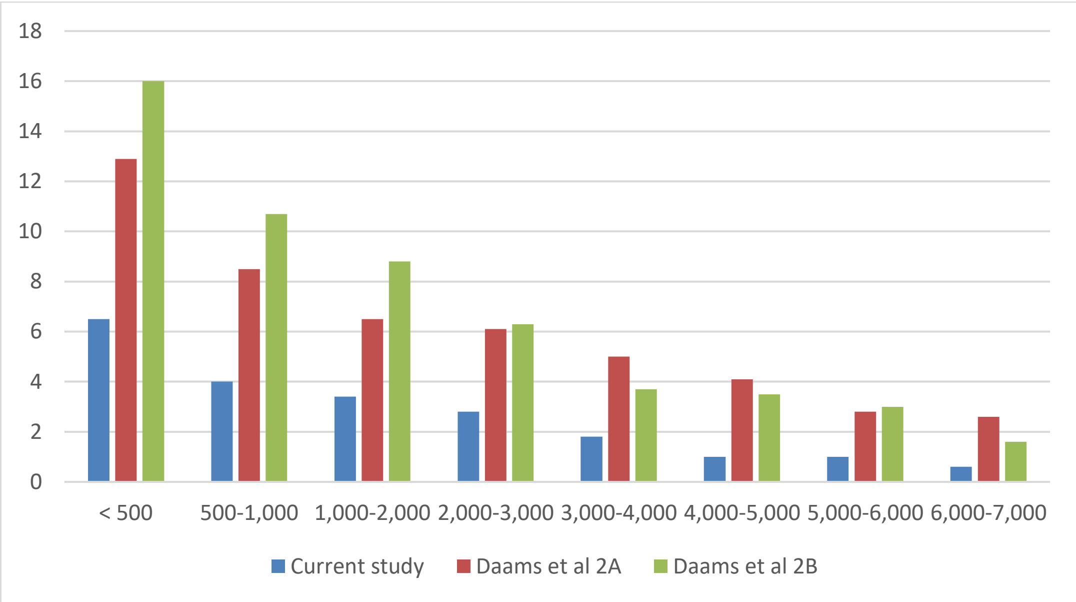


Figure 5: Visualization of the percentages presented in table 8.

Ad 4: With regard to differences in results, when the hedonic model is split by level of urbanization this relates to how this level is defined and using what spatial scale, as any distinct definition may come with different results, also if the modelling approach is held constant.

Ad 5: Finally, the overall amount of estimated housing value from (attractive) nature differs. The WWN estimates an aggregate CANA housing value of EUR 75 billion whereas the current study indicates EUR 22.5 billion. A key reason for the difference is the price level in the years of study. The price level is also different in the base years of the studies. WWN applies estimates from the Daams et al. (2016) study to 2012 WOZ data to estimate aggregate value, while the current study's estimate of this is based on 2013 WOZ data. Looking at the price index figure below one can see that this automatically leads to a lower absolute estimate of the aggregate value, since prices, and by association, WOZ-values in 2013 have been considerably lower than in 2012.

Also, the WWN estimate is for single family homes as well as apartments (assuming similarly sized impacts) – whereas apartments are not part of the current study's value aggregation.

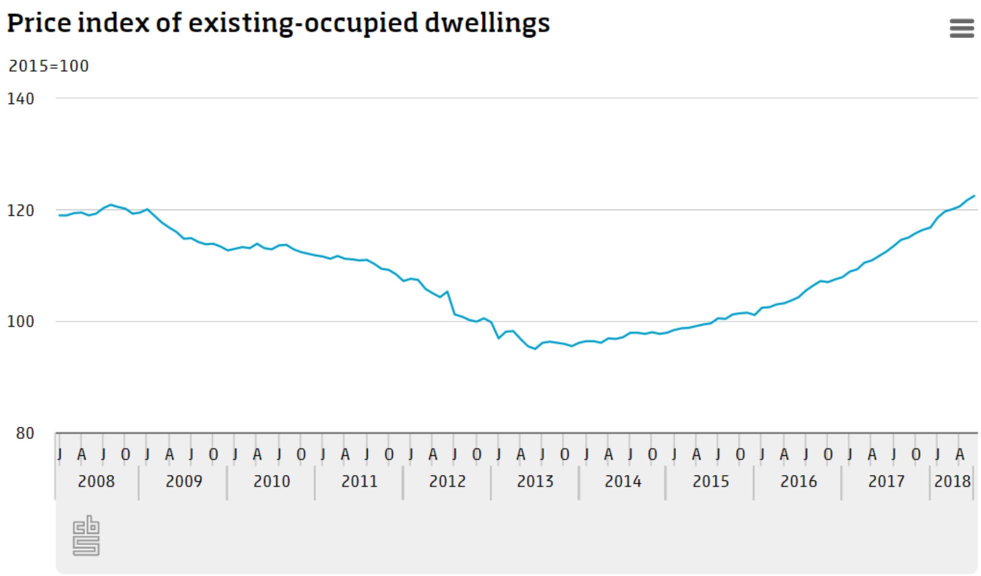


Figure 6: Price index for dwellings 2008-2018 (Source: https://www.cbs.nl/en-gb/news/2018/29/house-prices-continue-to-rise)

A further likely explanation is that intangible CANA value in the housing market may not be fully captured in assessed property (WOZ) values. These assessed property values are not the outcome of actual market behavior, but come from valuation procedures that may rely on computer-based models or, in some cases, site visits that considers a property or comparable homes that were recently sold. Hence, WOZ values may deviate from actual market value and may in particular be challenging to determine for homes for which there have been only few reference sales. This may apply in particular to (social or private) rental homes, which comprise about a quarter of this study's sample, whereas the full sample in Daams et al. (2016) consists of homes that transacted in the market. In this context, it is relevant to note that both studies find that the effect of nature stretches across 7 kilometers. This indicates that WOZ valuation procedures pick up on the value of nature in the (transaction-based) housing market. But, the smaller estimated effect-sizes raise the question of how well automated or visual inspection-based WOZ valuation procedures account for the added value of nature.

Overall, important is that the current study again confirms the 7 kilometer range of the impacts of CANA that was also reported in Daams et al. (2016). That underlines that nature impacts on the values of far more houses than was thought before. The size of the estimated impacts is, however, lower. The main reason for this is in differences between the studies (lower house values across the board in the current study's sample, the exclusion of apartments, and the use of assessed values instead of market prices) that are well-understood. However, since WOZ-based analysis likely underestimates the value of nature as compared to estimation from market prices, further research could investigate how well the added value of nature is accounted for in the valuation procedures that produce assessed property values (WOZ values) of houses.

9. Limitations, assumptions and sensitivity

Limitations in data

There are limitations with the data that is used in this research. Firstly, to the measure of closeness to nature. There are limitations to the definition of CANA, as was described in section 3. Additionally, defining ONA brings its limitations as the grid size of 100 by 100 meter and the cut-off of at least 80 percent nature are somewhat arbitrary. Regressions were also run on other grid sizes. However, it is believed that using 100 meter grid size makes a sufficiently granular distinction between CANA and ONA: where CANA creates economic benefits because of its recreational functions. ONA, which is very closely located to a dwelling, creates benefits because of green living area and maybe even view aspects.

Secondly, this research uses WOZ as a proxy for willingness to pay for a dwelling that is nearby nature. Transaction prices are more closely related to willingness to pay for nearby nature then WOZ. However, data for WOZ is available for all dwellings whereas transaction prices are only available for dwelling sold in a specific year. Besides, WOZ follows transaction prices, although with a lag. Therefore, it is assumed that WOZ as a proxy will suffice for this research.

Limitations in functional form

Regarding the functional form of the hedonic pricing model (2), there are some considerations to make. Most importantly, by using a first difference transformation only variation within PC4-areas are used in the regression analysis. Below is the correlation between mean WOZ of each PC4-area and mean distance to CANA and ONA. There is a negative correlation for both (see table 9), especially for CANA, meaning that on average larger PC4-areas are closer to nature areas. An explanation of this is that larger zip codes tend to be found in less densely populated areas, where CANA and ONA, in turn, are relatively more abundant than in more densely populated areas. This means that there might be an underestimation in the effects found in tables 3 and 4 for homes in more urbanized areas since smaller PC4-areas might absorb more of the CANA effect. On the other hand, it is possible that estimates for less urbanized areas are more prone to omitted variable bias since the spatial controls there are coarser than in more urbanized areas.

Table 9: Correlation matrix between mean WOZ-value and mean distance to nature area of PC4

	Mean WOZ-value	Mean distance to CANA	Mean distance to ONA
Mean WOZ-value	1.0000		
Mean distance to CANA	-0.2155	1.0000	
Mean distance to ONA	-0.1031	0.1795	1.0000

Another limitation to the functional form of model (2) is the concern of multicollinearity of using categorical variables of both CANA and ONA. Omitting either of these types of variables from the main model is not appropriate for two main reasons. First, this may cause omitted variable bias. Second, both sets of variables are of interest in this study. Therefore, we address the main concern with multicollinearity here: the stability of the estimates. To do so, the main model (2) is estimated in two additional specifications: one that omits the CANA variables and one that omits the ONA variables. Results for both these models give very similar results to those in table 3 (see table 10, columns 3a-3c). This suggests that multicollinearity due to inclusion of both ONA and CANA variables is of limited possible concern, as results for the main model remain stable.

The large value effects of the nearby CANA in very strong urbanized regions need to be further investigated. About 10 percent of the observations are located in these regions. As there are a few CANAs in Amsterdam (Vondelpark, Grachtengordel, het IJ) it could be that the results are skewed as they might pick up the relative high values of dwellings nearby Vondelpark and Grachtengordel in particular. Table 10 (columns 4a-4b) shows the results for the entire country and for very strongly urbanized areas excluding Amsterdam. The coefficients of interest remain stable as compared to table 3. This implies that the results are not driven in a concerning way by the relative high value of dwellings nearby CANA and ONA in Amsterdam.

Additionally, redistributing the value of each dwelling over the nature areas nearby is not as straightforward as it seems. By using the method described above (evenly spreading the value from CANA and ONA over all nature of these two kinds within a certain distance to the dwelling), we assume that people value the benefits from CANA and ONA homogenously.

Table 10: Regression results of robustness checks

Variables	(3a)	(3b)	(3c)	(4a)	(4b)
Living area (ln)	0.579***	0.580***	0.579***	0.579***	0.684***
	(0.000716)	(0.000716)	(0.000715)	(0.000716)	(0.00266)
Parcel size (ln)	0.0416***	0.0417***	0.0416***	0.0419***	0.0142***
	(0.000213)	(0.000213)	(0.000213)	(0.000214)	(0.000581)
Rent	-0.125***	-0.125***	-0.125***	-0.126***	-0.0836***
	(0.000370)	(0.000370)	(0.000370)	(0.000371)	(0.00103)
Construction year					
Before 1905	-0.154***	-0.154***	-0.154***	-0.154***	-0.0971***
	(0.00363)	(0.00362)	(0.00362)	(0.00363)	(0.0106)
Between 1906-1930	-0.156***	-0.156***	-0.156***	-0.156***	-0.0970***
	(0.00356)	(0.00356)	(0.00356)	(0.00357)	(0.0104)
Between 1931-1944	-0.126***	-0.126***	-0.126***	-0.126***	-0.0782***
	(0.00359)	(0.00359)	(0.00359)	(0.00360)	(0.0105)
Between 1945-1959	-0.152***	-0.152***	-0.152***	-0.152***	-0.124***
	(0.00357)	(0.00356)	(0.00356)	(0.00357)	(0.0106)
Between 1960-1970	-0.148***	-0.148***	-0.148***	-0.148***	-0.155***
	(0.00355)	(0.00354)	(0.00354)	(0.00355)	(0.0105)
Between 1971-1980	-0.125***	-0.125***	-0.125***	-0.125***	-0.141***
	(0.00355)	(0.00355)	(0.00355)	(0.00356)	(0.0105)
Between 1981-1990	-0.0818***	-0.0813***	-0.0818***	-0.0817***	-0.0916***
	(0.00355)	(0.00355)	(0.00355)	(0.00356)	(0.0104)
Between 1991-2000	-0.00636*	-0.00576	-0.00650*	-0.00607*	-0.0317***
	(0.00356)	(0.00356)	(0.00356)	(0.00357)	(0.0101)
Between 2001-2010	0.0128***	0.0129***	0.0126***	0.0129***	0.00128
	(0.00349)	(0.00349)	(0.00349)	(0.00350)	(0.00961)
End-of-terrace house	-0.274***	-0.274***	-0.273***	-0.273***	-0.295***
	(0.000589)	(0.000589)	(0.000588)	(0.000588)	(0.00321)
Semidetached house	-0.177***	-0.177***	-0.176***	-0.176***	-0.176***
	(0.000519)	(0.000520)	(0.000519)	(0.000519)	(0.00318)
Terraced house	-0.324***	-0.325***	-0.324***	-0.324***	-0.335***
	(0.000616)	(0.000616)	(0.000615)	(0.000615)	(0.00324)
Distance to nearest CANA					
Within 0-500 m		0.0703***	0.0644***	0.0646***	0.204***
		(0.00373)	(0.00372)	(0.00372)	(0.0120)
Within 500-1000 m		0.0443***	0.0411***	0.0408***	0.153***
		(0.00350)	(0.00348)	(0.00348)	(0.0111)

Variables	(3a)	(3b)	(3c)	(4a)	(4b)
Within 1000-2000 m		0.0361***	0.0344***	0.0343***	0.111***
		(0.00323)	(0.00322)	(0.00322)	(0.0102)
Within 2000-3000 m		0.0304***	0.0294***	0.0294***	0.0885***
		(0.00302)	(0.00300)	(0.00300)	(0.00892)
Within 3000-4000 m		0.0215***	0.0206***	0.0206***	0.0893***
		(0.00275)	(0.00274)	(0.00274)	(0.00788)
Within 4000-5000 m		0.0117***	0.0114***	0.0114***	0.0641***
		(0.00245)	(0.00244)	(0.00244)	(0.00673)
Within 5000-6000 m		0.0106***	0.0112***	0.0112***	0.0492***
		(0.00218)	(0.00217)	(0.00217)	(0.00505)
Within 6000-7000 m		0.00816***	0.00743***	0.00741***	
		(0.00173)	(0.00173)	(0.00173)	
Distance to nearest ONA					
Within 0-50 m	0.0492***		0.0482***	0.0482***	0.0421***
	(0.00120)		(0.00120)	(0.00121)	(0.00359)
Within 50-100 m	0.0350***		0.0342***	0.0341***	0.0341***
	(0.000803)		(0.000802)	(0.000805)	(0.00234)
Within 100-150 m	0.0211***		0.0204***	0.0203***	0.0273***
	(0.000689)		(0.000689)	(0.000691)	(0.00199)
Within 150-200 m	0.0142***		0.0135***	0.0134***	0.0216***
	(0.000636)		(0.000635)	(0.000636)	(0.00181)
Within 200-250 m	0.00832***		0.00779***	0.00778***	0.0118***
	(0.000596)		(0.000596)	(0.000597)	(0.00168)
Within 250-300 m	0.00462***		0.00421***	0.00419***	0.00617***
	(0.000559)		(0.000559)	(0.000561)	(0.00145)
Within 300-350 m	0.00153***		0.00127**	0.00124**	
	(0.000498)		(0.000498)	(0.000499)	
Constant	-6.02e-07	-3.97e-07	-6.48e-07	1.53e-08	-2.98e-05
	(0.000125)	(0.000126)	(0.000125)	(0.000126)	(0.000418)
Observations	4,564,195	4,564,195	4,564,195	4,521,899	425,609
R-squared	0.709	0.708	0.709	0.710	0.691
Adjusted R-squared	0.709	0.708	0.709	0.710	0.691
F statistic	98134	93430	72146	72247	4296
Root MSE	0.176	0.176	0.176	0.176	0.151

Note: Dependent variable is the natural log (ln) of WOZ-value. All variables are differenced with a random house within the same zip code area (PC4). The reference categories include construction year after 2010; detached house; not rented. Specifications (3a) - (3c) include only ONA, CANA and both, respectively. Specification (4) excludes observations for Amsterdam, respectively in the national dataset (4a) and for urbanization level 1 (4b). Clustered standard errors are in parentheses. *** p<0.01, ** p<0.05, * p<0.1.

10. Conclusion and discussion

This study has considered how ecosystems are implicitly valued in the Dutch housing sector, which is the country's largest economic source of material wealth in which nature adds value. From the total value of single-family homes, approximately EUR 31 billion can be attributed to the presence of nature. Of this estimated added value, 70% comes from the most highly appreciated natural places and 30% from other natural areas. This indicates that the ecosystem services that are provided by Dutch nature are highly valued by the users and owners of houses – on the condition that this nature is close to people's homes and especially when perceived attractive.

As is the case with all ecosystem services, amenity services are only provided where the benefits are (potentially) used, meaning only in locations where people live. This is also shown in figure 4, where it is clear that nature the Randstad receive higher values because most people live there.

At this stage it is important to note to what this research includes and what not. First of all, it is only the value of nature that is reflected in the value of primary houses. Nature has an impact on people's well-being (Davis et al., 2016). In assessing the value of that impact housing value is very powerful but it does not capture all well-being (Sijtsma et al., 2012). Especially the value of nature that is further away from primary living houses will not be reflected here. Furthermore, various non-use values will not be reflected in the current measurements. Even regarding to housing this study does not capture all value, since the value for recreational homes is not assessed.

Second, only single-family dwellings are taken into account. It is believed that apartments also entail economic benefits toward nearby nature. However, this is more complex to measure as apartments have a different price structure and the relationship toward nearby nature is more driven by aesthetic aspects to view.

Third, in terms of nature agricultural land is not taken into account fully. However, one seventh of the total area of CANAs is represented by agricultural land; agricultural land may indeed make part of attractive nature areas as it may spatially blend in between patches of other land uses that are (also) perceived attractive.

Fourth, it is rather difficult to define what is nature and what not. We have attempted this by including both CANA and ONA, however other definitions could also be used. For example, what is not included in our definition of nature areas are avenue trees even though it could be that these produce economic benefits to the dwellings next to it. The methodology of Greenmapper database is completely fit for identifying this type of nature, but this more linear shapes and (merely) locally valued types of nature, require a more extensive dataset (>1 percent of Dutch population, i.e. 200,000 respondents instead of the current 8,000).

Fifth, the aspect of view on nature is not taken into account. Daams et al. (2016) show that having a view on nature areas increases the property price by an additional 7 percent. Our model could be extended by incorporating a variable that describes view.

Finally, what is important to note is that some areas are identified as CANA within highly urbanized areas; areas without a formal nature protection status. Take for example the Canals in Amsterdam. They are perceived as attractive nature by the respondents to the Greenmapper survey. Obviously the nature here is closely connected with a combination of other (urban) characteristics of the area. Which goes to show that what is perceived as nature is subjective and dealing with this is complex, while it also shows that nature is not always related to ecological quality (Davis et al. 2016).

Additionally, it should be noted that in the definition of nature areas used it is not accounted for accessibility of the nature areas. Although not many nature areas in the Netherlands are non-accessible, accessibility is not solely necessary for the ecosystem services derived from the nature areas (such as enjoyment of clean air). However, it could be necessary for other ecosystem services (such as recreation).

References to section 3.8

Abbott, J. K., & Klaiber, H. A. (2011). An embarrassment of riches: Confronting omitted variable bias and multi-scale capitalization in hedonic price models. Review of Economics and Statistics, 93(4), 1331-1342.

Anderson, S. T., & West, S. E. (2006). Open space, residential property values, and spatial context. Regional Science and Urban Economics, 36(6), 773-789.

Bark, R. H., Osgood, D. E., Colby, B. G., & Halper, E. B. (2011). How do homebuyers value different types of green space?. Journal of Agricultural and Resource Economics, 395-415.

Bijker R.A. and Sijtsma F.J. (2017). A portfolio of natural places: Using a participatory GIS tool to compare the appreciation and use of green spaces inside and outside urban areas by urban residents. Landscape and Urban Planning 158 (2017) pp. 155-165. http://dx.doi.org/10.1016/j.landurbplan.2016.10.004

Czembrowski, P., Kronenberg, J., & Czepkiewicz, M. (2016). Integrating non-monetary and monetary valuation methods–SoftGIS and hedonic pricing. Ecological Economics, 130, 166-175.Daams, M.N., Sijtsma, F.J. and Van der Vlist, A.J. The effect of natural space on nearby property prices: accounting for perceived attractiveness. *Land Economics*, 92(3), 389-410.

Daams, M.N, Sijtsma, F.J. & Van der Vlist, A.J (2016). The effect of natural space on nearby property prices: accounting for perceived attractiveness. Land Economics, 92(3), 389-410.

Davis, N., Daams, M., Sijtsma, F., & van Hinsberg, A. (2016). How deep is your love - of nature? A psychological and spatial analysis of the depth of feelings towards Dutch nature areas. Applied Geography, 77, 38-48. http://dx.doi.org/10.1016/j.apgeog.2016.09.012

Diewert, W. E. (2003). Hedonic regressions: A review of some unresolved issues. In 7th Meeting of the Ottawa Group, Paris, May (Vol. 29).

Von Graevenitz, K., & Panduro, T. E. (2015). An alternative to the standard spatial econometric approaches in hedonic house price models. Land Economics, 91(2), 386-409.

Gibbons, S., Susana M., and Guilherme R.M. (2014). The Amenity Value of English Nature: A Hedonic Price Approach. Environmental and Resource Economics 57 (2): 175–96.

De Groot, R. S., Wilson, M. A., & Boumans, R. M. (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecological Economics, 41(3), 393-408.

Guo, X., Zheng, S., Geltner, D., & Liu, H. (2014). A new approach for constructing home price indices: The pseudo repeat sales model and its application in China. Journal of Housing Economics, 25, 20-38.

De Haan, J., & Diewert, W. E. (2011). *Handbook on residential property price indexes*. Luxembourg: Eurostat.

Ham, C., Champ P.A., Loomis J.B., and Reich R.M., (2012). Accounting for Heterogeneity of Public Lands in Hedonic Property Models. Land Economics 88 (3): 444–56.

Lancaster, K. J. (1966). A new approach to consumer theory. Journal of Political Economy, 74(2), 132-157.

Van Leeuwen, N., Zuurmond, M., de Jong, R. (2017). Ecosystem Unit map, product description. CBS: Den Haag/Heerlen, The Netherlands.

Liebelt, V., Bartke, S., & Schwarz, N. (2018). Revealing preferences for urban green spaces: a scale-sensitive hedonic pricing analysis for the city of Leipzig. Ecological Economics, 146, 536-548.

Luttik, J., (2000). The Value of Trees, Water and Open Space as Reflected by House Prices in the Netherlands. Landscape and Urban Planning 48 (3): 161–67.

McConnell, V., and Walls, M. (2005). The Value of Open Space: Evidence from Studies of Non-market Benefits. Washington, DC: Resources for the Future.

Melichar, J. & Kaprová K. (2013). Revealing Preferences of Prague's Homebuyers toward Greenery Amenities: The Empirical Evidence of Distance–Size Effect. Landscape and Urban Planning 109 (1): 56–66.

Millennium Ecosystem Assessment (MEA), 2005. Ecosystems and Human WellBeing: Synthesis. Island Press, Washington, DC.

OECD (2017). The governance of land use in OECD countries: policy analysis and recommendations. OECD Publishing, Paris.

Palmquist, R.B. (2005). Property value models. In Handbook of Environmental Economics, Vol. 2. ed. Karl Göran-Mähler and Jeffrey R. Vincent, 763–819. North Holland, Amsterdam.

Panduro, T. E, & Veie, K. L. (2013). Classification and valuation of urban green spaces: a hedonic house price valuation. Landscape and Urban Planning, 120, 119–128.

Pendleton, L. H., & Shonkwiler, J. S. (2001). Valuing bundled attributes: a latent characteristics approach. Land Economics, 77(1), 118-129.

Rosen, S. (1974). Hedonic Prices and Implicit Markets: Product Differentiation in Pure Competition. Journal of Political Economy 82 (1): 34–55.

Schläpfer, F., Waltert, F., Segura, L., & Kienast, F. (2015). Valuation of landscape amenities: a hedonic pricing analysis of housing rents in urban, suburban and periurban Switzerland. Landscape and Urban Planning, 141, 24-40.

Scholte, S.S.K., Daams, M.N., Farjon, H., Sijtsma, F.J., van Teeffelen, A.J.A., and Verburg, P.H. (2018). Mapping recreation as an ecosystem service: considering scale, interregional differences and the influence of physical attributes. Landscape and Urban Planning, 175 (2018), pp. 149-160. https://doi.org/10.1016/j.landurbplan.2018.03.011

Sijtsma, F.J., Daams M.N., Farjon H. and Buijs A.E., (2012). Deep feelings around a shallow coast. A spatial analysis of tourism jobs and the attractivity of nature in the Dutch Waddenarea. Ocean and Coastal Management, 68 (2012), November, pp138-148. DOI:http://dx.doi.org/10.1016/j.ocecoaman.2012.05.018

Sijtsma, F.J., Farjon, H., van Tol, S., van Hinsberg, A., van Kampen, P., and Buijs, A. (2013). Evaluation of landscape changes - Enriching the economist's toolbox with the Hotspotindex. In: W. Heijman, & C. M. J. v. d. Heide (Eds.), The Economic Value of Landscapes. Chapter 8, pp 136-164. London: Routledge.

Statistics Netherlands (2013). Demografische kerncijfers per gemeente 2013. Retrieved from https://www.cbs.nl/nl-nl/publicatie/2013/49/demografische-kerncijfers-per-gemeente-2013

Statistics Netherlands (2017). Ecosystem units map. Retrieved from https://www.cbs.nl/en-gb/background/2017/12/ecosystem-unit-map.

Statistics Netherlands (2018). Eigendom woningvoorraad. Retrieved from https://www.cbs.nl/nl-nl/onze-diensten/methoden/onderzoeksomschrijvingen/korte-onderzoeksbeschrijvingen/eigendom-woningvoorraad).

Statistics Netherlands (2019). Capital stocks; SIC; type of capital assets, National Accounts. Retrieved from http://opendata.cbs.nl/statline/#/CBS/en/dataset/84328ENG/table?ts=1551340947930

De Vries, S., A. Buijs, F. Langers, H. Farjon, A. van Hinsberg, F.J. Sijtsma (2013). Measuring the attractiveness of Dutch landscapes: identifying national hotspots using Google Maps. Applied Geography. Volume 45, December 2013, Pages 220-229. (5-year Impact Factor 3.4)

Waltert, F., & Schläpfer, F. (2010). Landscape amenities and local development: a review of migration, regional economic and hedonic pricing studies. Ecological Economics, 70(2), 141-152.

Xiao, Y., Li, Z., & Webster, C. (2016). Estimating the mediating effect of privately-supplied green space on the relationship between urban public green space and property value: evidence from Shanghai, China. Land Use Policy, 54, 439-447.