PERSPECTIVES AND SPATIAL APPROACHES FOR QUANTIFYING ECOSYSTEM SERVICES

M.J. PAULIN

Perspectives and spatial approaches for quantifying ecosystem services

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Perspectives and spatial approaches for quantifying ecosystem services

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Table of Contents

1

CHAPTER 1

General introduction

1.1 Nature and humans

The natural system endows humans with natural resources and processes that are fundamental for meeting their basic needs for survival and well-being. It could be argued that people were aware of their dependence on natural systems as far as ancient civilisations, when observers noted that human actions disrupt natural systems and their capacity to benefit humans (Costanza et al., 2017; Gómez-Baggethun et al., 2010). Early economic thinkers found value in natural systems' benefits to humans, in particular, due to their free nature (Gómez-Baggethun et al., 2010; Smith (1776), 1909; Ricardo, 1817; Marx (1891), 2008). This changed in the turn of the 19th century, as money increasingly became recognised as a central metric for social welfare (Gómez-Baggethun et al., 2010; Pigou (1920), 2006). As the scope of conventional economic analysis shifted to previously-marketed goods and services, natural systems' non-marketed benefits were gradually forgotten (Gómez-Baggethun et al., 2010). What came about was an economic system that venerates the maximisation of natural resource production while minimising labour and capital costs.

Fuelled by theories such as the Ricardian theory of comparative advantage (Ricardo, 1817) and the factor-proportions theory (Heckscher, 1919; Ohlin, 1933; Samuelson, 1948), economies began to specialise in the production of goods and services for which they had a comparative advantage. This led to higher productivity and access to a wider variety of goods and services for people across the globe (Johansson & Olaberría, 2014). For long, the environmental externalities associated with natural resource extraction and use were eclipsed by the benefits to social welfare that greater and more diverse consumption possibilities entail. As people's understanding of their relationship with natural systems languished, environmental degradation peaked to unprecedented levels (MA, 2005). Throughout history, humans had successfully devised innovative solutions to deal with the hardships they confronted, with technological innovations playing a central role in this process. However, technical solutions no longer sufficed to compensate for the negative environmental impacts that humans had brought upon themselves.

The forgotten benefits provided by natural systems reclaimed broad public attention in the 1960s, with the establishment of environmental science. One of the leading influences on this development included the publication of Rachel Carson's *Silent Spring* (Carson, 1962). Carson's book shed light on the pervasive ways in which humans alter their natural environment and how this often results in devastating consequences for the well-being of current generations, while jeopardising the well-being of future generations. Environmental science emerged as a response to environmental externalities and their negative effect on society that came about as a consequence of perceived solutions to time-specific problems (e.g., industrialisation, specialisation, agrochemical use). Despite the expanding body of knowledge on the many mechanisms that underpin human-nature interactions, tackling the threats to human well-being posed by environmental degradation remains a difficult challenge. The ecosystem services concept is instrumental for reifying the complex mechanisms through which natural systems provide benefits to humans. As such, it could serve as a powerful tool to support the sustainable management or Earth's natural systems in a way that meets the needs of current and future generations.

1.2 Ecosystem services

Ecosystem services are the direct and indirect contributions by natural capital (i.e., Earth's ecosystems and underpinning geophysical systems; Figure 1.1) to human well-being (Haines-Young & Potschin, 2018). While the concept has been around for decades (Westman, 1977; Ehrlich & Ehrlich, 1981; Ehrlich & Mooney, 1983; de Groot,1987), its mainstream use did not take place until the 1990s. This partly resulted from a growing body of interdisciplinary research that laid the initial foundations that made their assessment possible (de Groot, 1987; Braat, 1992; de Groot, 1992; Costanza & Daly, 1992; Perrings et al., 1992; Daily, 1997; Costanza et al., 1997). The recognition of ecosystem services within policymaking gained momentum after the release of the Millennium Ecosystem Assessment (MA) in 2005 (MA, 2005). The MA synthesised the results of hundreds of studies from various disciplines to assess the state and trends of global ecosystems, as well as their implications for human well-being. Gains and losses to human well-being were measured in terms of changes in ecosystem service values that have resulted from anthropogenically induced changes to ecosystems. The MA (2005) revealed that in the latter half of the twentieth century humans changed ecosystems more rapidly and extensively than in any other comparable period of time. While changes to ecosystems have resulted in net gains to well-being, this has taken place at the expense of ecosystem degradation and the exacerbation of poverty in regions that did not benefit from this process (MA, 2005). The MA (2005) concluded that, unless addressed, these problems could substantially diminish people's capacity to benefit from ecosystems in the future.

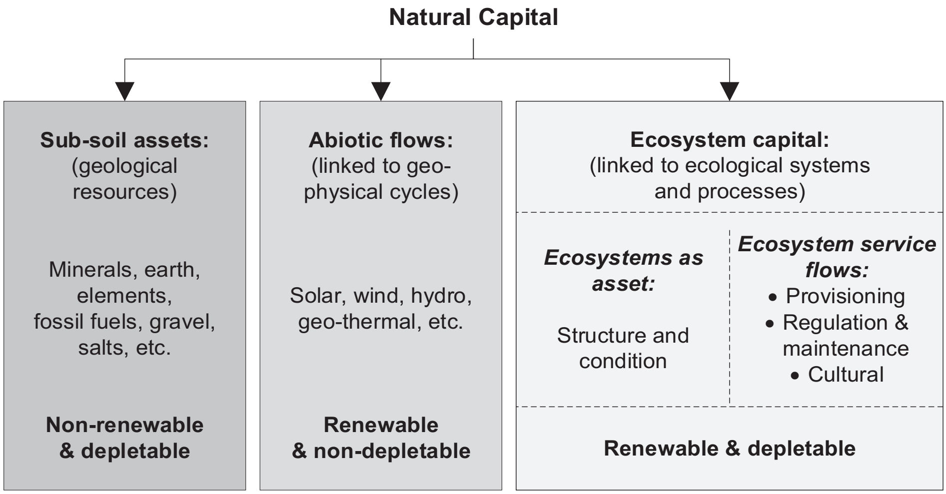

Figure 1.1: Natural capital and its components (credit/adapted from: Maes et al., 2013)

In the years ensuing the MA, the ecosystem services concept has been increasingly recognised as a promising instrument to tackle the challenge of ecosystem degradation and its implications for human well-being. Leading initiatives that fuelled this process included the publication of The Economics of Ecosystems and Biodiversity (TEEB, 2010), as well as the development of the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES; Pascual et al., 2017) and the Ecosystem Services Partnership (ESP; https://www.es-partnership.org/).

1.3 Operationalisation

Ecosystem services are highly complex in character, covering a wide range of value domains (i.e., values commonly regarded under distinct disciplines, such as biophysical, economic, and sociocultural values). Given their complexity, their spatial quantification can pose a difficult task. Recognising this challenge, a plurality of frameworks for operationalising ecosystem services and the delivery process has emerged throughout the years ensuing the MA. The delivery process is the process that underpins ecosystem service production and use, ranging across the biophysical, sociocultural, and economic spectra. Operationalisation can be described as the process by which concepts are made usable by practitioners (e.g., assessors, decision-makers; Potschin et al., 2014). In a broad sense, the aim of ecosystem service operationalisation frameworks is to bridge the gap between the use of the ecosystem services concept in theory and in practice (Carmen et al., 2018). They contribute to this aim by providing a consistent structure for assessing ecosystem services and by facilitating the communication of assessment results to end users (Carmen et al., 2018; Breure et al., 2012). Operationalisation frameworks have primarily focused on the operationalisation of ecosystem services into sections and of the delivery process into delivery components.

1.3.1 Sections

The MA (2005) operationalised ecosystem services into four sections, namely groups of services that share similar characteristics (https://cices.eu). Sections included: provisioning, regulating, cultural, and supporting services. This classification system became the backbone of numerous subsequent operationalisation frameworks, leading to its various adaptations (TEEB, 2010; Landers & Nahlik, 2013; Pascual et al., 2017; Haines-Young & Potschin, 2018). Currently, the Common International Classification of Ecosystem Services (CICES; Haines-Young & Potschin, 2018) classifies services into provisioning, regulation and maintenance (formerly 'regulating'), and cultural services. Provisioning services are the material contributions that natural capital provides to people, such as food consumed for nutritional purposes (MA, 2005). Regulation and maintenance services are ecological processes that directly or indirectly contribute to human well-being (MA, 2005). For instance, the retention of pollutants by vegetation in cities can contribute to urban health and consequently to reductions in health costs at the societal level (Remme et al., 2018; Janhäll, 2015). Cultural services are the non-material benefits that ecosystems provide to humans, such as spiritual or recreational values that people associate with natural landscapes or elements (MA, 2005).

1.3.2 Delivery components

Recognising the plurality of mechanisms by which natural systems provide benefits to humans, various frameworks have been developed for operationalising the delivery process into delivery components (Potschin & Haines-Young, 2016; Maes et al., 2015a; Vihervaara et al., 2019; Villamagna et al., 2013; Seppelt et al., 2012; Crossman et al., 2013; Johnson et al., 2010). Delivery components are the ecological and socioeconomic building blocks that together form the delivery process (Villamagna et al., 2013). Below, two well-established approaches for operationalising the delivery process are described.

Cascade framework

The cascade framework operationalises the delivery process into different stages that span across the biophysical and anthropogenic realms (de Groot et al., 2002; Potschin & Haines-Young, 2016; Figure 1.2). The cascade depicts the delivery process as flowing from ecological structures and processes, to ecosystem functions, to the benefits and values they generate for humans. Ecosystem functions are natural resources and processes that contribute to human well-being (de Groot et al., 2002). Together, ecosystem functions and accrued socioeconomic benefits and values constitute ecosystem services (Costanza et al., 2017). The cascade epitomises the interwoven relationship that is shared by the natural and the socioeconomic systems. It provides a blueprint for assessing the dichotomous yet interconnected elements that constitute the delivery process.

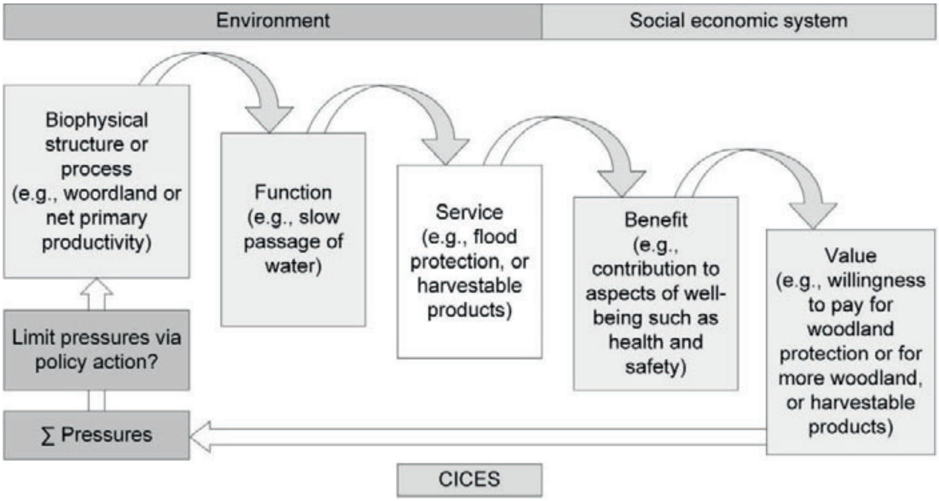

Figure 1.2: Ecosystem service cascade (credit/adapted from: Potschin & Haines-Young, 2016)

Supply and demand

Ecosystem services are scarce, given the rising human population and the restricted nature of Earth's boundaries. As the global population continues to increase, so does the demand for ecosystem services required to meet human needs. This growing demand is often difficult to meet, given the decreasing amount of space available to support ecosystem services. This results in trade-offs between ecosystem services. Trade-offs occur when the delivery of a particular service results in lost opportunities for the delivery of other services (i.e., opportunity costs; Turkelboom et al., 2018). Due to the existence of trade-offs, changes in the distribution of ecosystem services inevitably lead to societal gains and losses, despite potential net gains to society. Hence, an optimal (i.e., equitable, sustainable, efficient; Schröter et al., 2017) ecosystem service allocation requires the maximisation of gains and the minimisation losses across different sectors of society and throughout current and future generations. This can be achieved by ensuring that spatial mismatches between the demand for and supply of ecosystem services are minimised throughout time. Mismatches occur when the demand for a particular ecosystem service is not met by its supply in a particular point in time or space (Baró et al., 2015). Given the need for an optimal ecosystem service allocation to support human well-being, a plurality of frameworks has been developed for operationalising the supply side (i.e., ecological) and the demand side (i.e., anthropogenic) of ecosystem service delivery (e.g., Burkhard et al., 2012; Villamagna et al., 2013; Bagstad et al., 2013a; Wolff et al., 2015). Within these frameworks, supply and demand delivery components are conceptualised in a variety of ways, leading to their ambiguous interpretation (Villamagna et al., 2013).

1.4 Mapping

The distribution of ecosystem services is complex in character (Baró et al., 2015). The urban-rural and local-global gradients are characterised by continuously changing landscape structures, land-uses, climates, administrative structures, and demographic variability (Larondelle & Haase, 2013; Martín-López et al., 2012; Schram-Bijkerk, 2018). This results in variations in the distribution of areas where ecosystem service supply, demand, and actual use take place (Keeler et al., 2019). Recognising their spatial complexity, ecosystem services are often quantified spatially, their distribution visualised with the aid of

1

maps. To facilitate the process of spatially quantifying ecosystem services, a number of tools have been developed throughout the last decade (e.g., InVEST, ARIES, MAES, ESTIMAP; Tallis and Polaski, 2011; Villa et al., 2014; Maes et al., 2015a; Zulian et al., 2014). Their aim is to contribute to the standardisation and harmonisation of the assessment process, supporting the comparability of results across space and time. Therewith, these approaches seek to facilitate the assessment process to endorse the consideration of ecosystem services within decision-making. While useful at relatively large scales, well-established standardised ecosystem service models often make use of generalised spatial data (e.g., land-cover, remote-sensed data, spatially extrapolated field observations) and non-spatial data (e.g., from statistics, publications, field samples) as input, limiting their use to support decision-making at the subnational (e.g., local, regional) scales.

1.5 Decision-making

The ecosystem services concept can serve as a language through which science communicates to society, shedding light on the intricate mechanisms by which ecosystems generate value for humans (Potschin & Haines-Young, 2016; Breure et al., 2012). Recognising the concept's potential to inform decision-makers, a number of initiatives have called for its integration within policymaking (e.g., the Convention on Biological Diversity (CBD; 2010), Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES; Pascual et al., 2017), and UN Sustainable Development Goals (SDGs; UN, 2017). Under the Paris Climate Accord, 137 countries agreed to CBD Aichi Biodiversity Target 14, calling for the protection and enhancement of ecosystem services (CBD, 2010). Under the EU 2020 Biodiversity Strategy, EU Member States agreed to map and assess ecosystem services within national territories and to promote and integrate these values into national accounting and reporting (EC, 2011). To meet these targets, country-wide initiatives have emerged to support the integration of ecosystem services within policymaking at the national and subnational levels (UK NEA, 2011; EME, 2012, 2014; NOU, 2013; EZ, 2013a, b; Rugani et al., 2014). To aid countries in this difficult challenge, the System of Environmental-Economic Accounting (SEEA), developed under the auspices of the United Nations, acts as an international statistical standard and guideline for integrating ecosystem services within national accounting (UN, 2014). Despite these initiatives, consideration of the ecosystem services concept within decision-making at the local and regional (subnational) levels remains limited (Haase et al., 2014; Walsh et al., 2015). In this thesis, three factors that limit the integration of the ecosystem services concept within decision-making are addressed (Sections 1.5.1 - 1.5.3).

1.5.1 Limited consideration of spatial and thematic detail

Despite their practicality, standardised ecosystem service models often make use of generalised spatial and non-spatial data as input, leading to an oversimplification of the cross-scale heterogeneity that defines spatiotemporal gradients (Derkzen et al., 2015; Martínez-López et al., 2019). In doing so, they often sacrifice the spatial and thematic detail required to support the needs and expectations of decision-makers at subnational (e.g., local, regional) levels (Martínez-López et al., 2019; Hauck et al., 2013). To account for spatial and thematic detail relevant to the site under assessment, scale- and time-dependent harmonisation of ecosystem service models is needed, recognising the very real variations that define spatiotemporal gradients (Martínez-López et al., 2019; Hauck et al., 2013).

1.5.2 Ambiguous operationalisation of the delivery process

Today, a number of frameworks facilitate the spatial quantification of delivery components spanning across distinct value domains. Despite progress in this regard, substantial ambiguity in the operationalisation of the delivery process limits the consistent assessment of its delivery components (Villamagna et al., 2013; Luederitz et al., 2015; Wei et al., 2017). This reduces the comparability of assessment output across space and time and reduces confidence in output by end users (Villamagna et al., 2013; Luederitz et al., 2015; Wei et al., 2017). To add credibility to ecosystem service assessments, a flexible yet clear and consistent approach for operationalising the delivery process is needed.

1.5.3 Limited consideration of value domains

Ecosystem services and the delivery process span across different value domains. While ecosystem functions can be expressed in biophysical terms, the benefits they generate can be expressed as sociocultural and economic values. Despite general agreement regarding the need to engage with different aspects of the cascade within assessments, including both the supply side and demand side of delivery, in practice this rarely takes place (Villamagna et al., 2013; Luederitz et al., 2015; Wei et al., 2017; Lautenbach et al., 2019; Burkhard et al., 2012; Wolff et al., 2015). For instance, ecosystem services have been mainly assessed in biophysical terms, with regulation and maintenance services receiving a disproportionate degree of representation across the scientific literature (Maes et al., 2012; Martinez-Harms & Balvanera et al., 2012; Crossman et al., 2013; Luederitz et al., 2015; Malinga et al., 2015). Meanwhile, cultural services and local perceptions have been largely overlooked, disregarding the central role that culture plays on human-nature interactions (Diaz et al., 2018; de Groot et al., 2018; Maes et al., 2018; Peterson et al., 2018). To add legitimacy to assessments, a more holistic consideration of value domains is needed.

1.6 Thesis aim and research questions

The aim of this thesis is to develop approaches to facilitate the spatial quantification of ecosystem services at the subnational level. In particular, approaches are developed to tackle issues related to the ambiguous operationalisation of the delivery process, the limited consideration of distinct value domains, and the limited consideration of spatial and thematic detail within spatial assessments. Tackling these issues could lead to better uptake of the ecosystem services concept within decision-making at the local and regional (subnational) levels by adding consistency, credibility, and legitimacy to assessments. It is hypothesised that the assessment of ecosystem services in a consistent manner, considering location-specific spatial and thematic detail, and covering distinct value domains, is instrumental for informing decision-makers who wish to optimise the distribution of natural capital to support human well-being. The aim of this thesis is addressed by evaluating the following research questions (RQ):

RQ 1) How can existing approaches for operationalising the delivery process be harmonised to endorse clarity and consistency within ecosystem service assessments?

RQ 2) How can distinct value domains (i.e., biophysical, sociocultural, economic) be integrated within ecosystem service assessments?

RQ 3) How can ecosystem services be quantified at high spatial and thematic detail and across distinct value domains?

RQ 4) How can approaches for spatially quantifying ecosystem services be implemented to inform decision-making at the subnational (i.e., local, regional) level?

1.7 Thesis outline

This Section provides a brief overview of the subjects covered in Chapters 2 through 6, as well as how posed RQ are addressed in each Chapter. Figure 1.3 presents a schematic diagram displaying the output generated by Chapters 2 through 5, approaches adopted to obtain specified output, and the RQ that each type of output contributes to answering.

Chapter 2: 'Review: Ecosystem service operationalisation, conceptualisation, and mapping across value domains'

Addressing RQ 1, this Chapter seeks to provide clarity and consistency regarding the operationalisation of the delivery process across value domains (i.e., biophysical, sociocultural, economic domains). For clarity, common approaches for operationalising ecosystem services into delivery components are reviewed. For consistency, based on existing approaches, a framework for operationalising and conceptualising delivery components is presented. Addressing RQ 2, this Chapter also evaluates the state of the art regarding the consideration of distinct value domains across ecosystem service mapping studies. To achieve this, a systematic review has been performed to analyse the frequency with which ecosystem service sections and delivery components are mapped within scientific publications.

Chapter 3: 'Towards nationally harmonised mapping and quantification of ecosystem services'

Contributing to RQ 3, this Chapter presents the Natural Capital Model (NC-Model), a spatially explicit set of models for mapping and quantifying ecosystem services within the Netherlands. The aim of the NC-Model is to support the integration of ecosystem services within spatial planning and decision-making in the Netherlands, contributing to the fulfilment and tuning of local, regional (subnational), national, and international environmental policy targets. Accounting for spatial detail, models can be applied to spatially quantify ecosystem services across the urban-rural gradient and at a high resolution (10 x 10 m). Accounting for thematic detail across value domains, models can be applied to assess the supply side and the demand side of ecosystem services, capturing biophysical, sociocultural, and economic aspects of the delivery process. For demonstration purposes, six urban ecosystem service models comprised within the NC-Model are presented and applied within the Municipality of Amsterdam.

Chapter 4: 'Application of the Natural Capital Model to assess changes in ecosystem services from changes in green infrastructure in Amsterdam'

The high level of spatial and thematic heterogeneity that defines the urban landscape suggests that a universal toolkit for assessing the value of urban nature is unlikely to occur (Keeler et al., 2019). Addressing RQ 4, this Chapter presents an application of the NC-Model to assess the effect of changes in green infrastructure (i.e., soil, vegetation, and water; sometimes denoted as green and blue infrastructure) on ecosystem services in the Municipality of Amsterdam, the Netherlands. This assessment was performed to inform decision-makers involved in the development of the Green Quality Impulse (*KwaliteitsImpuls Groen*; Amsterdam Municipality, 2017a), a spatial plan for the expansion and improvement of Amsterdam's green infrastructure by the year 2025.

Chapter 5: 'Integration of local knowledge and data for spatially quantifying ecosystem services in the Hoeksche Waard, the Netherlands'

Contributing to RQ 3, this Chapter presents an approach for incorporating local knowledge and data (i.e., biophysical, sociocultural, economic data) within ecosystem service assessments, integrating well-established spatial quantification methods. Incorporation of local knowledge and data within assessments is necessary for capturing site-specific sociocultural and ecological factors that influence the supply and actual use of ecosystem services (Díaz et al., 2018). Contributing to RQ 4, the approach is implemented for mapping and quantifying seven ecosystem services in the rural Municipality of the Hoeksche Waard, the Netherlands. The Hoeksche Waard is a particularly interesting study site, given notable interest by local stakeholders to collaborate to meet their common objectives.

Chapter 6: Synthesis

This Chapter synthesises the key findings of this thesis and discusses how RQ posed are addressed in each Chapter. Based on this information, the methodological advances of this thesis and its relevance in a decision making context are synthesised. Finally, remaining challenges that require attention to endorse the integration of ecosystem services within decision-making are presented. Based on these challenges, recommendations for future research are provided.

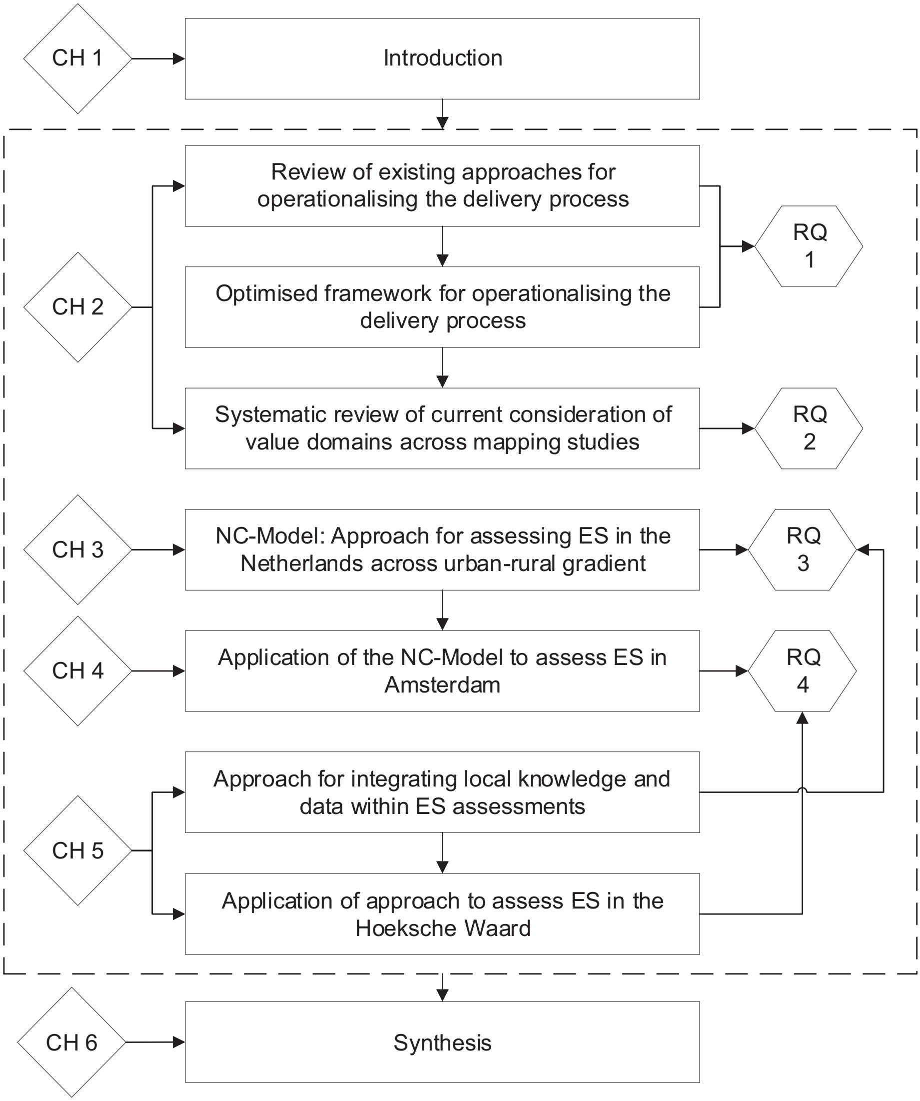

Figure 1.3: Schematic overview of Chapters in this thesis and the research questions they address. CH: chapter; RQ: research question; ES: ecosystem service(s); NC-Model: Natural Capital Model.

2

CHAPTER 2

Review: Ecosystem service operationalisation, conceptualisation, and mapping across value domains

Martina J. Paulin, Aalbert J. Hendriks, Michiel Rutgers,
Ton de Nijs, Anton M. Breure

Ecosystem Services (submitted).

Abstract

Limited uptake of the ecosystem services concept within decision-making often results from limited consideration of distinct value domains across assessments, as well as ambiguity in the operationalisation of the delivery process and conceptualisation of its components. This paper evaluates the current consideration of distinct value domains (i.e., biophysical, sociocultural, economic) within ecosystem service mapping studies. Therewith, we make an attempt at providing clarity and consistency regarding the operationalisation and conceptualisation of ecosystem services across value domains. This study's aim is addressed in two parts. First, we review common approaches for operationalising ecosystem services into delivery components (i.e., biophysical and socioeconomic aspects of the delivery process). Based on this information, a framework for operationalising ecosystem services is developed, synthesising well-established frameworks and concepts. Second, publications mapping ecosystem services are systematically analysed to assess the consideration of different value domains across sections (i.e., provisioning, regulation and maintenance, cultural services) and delivery components (i.e., supply and demand side). Delivery components are systematically analysed by use of the proposed framework. We find that the supply side of delivery is substantially overrepresented within assessments. Regulation and maintenance services are generally overrepresented, provisioning service representation is sharply declining, and cultural service representation is gradually increasing.

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2.1 Introduction

Throughout history, humans have manipulated Earth's life-support systems to overcome biophysical constraints to human development and correct negative environmental consequences (DeFries et al., 2012). In modern times, human ingenuity has not yet enabled societies to avoid an array of phenomena that threaten their very existence (e.g., climate change, biodiversity loss, food security; DeFries et al., 2012). To mitigate environmental degradation and its impact on human well-being it is necessary to generate a better understanding on the mechanism through which natural capital (i.e., Earth's ecosystems and underpinning geophysical systems; Haines-Young & Potschin, 2018) generates value for humans, thereby enabling the use of such knowledge to support decision-making (Haase et al., 2014; Walsh et al., 2015). In recent years, the ecosystem services framework has been increasingly recognised as a promising instrument to support scientifically-sound environmental decision-making (Vigl et al., 2017).

Ecosystem services are the benefits that ecosystems provide to humans (MA, 2005). Throughout the years, a plurality of conceptual frameworks has been developed to facilitate the integration of ecosystem services within decision-making. This has led to progress regarding the operationalisation of ecosystem services and the delivery process into sections and delivery components. Operationalisation is the process by which concepts are made usable by practitioners (e.g., assessors, decision-makers; Potschin et al., 2014). Sections are groups of ecosystem services that share similar characteristics (https://cices.eu). The Millennium Ecosystem Assessment (2005) operationalised ecosystem services into four sections, which have been adapted within subsequent frameworks (e.g., TEEB, 2010; Landers & Nahlik, 2013; Pascual et al., 2017; Haines-Young & Potschin, 2018). These sections generally include provisioning, regulation and maintenance (also 'regulating'), and cultural services, with some variations. Delivery components are the ecological and socioeconomic building blocks that constitute the delivery process (Villamagna et al., 2013). One of the leading approaches for operationalising the delivery process is the 'cascade' framework (de Groot et al., 2002; Potschin & Haines-Young, 2016). The framework envisions the delivery process as cascading from ecological structures and processes, to ecosystem functions (i.e., ecological structures and processes beneficial to humans), to the benefits and values they generate for society (Costanza et al., 2017). It epitomises the fundamental link that exists between ecosystem functions and the socioeconomic benefits they generate, where the existence of one is not possible without the other. Other leading approaches operationalise the delivery process into delivery components capturing the supply side (i.e., ecological) and demand side (i.e., socioeconomic) of delivery (Villamagna et al., 2013; Burkhard et al., 2012).

Given their spatial complexity, ecosystem services are often quantified spatially, and their distribution illustrated within maps (Burkhard et al., 2012). Ecosystem services do not behave as individual elements in a vacuum but rather as interrelated parts of a network. They coexist across heterogeneous spatiotemporal and thematic gradients, influenced by a diverse and continuously changing array of natural and anthropogenic phenomena (e.g., climate, geology, soil properties, biodiversity, population distribution, cultural values; Erhard et al., 2017; Díaz et al., 2015a; Villamagna et al., 2013). Their fundamental building blocks are interrelated, often in a non-linear fashion (Bennett et al., 2009; Costanza et al., 2017). Mapping the spatial dimension of ecosystem services across different value domains (i.e., values commonly regarded under distinct disciplines, such as biophysical, economic, and sociocultural values) is instrumental for illustrating our complex environment (Traun et al., 2017). It can contribute to the identification of trade-offs and synergies among ecosystem services (Crossman et al., 2013), as well as their drivers of change (Paulin et al., 2020a). It can serve as a means for reifying spatial mismatches between the supply and demand sides of delivery (Crossman et al., 2013; Bagstad et al., 2013a). This is useful for analysing the extent to which the current availability of ecosystem services meets the amount that is required to support the needs and preferences of a growing population.

Despite its potential, a number of limitations hinder the application of the ecosystem services concept within decision-making. We focus on two of these limitations. First, structural ambiguity in the operationalisation of the delivery process complicates the application and interpretation of the concept by practitioners (Stępniewska et al., 2018; Potschin-Young et al., 2018). Second, multifunctional ecosystem service mapping studies often consider a narrow range of value domains. In general, ecosystem services research has often been performed within the field of ecology (Luederitz et al., 2015), carried out separately from economic studies (Rasmussen et al., 2016) and failing to engage local knowledge (Díaz et al., 2018). As a consequence, mapping studies have often failed to fully engage with different aspects of the cascade, disregarding the pervasive relationship that links biophysical and socioeconomic aspects of delivery (Villamagna et al., 2013; Luederitz et al., 2015; Wei et al., 2017). Despite general agreement regarding the need to include the demand side (i.e., anthropogenic aspects) within assessments, most studies have primarily focused on the supply side (i.e., biophysical aspects), considering their interlinkages mainly in conceptual terms (Lautenbach et al., 2019; Burkhard et al., 2012; Wolff et al., 2015). This might explain why regulation and maintenance services have been substantially overrepresented within mapping studies, while cultural services have been substantially underrepresented (Maes et al., 2012; Martinez-Harms & Balvanera et al., 2012; Crossman et al., 2013; Luederitz et al., 2015; Malinga et al., 2015). The systematic underrepresentation of sociocultural values within assessments has become a source of critique in the field, given the fundamental role that culture plays on human-nature interactions (Diaz et al., 2018; de Groot et al., 2018; Maes et al., 2018; Peterson et al., 2018).

The aim of this paper is to evaluate the current consideration of diverse value domains within multifunctional mapping studies. Therewith, we seek to provide clarity and consistency regarding the operationalisation of the delivery process across value domains. In particular, the aim seeks to tackle issues related to the ambiguous operationalisation of the delivery process and to the limited consideration of distinct value domains within multifunctional mapping studies. This study's aim is addressed in two parts. First, for clarity, we review common approaches for operationalising the delivery process (Section 2.3). For consistency, based on the information obtained, we develop a framework for operationalising the delivery process and conceptualising its components. The framework links supply and demand components to elements of the cascade. It also serves as a blueprint for the systematic assessment of ecosystem services in the second part of this study (Section 2.4). In this part, publications mapping ecosystem services are systematically reviewed to evaluate the state of the art regarding the consideration of distinct value domains within multifunctional mapping studies (i.e., studies where at least three ecosystem services have been mapped). To evaluate progress towards the consideration of value domains, we assess the frequency with which sections and delivery components have been considered within mapping studies in recent years.

2.2 Materials and methods

This study was performed in two parts. Part 1 seeks to provide clarity and consistency regarding the operationalisation of the delivery process and conceptualisation of its components. Part 2 reviews the state of the art in ecosystem services mapping across value domains.

2.2.1 Part 1 – Operationalisation and conceptualisation of the delivery process

Various attempts have been made at operationalising the delivery process to facilitate the assessment of ecosystem services (Czúcz et al., 2020). This limits the possibility of assessing ecosystem services consistently across space and time, reducing the comparability of results across space and reducing confidence in assessment output (Villamagna et al., 2013; Luederitz et al., 2015; Wei et al., 2017). To address this challenge, we first identify, analyse, and discuss common approaches for operationalising the delivery process and conceptualising its delivery components (Section 2.3). Delivery components considered in this study comprise components that constitute the cascade framework, as well as supply and demand components. Based on identified formulations and interpretations of delivery components, we present a framework linking cascade components with supply and demand components. This framework is then used as a blueprint for systematically analysing the ecosystem services mapping literature.

2.2.2 Part 2 – Systematic review

The mapping literature was identified by performing a search in the Scopus database. An initial search in Scopus applying keyword combinations presented in the Supplementary Material (Appendix 1 – 1) resulted in 1,173 peer-reviewed articles, which were further analysed to identify those of relevance to this study. Ecosystem services coexist within multifunctional landscapes, reacting in synergistic or antagonistic fashion to anthropogenic and ecological drivers of change (Dade et al., 2019). To capture this multifunctionality, case studies mapping at least three ecosystem services were considered. We identified 111 publications comprising 123 case studies mapping at least three ecosystem services between the years 2017-2019. The complete list of publications analysed can be found in the Supplementary Material (Appendix 1 – 2).

Case studies were assessed to determine the number of ecosystem service indicators mapped, as well as the respective section and the delivery component under which each indicator falls. Indicators mapped in the literature were operationalised into delivery components based on the framework developed in Part 1 of this study. Indicators were operationalised into sections based on the Common International Classification of Ecosystem Services (CICES; Haines-Young & Potschin, 2011; https://cices.eu/). CICES categorises ecosystem services into three overarching sections: provisioning, regulation and maintenance, and cultural services. Provisioning services are the material benefits ecosystems provide for humans (MA, 2005), such as food for consumption purposes and accrued nutritional benefits. Regulation and maintenance services are ecological processes that contribute to human well-being (MA, 2005), such as particulate matter retention by vegetation and its contribution to human health (Derkzen et al., 2015). Cultural services are non-material benefits that ecosystems generate for humans (MA, 2005), such as the spiritual, recreational, educational, or intrinsic value people assign to natural elements. Part 1 – Operationalisation of the delivery process.

2.3 Part 1 – Operationalisation of the delivery process

2.3.1 General overview

2.3.1.1 Cascade framework

The ecosystem service delivery process comprises various stages that are dichotomous, highly complex, and interdependent in character. The ecosystem services cascade (Figure 2.1; de Groot et al., 2002; Potschin & Haines-Young, 2016) operationalises these stages by drawing a clear distinction between ecological structures and processes, ecosystem functions, and the benefits and values these generate for humans. In this context, ecosystem functions comprise a subset of ecological structures and processes unique in their capacity to directly or indirectly benefit humans (de Groot et al., 2002). In the cascade, ecosystem services comprise an independent delivery component that links ecosystem functions and the human aspect of ecosystem service delivery (i.e., benefits and values). The supply and demand of ecosystem services is influenced by ‘pressures’, or natural and anthropogenic stressors that affect the quantity and quality of ecosystem services, as well as people’s dependency on them (Villamagna et al., 2013).

2.3.1.2 Ecosystem services

Ecosystem services have been conceptualised in a variety of ways, becoming subject to a wide range of interpretations (Costanza et al., 2017). In the work of de Groot et al. (2002), ecosystem services are defined as beneficial ecosystem functions and their benefits to society. In the MA (2005), they constitute "the benefits that people obtain from ecosystems." In The Economics of Ecosystems and Biodiversity (TEEB, 2010), they encompass "the direct and indirect contributions of ecosystems to human well-being." Within CICES, they comprise the contributions (final products or outputs) from ecosystems to human well-being, arising from the interaction of biotic and abiotic processes (Haines-Young & Potschin, 2011). Within Burkhard et al. (2012), they are defined as "the contributions of ecosystem structure and function – in combination with other inputs – to human well-being." In addition, ecosystem services have been operationalised into intermediate and final services (Boyd & Banzhaf, 2007), where delivery components are considered intermediate or final depending on their degree of connection to human well-being (Crossman et al., 2013).

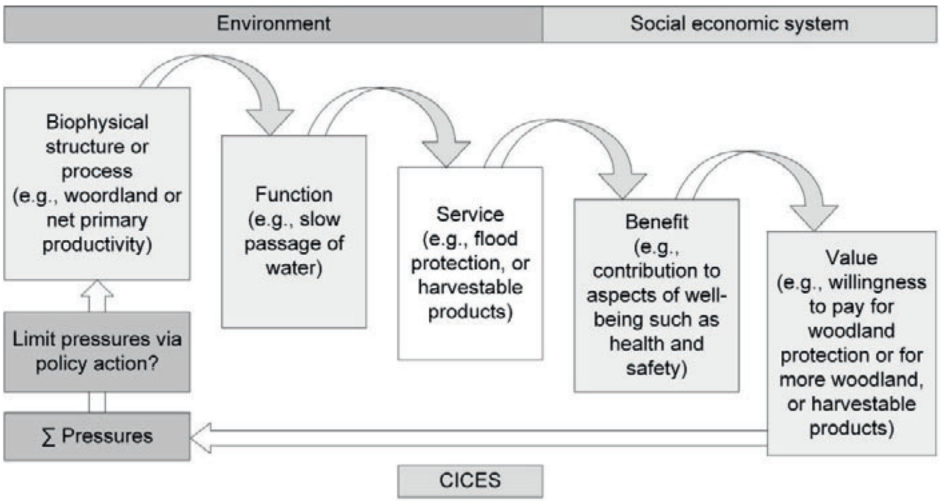

Figure 2.1: Ecosystem service cascade (credit/adapted from: Potschin & Haines-Young, 2016)

Such definitions are instrumental due to their simplicity, yet they offer a range of possibilities for interpretation. This aligns well with the notion that operationalisation and conceptualisation approaches should provide enough flexibility to enable different interpretations and adaptations in diverse contexts (Potschin-Young et al., 2018). However, too much flexibility may generate confusion regarding whether ecosystem services encompass ecosystem functions, the benefits and values enjoyed by humans, a combination of these components, or perhaps a separate component of the delivery process (Costanza et al., 2017; Potschin & Haines-Young, 2016). This may explain, for instance, why a number of ecosystem service assessment studies disregard the notion of ecosystem functions (Potschin-Young et al., 2018).

2.3.1.3 Benefits and values

In the ecosystem service cascade, ecosystem services constitute an independent delivery component that links ecosystem functions with the benefits and values they provide for humans. Potschin & Haines-Young (2017) argue that the difference between an ecosystem service and a benefit is that benefits exist when people assign value to them. Meanwhile, Kenter et al. (2015) define values as the importance people ascribe to nature, influenced by their attitudes, preferences, behaviours, and overarching life goals and principles. The existence of overlaps in the conceptualisation of benefits and values emphasises the need to conceptualise values and benefits in a consistent manner (Potschin-Young et al., 2018). It also raises questions regarding the way in which benefits differ from values, and whether values merely constitute a metric for reification of natural capital's benefits to humans. This may explain, for instance, why a number of ecosystem service assessment studies have chosen to merge the concepts of benefits and values (Potschin-Young et al., 2018). Moreover, the conceptualisation of benefits and values as those only perceived by humans, conflicts with the generally-accepted premise that ecosystem services contribute to human well-being as a whole (Costanza et al., 2017; Potschin-Young et al., 2018). This is the case since well-being is a state that is determined by a range of factors that characterise people's everyday lives from both a local and global perspective (e.g., survival, health, principles, preferences; Costanza et al., 2017; Potschin-Young et al., 2018), irrespective of whether these are perceived by their very receptors (Costanza et al., 2017; Fioramonti, 2014). In this study, ecosystem service benefits and values constitute nature's contributions to human well-being as a whole, irrespective of whether these are perceived by their very receptors.

2.3.1.4 Supply and demand

Ecosystem services are not always produced and consumed *in situ* (Fisher et al., 2009; Syrbe & Walz, 2012; Crossman et al., 2013). Failing to address the spatiotemporal mismatches between ecosystem service providing areas and benefitting areas could lead to inaccuracies in ecosystem service valuation and trade-off analyses (Crossman et al., 2013; Bagstad et al., 2013a). This realisation has led to a plurality of authors calling for a distinction between the supply side (ecological) and demand side (anthropogenic) of ecosystem service delivery (Burkhard et al., 2012; Tallis et al., 2012; Villamagna et al., 2013; Crossman et al., 2013; Bagstad et al., 2014; Baró et al., 2017; Vigl et al., 2017; Wei et al., 2017). Despite some similarities, however, the operationalisation of the delivery process into supply and demand components varies largely across the literature (Bagstad et al., 2014; Potschin-Young et al., 2018). We expand on some of these approaches hereunder.

Supply

Ecosystem service supply has been conceptualised in a plurality of ways. A commonly adopted definition was coined by Burkhard et al. (2012), who defined supply as "the capacity of a particular area to provide a specific bundle of ecosystem goods and services within a given time period." In this context, Burkhard et al. (2012) made a distinction between 'capacity' and 'potential supply,' where potential supply constitutes the hypothetical maximum yield of ecological resources and processes, whereas capacity constitutes actually used natural resources and processes. Tallis et al. (2012) defined supply as "the full potential of ecological functions or biophysical elements in an ecosystem to provide a potential ecosystem service, irrespective of whether humans actually use or value that function or element currently." This conceptualisation aligns with the term 'potential supply' as suggested by Burkhard et al. (2012), which also aligns with other terms implemented in the literature, such as 'capacity' (Villamagna et al., 2013; Baró et al., 2017; Vigl et al., 2017) and 'theoretical service provision' (Bagstad et al., 2014). The term 'capacity', as conceptualised by Burkhard et al. (2012), aligns with other commonly adopted terms, such as 'flow' (Villamagna et al., 2013; Baró et al., 2017) and 'actual use' (Bagstad et al., 2013a). The use of the term 'flow' to refer to the actual provision of an ecosystem service conflicts with its use within a range of studies where it refers to the path that links service providing areas and benefiting areas (Johnson et al., 2010; Bagstad et al., 2013a; Bagstad et al., 2014). Generally, a key distinction between an ecosystem's potential to provide services and actual service provision is that actual provision requires the existence of beneficiaries, regardless of whether they are aware of it or recognised as such (Bagstad et al., 2014). In this study, ecosystem service supply refers to the full potential of ecological resources and processes to generate benefits that contribute to human well-being.

Demand

Wolff et al. (2015) reviewed conceptualisations of demand across the ecosystem services literature. This led to its operationalisation into two main components. The first type of demand captures the 'direct use' (e.g., visits to parks, water use for irrigation) or 'consumption' (e.g., water or energy consumption by humans) of natural resources and processes (Wolff et al., 2015). This conceptualisation aligns with the terms ‘flow’ and ‘demand’, as suggested by Villamagna et al. (2013) and Burkhard et al. (2012), respectively. That is, “the sum of all ecosystem goods and services currently consumed or used in a particular area over a given time period” (Burkhard et al., 2012). The second type of demand suggested captures human ‘desires’, or the amount of an ecosystem service desired by society (Wolff et al., 2015). This definition aligns with the concept of ‘demand’ by Villamagna et al. (2013), namely “the amount of a service desired or required by society”. In this context, desires for ecosystem services may exceed their actual use, whereas ecosystem service use constitutes ecosystem service desires and preferences which are met by their available supply (Wolff et al., 2015). Wolff et al. (2015) further operationalised ‘desires’ for ecosystem services into two components. The first form of desire constitutes human preferences for particular ecosystem services. Information regarding people’s preferences for ecosystem services is often obtained by use of stated preference and revealed preference valuation techniques (Wolff et al., 2015). The second form of desire constitutes societal needs for risk reduction (Wolff et al., 2015). For instance, the incidence of flooding or health hazards in urban areas can be mitigated by expanding and enhancing the quality of green infrastructure (Paulin et al., 2020a).

Considering human desires for an ecosystem service is of particular importance since, in considering only the actual use of an ecosystem service as a metric for demand, it is assumed that all human demand for an ecosystem service is met (Wolff et al., 2015). In reality, ecosystem service consumption and use does not take place in a level playing field. Rather, people’s consumption possibilities are often limited by circumstantial factors, such as individual resource endowments (e.g., time, money, freedom), or the availability of and accessibility to potentially beneficial ecological resources and processes. Consideration of ecosystem service consumption and use as a single metric for demand assumes a higher demand in areas where consumption is currently higher. In doing so, it disproportionately prioritises areas where higher consumption possibilities predominate, rather than areas where consumption needs and preferences are unmet. For instance, the use of drinking water is much higher in the western world than in developing countries (WHO, 2017). However, the amount of drinking water required for individual survival in the two areas is approximately the same. In this study, ecosystem service demand constitutes human consumption and use of, as well as desires for ecosystem services. More specifically, we define demand as human desires for ecological resources and processes that support their well-being, as well as the actual benefits and values that they enjoy due to the current availability of ecological resources and processes.

2.3.2 Framework for operationalising the ecosystem service delivery process

The analysed literature indicates that substantial ambiguity exists regarding the operationalisation of the ecosystem service delivery process and conceptualisation of its components. In an attempt to provide clarity and reduce identified inconsistencies, we introduce an operationalisation framework linking cascade components with supply and demand components (Figure 2.2). The proposed framework is implemented in Section 2.4 in order to systematically review the extent to which delivery components are mapped across the literature. In this context, ecosystem services constitute the direct and indirect contributions of ecosystems to human well-being (TEEB, 2010). As such, they encompass both ecosystem functions (i.e., ecological structures and processes of functional use for humans; de Groot et al., 2002), as well as their accrued benefits and associated socioeconomic values.

Supply has been operationalised into two main components. ‘Potential supply’ refers to the range of ecological resources and processes that may potentially benefit humans, irrespective of whether they are actually used or valued by humans (Tallis et al., 2012). ‘Actual supply’ refers to ecological resources and processes that effectively benefit humans, namely ecosystem functions. In this context, an ecosystem function may comprise any ecological resource or process that is beneficial for humans.

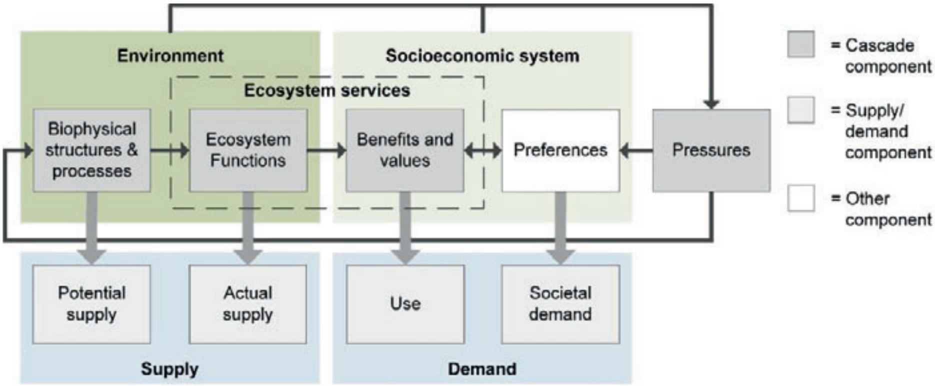

Figure 2.2: Framework operationalising the ecosystem service delivery process into nine delivery components (adapted from Potschin & Haines-Young, 2016)

Demand has been operationalised into two main components. 'Use' comprises socioeconomic benefits, as well as their respective values, that humans derive from their direct or indirect use of ecological resources and processes (Paulin et al., 2020b). In this context, consumption constitutes a form of use. Benefits are the contributions by ecosystems to human well-being, their value expressed in sociocultural or economic units. 'Societal demand' constitutes human desires for ecosystem services (Wolff et al., 2015), irrespective of whether these are met by their actual supply and use. The societal demand for ecosystem services is shaped by people's needs and preferences (Wolff et al., 2015). For instance, people's preference for a particular food source may be influenced by media campaigns and cultural values that endorse their consumption. Meanwhile, people's need for a particular food source may be influenced by their basic survival necessities for nutrition and health. Societal demand is additionally influenced by ecological and anthropogenically induced pressures that require mitigation (Wolff et al., 2015). For instance, the excessive use of pesticides in agriculture is often detrimental for pollinator populations and surface water quality (Cresswell et al., 2018), affecting the supply of pollinating services while increasing people's dependency on water quality regulation services.

2.3.2.1 Application

For demonstration purposes, the ecosystem service Air Quality Regulation has been operationalised into delivery components, based on the proposed framework. Atmospheric PM_{10} (i.e., particulate matter with a diameter of up to 10 μm) is a common problem within cities, caused by sources such as traffic and industry (Paulin et al., 2020b). It is commonly associated with respiratory and cardiovascular diseases, as well as mortality (Derkzen et al., 2015; Santibañez et al., 2013). In this example, Air Quality Regulation refers to the contribution by vegetation and water to reductions in atmospheric PM_{10} concentrations, as well as accrued health benefits enjoyed by urban dwellers. Delivery components were mapped for the Municipality of Amsterdam, the Netherlands (Figure 2.3). All but one indicator (i.e., PM_{10} concentration) were spatially quantified by use of the Natural Capital Model (NC-Model; Paulin et al., 2020b). Input data and models implemented for mapping each delivery component are described in the Supplementary Material (Appendices 1 – 3 and 1 – 4, respectively).

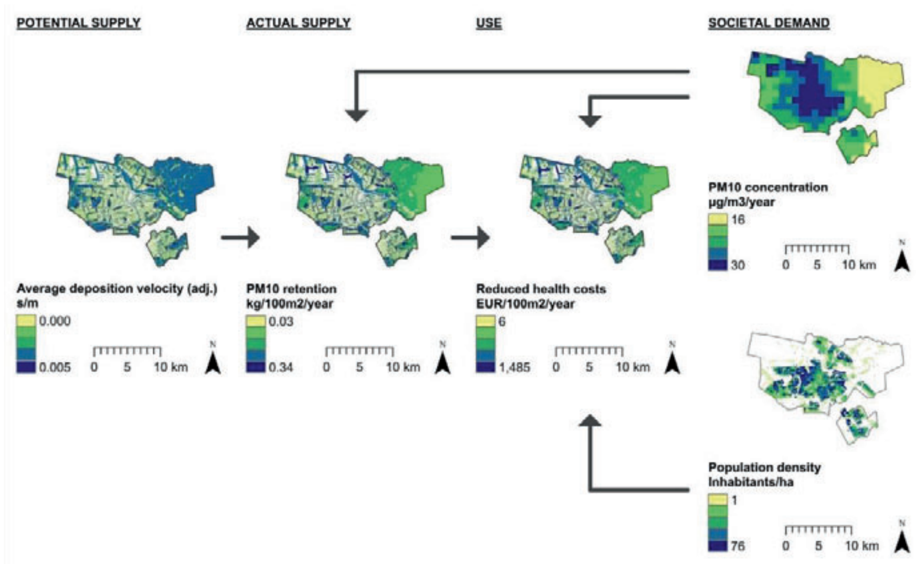

Figure 2.3: Operationalisation of delivery components for the ecosystem service Air Quality Regulation. For all maps, legends show quantile values. All quantile thresholds values are presented in the Supplementary Material (Appendix 1 – 5).

Potential supply refers to the potential for vegetation and water to retain PM_{10}. The potential for PM_{10} retention was modelled as a function of the deposition velocity and resuspension of suspended particles associated with water and different vegetation types (i.e., low vegetation, bushes and shrubs, trees). The deposition velocity is the speed with which particulate matter deposits to the natural surface (Chen et al., 2012). Resuspension occurs when deposited particles are re-emitted into the air due to various factors (e.g., physical characteristics of the contaminated surface, physicochemical nature of the contaminant, meteorological conditions; Gradoń, 2009). The societal demand for PM_{10} retention is a function of actual atmospheric concentrations of PM_{10} and population density, which reflects the human component of societal demand. Atmospheric concentrations of PM_{10} act as a pressure that requires mitigation, as it negatively affects human health. The actual Air Quality Regulation supply consists of actual PM_{10} retention by vegetation and water. It is modelled as a function of the capacity for PM_{10} retention (i.e., potential supply) and actual PM_{10} concentrations (i.e., societal demand). Use refers to the socioeconomic benefits that result from actual PM_{10} retention. In this case, use captures the reduction in health costs within a particular area that results from reduced PM_{10}-related mortalities (CE-Delft, 2017; Paulin et al., 2020b). In this example, ecosystem service use has a linear relationship with actual supply and is weighted based on the population density. This explains the similarity in distribution for the actual supply and use maps.

2.4 Part 2 – State of the art in ecosystem services mapping

2.4.1 General overview

A systematic literature search resulted in 123 case studies, mapping a total of 839 delivery component indicators. Mapped indicators represent 742 ecosystem services and their respective sections, or an average of 6 ecosystem services per case study. Most case studies were conducted at the subnational scale (84%), while only a few were conducted at the national (11%) and regional (supranational) scales (4%; Figure 2.4). No case study was conducted at the continental scale and one case study was conducted at the global scale (Wolff et al., 2017). Only 9% of case studies assessed urban ecosystem services. Case studies were systematically analysed to determine the frequency with which distinct value domains (i.e., biophysical, sociocultural, economic) are considered. In particular, the literature was analysed to determine the frequency with which ecosystem service sections and delivery components are mapped within the literature. Delivery components were conceptualised based on the operationalisation framework introduced in Section 2.3.2.

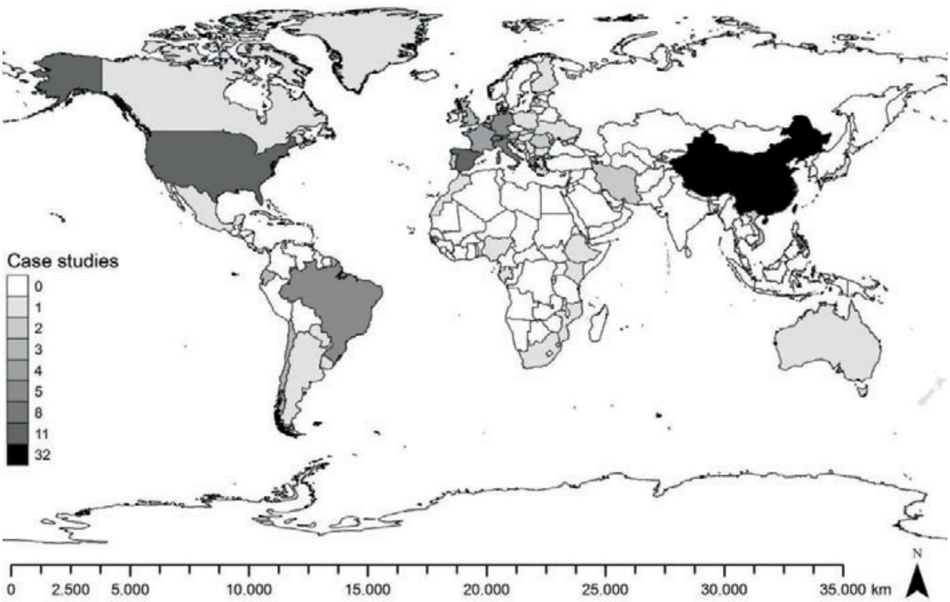

Figure 2.4: Number of case studies per country mapping at least three ecosystem services. Some case studies covered more than one country. Global case study not considered in this map, as it covers all countries.

Roughly 40% of studies did not specify the delivery components that were mapped. As a result, it became difficult to determine whether an indicator was meant to capture potential or actual supply. This occurred since the actual supply of a particular service is often equivalent to its potential supply in situations where the societal demand exceeds the potential supply (Baró et al., 2017). For instance, it can be assumed that the potential supply for carbon sequestration is equivalent to its actual supply, as current carbon emissions exceed their desired emission rates (Baró et al., 2017; Schröter et al., 2014). Given the difficulty in the differentiation of potential and actual supply in situations where delivery components mapped were not made explicit, the systematic review considered one overarching ‘supply’ category, encompassing both forms of supply. Table 2.1 provides various examples on how ecosystem services indicators that were mapped in the reviewed literature were operationalised.

Table 2.1: Example of categorisation of ecosystem service indicators mapped in the literature into sections and delivery components

Ecosystem service	Delivery component	Indicator	Unit	Source
Provisioning services				
Honey Production	Supply	Honey production	number of hives/km^2	Balzan et al. (2018)
	Societal demand	Importance to beekeepers of different land covers and plant species for honey production	dimensionless index (expert elicitation)	Balzan et al. (2018)
Food Provision	Supply	Crop production	kg/ha/year	Baró et al. (2017)
	Use	Crop output value per cropland area	yuan/km^2	Hou et al. (2018)
	Societal demand	Population density	inhabitants/ha	Baró et al. (2017)
Regulation and maintenance services				
Air Quality Regulation	Supply	NO_2 dry deposition velocity	mm/s	Baró et al. (2017)
	Supply	NO_2 removal flux	T/ha/year	Balzan et al. (2018)
	Societal demand	NO_2 concentration levels	µg/m^3/year	Baró et al. (2017)
Climate Regulation	Supply	Carbon sequestration	kg C/ha/year	Baró et al. (2017)
	Societal demand	Carbon emissions	kg C/ha/year	Baró et al. (2017)
Erosion Control	Supply	Erosion control capacity	dimensionless index (expert elicitation)	Baró et al. (2017)
		Capacity of vegetation cover to avoid soil erosion	Tm/ha/year	Quintas-Soriano et al. (2019)
	Societal demand	Soil loss potential	dimensionless index (based on population density)	Baró et al. (2017)
		Soil loss (Universal Soil Loss Equation, USLE)	ton/km^2	Hou et al. (2018)
Pollination	Supply	Pollinator visitation probability	visitation probability (%)	Balzan et al. (2018)
	Supply	Pollination contribution by ecosystems	dimensionless index (expert elicitation)	Vigl et al. (2017)
	Societal demand	Additional area needed in the absence of pollinators	m^2 per capita	Wolff et al. (2017)

Cultural services				
Aesthetic Value	Use	Landscape beauty metrics index	dimensionless index (expert elicitation)	Vigl et al. (2017)
		Physical use of landscapes	dimensionless index (expert elicitation)	Balzan et al. (2018)
Cultural Tourism	Use	Number of annual person-days of photographs uploaded to Flickr (https://www.flickr.com/)	person-days of photographs/km^2	Quintas-Soriano et al. (2019)
Mushroom Picking	Supply	Potential area for mushroom picking	ha	Vigl et al. (2017)
Recreation	Supply	Recreation surface per capita	ha/capita	Vigl et al. (2017)
		Recreational potential based on natural landscape characteristics and distance to population centre	dimensionless index (expert elicitation)	Hou et al. (2018)
Rest and Nature Experience	Supply	Capacity for rest and nature experience	dimensionless index (Hemeroby index)	Arnold et al. (2018)

2.4.2 Sections

Figure 2.5 and Table 2.2 provide an overview of the frequency with which sections have been mapped across the reviewed literature. Regulation and maintenance services were generally overrepresented in the literature (mapped in 88% of case studies), compared to provisioning (68%) and cultural services (56%; Figure 2.5a). They were also mapped more frequently within case studies (3 times on average per case study), compared to provisioning (1 time on average) and cultural services (2 times on average). Despite a higher representation of regulation and maintenance services, a pattern of convergence in the consideration of value domains considered in this study seems to be taking place. For instance, more than 40% of case studies mapped all sections simultaneously in 2019 (Table 2.2c) and the gap between the representation of sections seems to be diminishing (Figure 2.5b). In particular, the mapping of cultural services saw a noticeable increase in 2019 (64% of case studies), compared to during previous years (2017=53%; 2018=46%). Conversely, provisioning services mapping has been visibly diminishing, with only 59% of case studies mapping provisioning services in 2019, compared to 83% in 2017 and 70% in 2018.

2.4.3 Delivery components

Figure 2.6 and Table 2.3 provide an overview of the frequency with which delivery components have been mapped across the reviewed literature. Supply is highly overrepresented within the literature (mapped in 83% of case studies), compared to use (24%) and societal demand (22%; Figure 2.6a). In fact, more than half of the reviewed literature mapped supply exclusively. Supply is also mapped more frequently per case study (5 times on average per case study), compared to use (1 time on average) and societal demand (1 time on average). It could be argued that the overrepresentation of supply is a consequence of including potential supply and actual supply as a single category. However, the rate at which supply is mapped within the literature (83% of case studies) far exceeds the rate at which demand as a whole is mapped (46%). The same conclusion can be reached if we consider the rate at which supply and demand indicators have been mapped, with 68% of mapped indicators constituting supply and only 32% constituting demand. Despite the stark contrast in the representation of supply and demand, a sharp increase was seen in the mapping of ecosystem service use in 2019 (32% of case studies), compared to previous years (2017=13%; 2018=19%). This does not apply to societal demand, which has been mapped at fluctuating rates (2017=23% of case studies; 2018=14%; 2019=27%; Figure 2.6b). Supply is most often mapped to represent regulation services (66% of supply indicators mapped), compared to provisioning (24%) and cultural services (10%). Use is most often mapped to represent cultural services (50% of use indicators mapped), compared to provisioning (24%) and regulation and maintenance services (26%). Societal demand is most often mapped to represent cultural services (56% of societal demand indicators mapped), compared provisioning (19%) and regulation and maintenance services (25%).

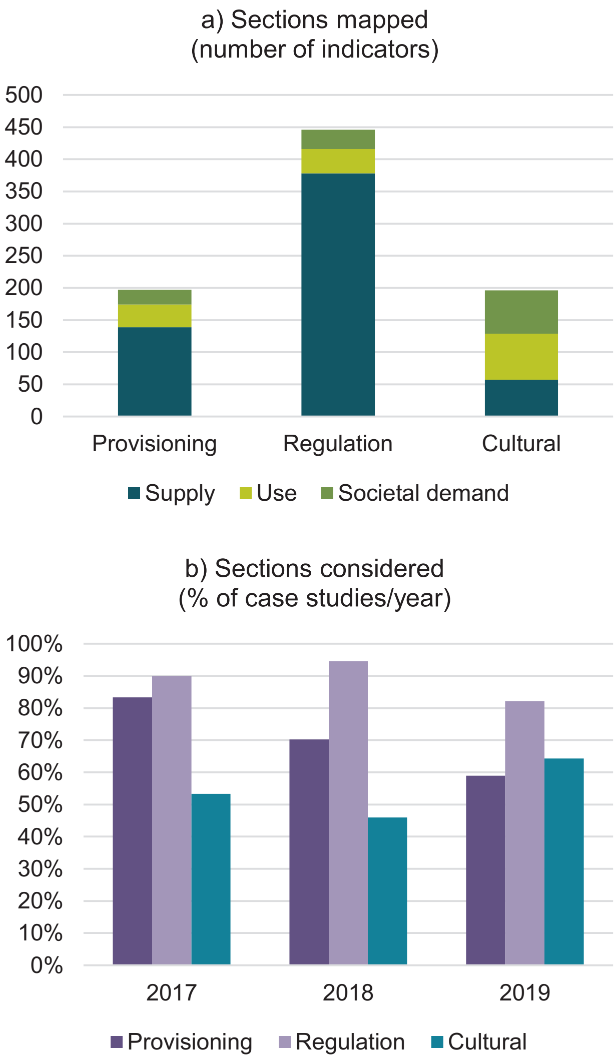

Figure 2.5: Ecosystem service sections mapped across the literature (2017-2019)

Table 2.2: Ecosystem service sections mapped within multifunctional landscapes (at least three ecosystem services mapped) during the period 2017-2019. In section 'b', the number of case studies mapping each section does not add up to the total number of case studies assessed (=123), as case studies often map more than one delivery component.

Year(s)	**2017-2019**		**2017**	**2018**	**2019**
a) Number (%) of sections mapped	**n**	**%**	**%**	**%**	**%**
Provisioning	173	23%	26%	23%	22%
Regulation & maintenance	389	52%	48%	62%	48%
Cultural	180	24%	26%	14%	30%
Total number of ecosystem services	**742**	**100%**	**100%**	**100%**	**100%**
b) Number (%) of case studies mapping sections	**n**	**%**	**%**	**%**	**%**
Provisioning	84	68%	83%	70%	59%
Regulation & maintenance	108	88%	90%	95%	82%
Cultural	69	56%	53%	46%	64%
Total number of case studies	**123**	**100%**	**100%**	**100%**	**100%**
c) Number (%) of case studies mapping combinations of sections	**n**	**%**	**%**	**%**	**%**
Provisioning (only)	1	1%	0%	0%	2%
Regulation & maintenance (only)	20	16%	7%	16%	21%
Cultural (only)	12	10%	7%	5%	14%
Provisioning + regulation & maintenance (only)	33	27%	40%	38%	13%
Provisioning + cultural (only)	2	2%	3%	0%	2%
Regulation & maintenance + cultural (only)	7	6%	3%	8%	5%
Provisioning + regulation & maintenance + cultural	48	39%	40%	32%	43%
Total number of case studies	**123**	**100%**	**100%**	**100%**	**100%**

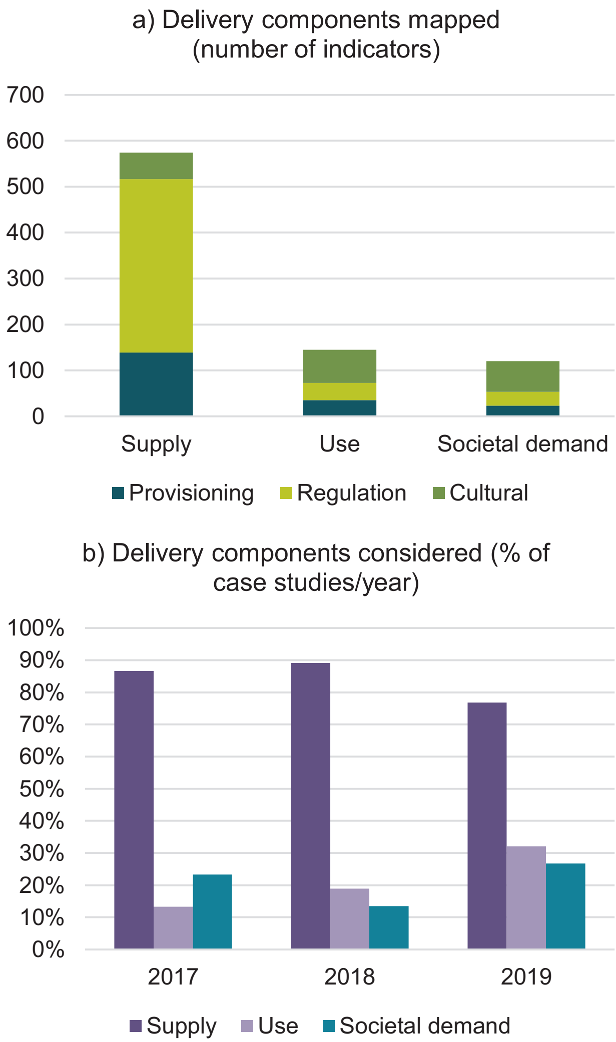

Figure 2.6: Ecosystem service delivery components mapped across the literature (2017-2019)

Table 2.3: Ecosystem service delivery components mapped within multifunctional landscapes (2017-2019). In section 'a', the number of delivery components mapped exceeds the number of sections mapped, as more than one delivery component can be mapped per section. In section 'b', the number of case studies mapping each delivery component does not add up to the total number of case studies (=123), as case studies often map more than one delivery component.

Year(s)	**2017-2019**		**2017**	**2018**	**2019**
a) Number (%) of delivery components mapped	**n**	**%**	**%**	**%**	**%**
Supply	575	68%	72%	75%	62%
Use	145	17%	17%	15%	19%
Societal demand	120	14%	11%	9%	19%
Total number of indicators	**840**	**100%**	**100%**	**100%**	**100%**
b) Number (%) of case studies mapping delivery components*	**n**	**%**	**%**	**%**	**%**
Supply	102	83%	87%	89%	77%
Use	29	24%	13%	19%	32%
Societal demand	27	22%	23%	14%	27%
Total number of case studies	**123**	**100%**	**100%**	**100%**	**100%**
c) Number (%) of case studies mapping combinations of delivery components	**n**	**%**	**%**	**%**	**%**
Supply (only)	73	59%	70%	70%	46%
Use (only)	9	7%	7%	8%	7%
Societal demand (only)	10	8%	3%	3%	14%
Supply + use (only)	14	11%	0%	8%	20%
Supply + societal demand (only)	11	9%	13%	8%	7%
Use + societal demand (only)	2	2%	3%	0%	2%
Supply + use + societal demand (only)	4	3%	3%	3%	4%
Total number of case studies	**123**	**100%**	**100%**	**100%**	**100%**

2.4.4 Spatial quantification methods

All ecosystem services have been spatially quantified by use of social-ecological assessment models (Martinez-Harms & Balvanera, 2012). Based on our mechanistic understanding of coupled human-natural systems, social-ecological assessment models spatially quantify ecosystem service proxy indicators by modelling the relationship among measurable biophysical (e.g., land-cover, remote-sensed data, spatially extrapolated field observations) and socioeconomic variables (e.g., population, survey data, statistical data). We identified two main types of social-ecological assessment models that have been implemented to spatially quantify delivery components. Data-based approaches integrate spatial and non-spatial data (e.g., from statistics, publications, field samples) to capture the relationship between biophysical and socioeconomic variables. Expert elicitation involves the mobilisation of experts to assign values to spatial and non-spatial variable categories (e.g., through ranking, rating, assigning, weights, photo elicitation) based on their knowledge (Martinez-Harms & Balvanera, 2012; Paulin et al., 2020c). In this context, an expert is any individual who may provide key information that is necessary for understanding and modelling the system under assessment (e.g., scientist, technician, stakeholder; Jacobs & Burkhard, 2017). This method is often implemented in situations where resource constraints (i.e., knowledge, data, time) limit the possibility of developing exhaustive models capturing the mechanistic relationship between biophysical and socioeconomic variables (Jacobs & Burkhard, 2017; Paulin et al., 2020c).

Supply

The supply of ecosystem services has been spatially quantified by use of data-based and expert elicitation approaches, simulating the distribution of natural resources and processes that potentially or actually benefit humans. Data-based approaches can vary in difficulty, from complex mechanistic (e.g., Sharps et al., 2017; Braun et al., 2018) and statistical models (e.g., regressions; Hermoso et al., 2018; Kong et al., 2018) integrating multiple variables, to more simplistic value transfer models linking secondary data (e.g., from statistics or publications) to spatial typologies (usually land cover; e.g., Lima et al., 2017; Zhang et al., 2019). The spatial quantification of supply based on expert elicitation was mainly achieved by mobilising scientists and technicians, who then assign values to spatial and non-spatial variable categories (e.g., Zidar et al., 2017; Baró et al., 2017; Lima et al., 2017). In a few cases, stakeholders that were relevant to a particular study site were mobilised to provide information on the location of ecosystem service supply (e.g., Palomo-Campesino et al., 2018; Jaligot et al., 2019). The supply of ecosystem services can be spatially quantified by implementing data-based and expert elicitation approaches independently or through their combined incorporation within social-ecological assessment models.

Use

Use, which constitutes the socioeconomic benefits that ecosystems endow for humans, has been expressed by means of sociocultural indicators (52% of use indicators mapped) and monetary indicators (48%). Sociocultural use indicators have been spatially quantified mainly by implementation of sociocultural valuation approaches (69% of sociocultural use indicators). Less frequently, sociocultural use indicators have been estimated by use of sociocultural data-based approaches (31% of sociocultural indicators). Sociocultural valuation approaches are methods that estimate people's use of and preferences for ecosystem services through expert elicitation (e.g., preference assessment method; participatory mapping method; deliberative methods; Santos-Martín et al., 2017). In this case, sociocultural valuation approaches were implemented to obtain information on people's use of ecosystem services. For instance, in Canedoli et al. (2017), stakeholders were requested to assign scores to cultural services and identify areas where they made use of each service. In sociocultural data-based approaches, readily-available data (e.g., photos obtained through social media platforms; park visitation rate statistics) were analysed to estimate people's use of ecosystem services (e.g., Clemente et al., 2019; Retka et al., 2019). Monetary use indicators have been spatially quantified by use of well-established economic valuation approaches. Economic valuation approaches commonly include data-based (i.e., direct market valuation approach; revealed preference approach; TEEB, 2010) and expert elicitation approaches (i.e., stated-preference approach; TEEB, 2010). In data-based approaches, readily-available data capturing individual behaviour and market transactions are directly related to ecosystem services to estimate their monetary value (e.g., market price method; travel cost method; benefit transfer method; TEEB, 2010; Plummer, 2009). In expert elicitation, approaches people's willingness to pay for ecosystem services is estimated by use of surveys that simulate markets (e.g., contingent valuation method; choice modelling method; TEEB, 2010). The latter approach was not implemented a single time across the analysed literature.

Societal demand

Societal demand has been expressed by use of sociocultural indicators (80% of societal demand indicators mapped) and biophysical indicators (20%). Sociocultural indicators for societal demand have been spatially quantified by implementation of sociocultural valuation approaches (83% of sociocultural indicators for societal demand) and data-based approaches (17%). Sociocultural valuation approaches have been described earlier. In this case, sociocultural valuation approaches were implemented to obtain information on people's preferences for ecosystem services, regardless of their realised use (e.g., Johnson et al., 2019; Palomo-Campesino et al., 2018). In data-based approaches, readily-available data (e.g., from statistics and publications) capturing people's needs and preferences were directly related to ecosystem services to estimate their value. For instance, Wolff et al. (2017) spatially quantified the global reliance on wild medicinal plants by focusing on the distribution of rural-poor segments of developing societies, where the availability, accessibility, and affordability of conventional medicine is usually low. Similarly, Peng et al. (2017) considered population density as a proxy indicator for areas that could benefit from landscape aesthetics. The societal demand for ecosystem services was also expressed as biophysical indicators by use of data-based approaches. For instance, Wolff et al. (2017) spatially quantified the global reliance on pollinators by focusing on the per-capita area demand for pollinator-dependent crops. On a different perspective, Chen et al. (2019) spatially quantified the demand for PM_{10} retention and carbon sequestration by focusing on environmental quality standards (i.e., PM_{10} concentration and carbon emission targets).

2.5 Discussion

2.5.1 Operationalisation of the delivery process

It has been argued that that which can be measured can be managed (Haines-Young & Potschin-Young, 2018). It has also been argued that the ecosystem services concept can provide a means for measuring the many benefits that ecosystems generate for humans. Measuring ecosystem's contributions to people is useful for creating awareness of their importance for human well-being and for supporting their integration within decision-making (Vigl et al., 2017). Despite substantial progress, the consistent operationalisation of the delivery process into measurable indicators seems to remain a challenge. It is common that identical terms with various conceptualisations are used interchangeably, while a diverse terminology is adopted to refer to identical concepts. This hinders the comparability of assessment results and confuses their interpretation by end users. Hence, the field could benefit from a flexible yet consistent operationalisation framework (Villamagna et al., 2013). In an attempt to provide clarity and consistency, we developed an overarching framework synthesising common approaches for operationalising the delivery process. The framework is flexible in that delivery components can be modelled in a variety of ways depending on case-specific resource endowments (i.e., knowledge, data, time). However, each delivery component has been conceptualised in such detail as to avoid ambiguity in the interpretation and subsequent use of each concept in practice. To ensure consistency, we additionally provided case examples of the implementation of the framework to operationalise the delivery process (Figure 2.3; Table 2.1).

In particular, two influential approaches for operationalising ecosystem services were integrated within the presented framework. The first approach, namely the ecosystem services cascade, operationalises the delivery process as a means to capture the ecological and socioeconomic components that constitute ecosystem service delivery. Reification of the ecological aspect of delivery is key for generating a better understanding on the diverse ecological processes and resources that humans rely on, as well as the mechanism that underpins their production. It can serve as a means to generate awareness in the way ecological resources and processes are interrelated, often in a non-linear fashion (Bennett et al., 2009; Costanza et al., 2017). Reification of the socioeconomic aspect of ecosystem service delivery is central for providing decision-makers with information that relates to socioeconomic targets, and for providing the public with information that relates to their needs and preferences (Paulin et al., 2020a). Offering a view rooted in the economic sciences, the second approach operationalises the delivery process into supply and demand indicators. This approach is instrumental for understanding the relationship between the potential production, actual production, use, and societally desired amount of ecosystem services. Reification of the spatial matches and mismatches between supply and demand components is useful for decision-makers who wish to identify areas where the societal demand for ecosystem services is not met by their use, and areas where the potential supply can be further explored to optimise its use.

2.5.2 Mapping of ecosystem services across value domains

2.5.2.1 Delivery components

A glance at the representation of delivery components across the literature indicates that the supply side has been substantially overrepresented, compared to the demand side. This finding aligns with recent findings by Lautenbach et al. (2019), who analysed case studies assessing ecosystem services prior to March 2016. They found that the supply side was represented in 70% of case studies assessing ecosystem services, while only 12% assessed the demand side and 17% assessed both sides simultaneously. In this study, we found that 59% of mapping studies assessed supply exclusively, 15% assessed demand exclusively, and 26% assessed both sides simultaneously. This finding indicates that there has been mild to moderate progress in the consideration of delivery components across value domains. Lautenbach et al. (2019) also found that 56% of assessed indicators were expressed in biophysical terms, 32% in monetary terms, and 15% by use of rankings (i.e., sociocultural valuation approach). Our study found that 80% of mapped indicators were expressed in biophysical terms, 8% in monetary terms, and 21% in sociocultural terms. This finding indicates that there has been a moderate to large decline in the consideration of distinct value domains across indicators. Differences in this study's findings and the findings of Lautenbach et al. (2019) could be attributed to variations in methodological aspects. Regardless of these variations, these findings reinforce those of previous reviews, which unambiguously found a biased representation of distinct value domains across assessments (Maes et al., 2012; Martinez-Harms & Balvanera et al., 2012; Crossman et al., 2013; Luederitz et al., 2015; Malinga et al., 2015).

A further study of the drivers and consequences of this observation is needed. An important driver might be that the actual supply of ecosystem services is often smaller than their societal demand (e.g., carbon sequestration, PM_{10} retention). This can generate bias towards a "more means better" perspective when evaluating the supply of ecosystem services, while overlooking their actual use by individuals from diverse backgrounds (e.g., income, education, age, gender, health status, cultural background, individual perceptions, institutional perceptions). Another driver might be that the spatial quantification of ecosystem services mainly takes place in the field of ecology, with a lower representation in the socioeconomic sciences. This could be a consequence of long-standing critique on methodological and ethical aspects attributed to the valuation of nature (Scholte et al., 2015; Chee, 2004; Gómez-Baggethun & Ruiz-Pérez, 2011). Despite shortcomings associated with these approaches (Scholte et al., 2015; Chee, 2004), the socioeconomic valuation of ecosystem services has numerous advantages. It serves as a common language for raising awareness on the functionality of ecosystems for human well-being, endorsing the integration of environmental externalities within decision-making (Schröter et al., 2014). It also enables consideration of stakeholder perceptions within assessments, adding political legitimacy to environmental decision-making (Díaz et al., 2018). Hence, the demand side of delivery deserves a more prominent place within ecosystem service assessments, considering the perspectives of diverse disciplines (Haines-Young & Potschin-Young, 2018; Wei et al., 2017; Villamagna et al., 2013). By juxtaposing how much of a service is used against how much is desired by society, incentives can be formulated to optimise the distribution of ecosystem service supply. An optimal distribution should not just consider people's consumption possibilities but also their needs and preferences, if the ecosystem services concept is to be used to endorse inclusive well-being. Bearing this in mind, the need for proper recognition of the demand side does not imply that reduced efforts should take place in the assessment of the supply side. Our limited understanding of the complex mechanisms through which ecological processes take place calls for rigorous and robust models for their assessment in order to support the scientifically sound management of Earth's ecosystems.

2.5.2.2 Sections

A glance at the representation of sections indicates that regulation and maintenance services are overrepresented across the mapping literature, compared to provisioning and cultural services. In particular, provisioning services mapping has been sharply decreasing in recent years. A driver of this decline might be that, due to their tangible character, provisioning services are often included in economic markets, facilitating their valuation by use of economic valuation methods. Hence, it is possible that long-standing critique on the methodological and ethical aspects of economic valuation methods (Chee, 2004; Gómez-Baggethun & Ruiz-Pérez, 2011) may not only explain the substantial underrepresentation of ecosystem services in monetary units, but also the underrepresentation of provisioning services. In contrast, cultural services representation has been increasing, with a sharp increase taking place in 2019. This may have resulted from recent calls for better integration of pluralistic values within assessments, which received broad attention within the ecosystem services discipline (Díaz et al., 2018; Peterson et al., 2018). A particularly interesting finding is that regulation and maintenance services were mainly constrained to the supply side of ecosystem service delivery, while provisioning and cultural services were mainly constrained to the demand side. This raises questions regarding the general utility of classifying ecosystem services into sections.

We expand on two situations where ambiguity could result from the operationalisation of ecosystem services into sections. First, a single ecosystem service could constitute more than one section based on the indicator implemented for its reification. For instance, mushroom picking could be considered a provisioning service if measured by its volume (i.e., actual supply), nutritional value (i.e., use), or economic value (i.e., use). It could also be considered as a cultural service if viewed as a recreational or cultural activity (i.e., use). For this reason, during the systematic review it sometimes became difficult to determine which section an ecosystem service belonged to, in situations where it was not made explicit by the authors. Second, a single ecosystem service could constitute more than one section based on the stage of the delivery process under assessment. For instance, pollination would be generally considered as a regulation and maintenance service, as it constitutes an ecological process. However, an analysis of its socioeconomic benefits (i.e., demand side) could lead to it being considered as a provisioning or cultural service. If the benefits of pollination are quantified in terms of its contribution to the production yield of honey or crops, measured in terms of their volume (i.e., actual supply) or market value (i.e., use), pollination could be perceived as a provisioning service. If the benefits of pollination are quantified in terms of its contribution to the availability of flowers (i.e., actual supply) that contribute to the aesthetics of the landscape, pollination could be perceived as a cultural service. In these two examples, pollination constitutes a central aspect of ecosystem service supply, ultimately contributing to the delivery of provisioning and cultural services. The recognition of an ecosystem service as belonging to a particular section is thus determined by the endpoint under assessment (i.e., intermediate or final). If the aim of operationalisation systems is to contribute to clarity and consistency, it might be more pragmatic to operationalise ecosystem services into delivery components rather than into sections. Despite this implication, ecosystem service sections constitute simplified concepts that can be communicated to nonexpert receptors of assessment results with relative ease. Hence, use of the concept might be well-suited during stakeholder engagement in the early and final phases of assessments.

2.5.3 Limitations

The framework presented in this study provides but one of various potential interpretations of the ecosystem service delivery process. Like other operationalisation frameworks, it is meant to be viewed as a baseline for assessing ecosystem services in a consistent manner and for aiding assessors facing issues regarding the conceptualisation of delivery components. Despite the novelty of all frameworks and concepts analysed, choices were made regarding the terms and concepts that would define delivery components in the proposed framework. Given the array of ambiguous terms and concepts adopted across the literature, this framework both aligns and conflicts with various ground-breaking frameworks. The purpose of the presented framework was not to create a new operationalisation framework but rather to synthesise and redesign existing frameworks to promote consistency and clarity. Despite potential conflicts with existing concepts and terminology, we conjecture that clarity and consistency could contribute to a higher uptake of the many terms and concepts that have been developed through the numerous efforts of scientists from various disciplines.

The systematic analysis of the literature led to a number of significant conclusions regarding the consideration of distinct value domains in the mapping literature. It is important to bear in mind that these results bear uncertainty and should thus be interpreted as such. For instance, grey literature was not considered in this study. This suggests that the results might be affected by academic writing aims and standards. This could in turn lead to bias towards assessments considering less ecosystem services yet in a more comprehensive manner. Perhaps this partially explains the identified bias towards the assessment of supply indicators, as well as regulation and maintenance services. Given the complexity of ecological processes and the consequences that assessment results could entail for the actual management of ecosystems, it is vital that models for the spatial quantification of ecological processes display a high degree of rigour and robustness. This uncertainty was partially corrected for by systematically assessing studies that mapped at least three ecosystem services simultaneously. The tendency towards the spatial quantification of the supply side could also partially be explained by the structure of the proposed framework, implemented in the systematic review. In the framework, ecosystem service use does not consider biophysical but only socioeconomic aspects of delivery. As some studies consider biophysical aspects as a constituent of ecosystem service use, this could lead to an overestimation of supply side representation. Despite this possibility, this does not alter the observation that anthropogenic aspects of delivery are largely underrepresented in the mapping literature. This finding aligns with those of preceding systematic reviews, adding credibility to this observation.

2.6 Conclusions

This study's aim was to provide clarity and consistency regarding the operationalisation of the ecosystem service delivery process across value domains, and to evaluate whether progress has been made regarding the consideration of diverse value domains within multifunctional mapping studies. First, we made an attempt at bringing clarity and consistency to structural ambiguities regarding the operationalisation of the delivery process and the conceptualisation of its components. This resulted in the development of a framework synthesising different elements from well-established frameworks for operationalising the delivery process. A key methodological advantage of the presented framework is that it provides a consistent yet flexible approach for operationalising the delivery process. Its flexible character enables the application of various spatial quantification methods based on the preferred approach of individual assessors. A clear cut conceptualisation of delivery components supports their consistent and unambiguous assessment. This is important for avoiding potential overlaps of the supply side and the demand side within studies, given their fundamental dichotomy and interwoven relationship that deserves proper consideration within assessments. Second, we analysed recent literature mapping ecosystem services within multifunctional landscapes to evaluate current progress in the consideration of different value domains. A systematic review of the literature revealed that supply side consideration continues to overshadow demand side consideration within mapping studies. In addition, regulation and maintenance services are generally overrepresented, provisioning service representation has been sharply declining, and cultural service representation has been gradually increasing within mapping case studies. The indicators adopted to represent ecosystem service delivery components are generally constricted to biophysical indicators, with sociocultural indicators gaining pace and monetary indicators largely overlooked.

Our findings strongly align with previous findings, exhibiting mild to moderate progress in the representation of distinct value domains across ecosystem service assessment studies. This calls for increased efforts for greater diversity in value domains considered within assessments, with enhanced integration of perspectives from a wider range of disciplines. The complexity of ecological processes explains the need for rigorous and robust models for assessing the supply side of delivery, as well as regulation and maintenance services. However, the demand side, encompassing sociocultural and economic aspects of delivery, clearly deserves a more prominent place in assessments than the one it is currently assigned. A better integration of the demand side should consider both the use of and societal demand for ecosystem services, as well as monetary estimates of ecosystem service use. While contested, the monetary valuation of ecosystem service benefits is key for creating awareness of the many benefits that ecosystems generate for humans, many of which are overlooked within markets. It is also necessary for building a case for investments in the sustainable management of natural capital, which could otherwise be viewed as an economic burden (e.g., due to potential maintenance costs and foregone opportunities for alternative land uses). It is the demand side that determines the very nature of ecosystem services, as it determines the functionality of the vast range of ecological resources and processes that we rely on. In creating a better understanding on how these ecosystem functions are produced and how they contribute to human well-being, it is possible to optimise their use in an equitable manner and ensure their sustainable use for current and future generations.

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3

CHAPTER 3

Towards nationally harmonised mapping and quantification of ecosystem services

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Abstract

The EU 2020 Biodiversity Strategy requests EU Member States to map and assess ecosystem services within national territories, and to promote and integrate these values into policymaking. This calls for standardised and harmonised data, indicators, and methods to assess ecosystem services within national boundaries. Current approaches for assessing ecosystem services often oversimplify cross-scale heterogeneity, sacrificing the spatial and thematic detail required to support the needs and expectations of decision-makers at different levels. Hence, nationally harmonised models for mapping and quantifying ecosystem services are needed. This paper presents the Natural Capital Model (NC-Model), a spatially-explicit set of models for quantifying and mapping ecosystem services within the Netherlands. Its aim is to support the integration of ecosystem services within spatial planning and policymaking at the national level, contributing to the fulfilment of national and international environmental policy targets. Models introduce previously unexplored combinations of explanatory variables for modelling ecosystem functions and the socioeconomic benefits they accrue, making use of publicly-available and high-resolution spatial data. To capture spatial and thematic heterogeneity across the urban-rural gradient, the NC-Model comprises a subset of ecosystem service models tailored to the urban environment. To demonstrate the model's application, we expand on six urban ecosystem service models and implement them to quantify and map ecosystem services for Municipality of Amsterdam. High-resolution ecosystem supply and use maps provide detailed spatial information useful for supporting spatial planners and decision-makers who wish to optimise the allocation of natural elements while supporting the needs of citizens. They paint a picture on the interlinkages that exist between natural elements, ecosystem functions, and socioeconomic well-being in a friendly manner, tailored to various audiences with differing priorities. Their open-access nature enables their customisation, supporting the sharing of knowledge and data to endorse ecosystem service modelling efforts by external parties within and outside the Netherlands.

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3.1 Introduction

Ever since the release of the Millennium Ecosystem Assessment (MA; 2005), the need to integrate ecosystem services within policymaking has gained prominence, targeted by initiatives such as the Convention on Biological Diversity (CBD, 2010), Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES; Pascual et al., 2017), and U.N. Sustainable Development Goals (SDGs; U.N., 2017). At the global scale, Aichi Biodiversity Targets 14 and 15, formulated under the CBD, call for the protection and enhancement of ecosystem services by ratifying parties (CBD, 2010). At the EU level, the EU 2020 Biodiversity Strategy called EU Member States to map and assess ecosystem services within national territories, and to promote and integrate these values into national accounting and reporting at the national and EU level (EC, 2011). At the urban level, Target 2 from the Strategy calls Member States to maintain and enhance ecosystem services by restoring and promoting green infrastructure (EC, 2013, 2019a). To monitor developments towards these objectives, standardisation and harmonisation of data, indicators, and methods to assess ecosystem services, are necessary (Schröter et al., 2016). This is instrumental for systematically monitoring the impact of policies on ecosystems and the socioeconomic benefits they support (Zulian et al., 2014).

Despite the need for a common evidence base, a "one-size-fits-all" approach is difficult to attain due to scale-dependent conditions (Schröter et al., 2016; Martínez-López et al., 2019). The urban-rural and local-global gradients are characterised by heterogeneous landscape structures, land uses, climates, administrative structures, and demographic variability (Larondelle & Haase, 2013; Martín-López et al., 2012; Schram-Bijkerk et al., 2018). This leads to variations in the performance of ecosystem functions (i.e., ecological structures and processes that satisfy human needs; de Groot et al., 2002; Potschin and Haines-Young, 2011) as well as in socioeconomic factors that determine the exposure to and consumption of ecosystem services (Keeler et al., 2019). This cross-scale heterogeneity is often oversimplified within standardised ecosystem service assessment approaches, sacrificing the spatial and thematic detail associated with different geographical locations and extents (Derkzen et al., 2015; Martínez-López et al., 2019). This calls for scale-dependent harmonisation, supporting the needs and expectations of decision-makers at different levels (Martínez-López et al., 2019; Hauck et al., 2013). Harmonisation at different scales is especially useful when it can lead to better informing decision-makers and integrating ecosystem services within policymaking and spatial planning (Breure et al., 2012).

Contributing to this need, recent years have seen a rise in the number of tools and guidelines for conducting ecosystem service assessments at different geographical locations and extents. At the large scale, tools such as InVEST (Integrated Valuation of Ecosystem Services and Trade-offs; Tallis & Polaski, 2011), ARIES (Artificial Intelligence for Ecosystem Services; Villa et al., 2014), and IMAGE (Integrated Model to Assess the Global Environment; Doelman et al., 2018) have received broad attention and uptake (Bagstad et al., 2013b). At the EU-level, approaches for assessing ecosystem services at the pan-European and regional scales are under development through initiatives such as MAES (Mapping and Assessment of Ecosystems and their Services), ESMERALDA (Enhancing ecoSysteM sERvices mApping for poLicy and Decision mAking), and ESTIMAP (Ecosystem Services Mapping Tool; Maes et al., 2015a; Vihervaara et al., 2019; Zulian et al., 2014). Despite their usefulness at relatively large scales, most of these approaches lack the level of spatial and thematic detail required to conduct assessments at the regional and local level (Hauck et al., 2013; Derkzen et al., 2015; Martínez-López et al., 2019). Customizable ecosystem service models provide a useful solution to this issue (Villa et al., 2014; Maes et al., 2015a), yet customisation requires (i) readily-available place-specific knowledge and data that can directly substitute pre-set knowledge and data; or (ii) readily-available models integrating place-specific knowledge and data to directly substitute pre-designed models (Martínez-López et al., 2019). Recognising these needs, governments are (i) establishing platforms for data harmonisation and sharing (EC, 2007; Cetl et al., 2017; VROM, 2010), and (ii) developing approaches for assessing ecosystem services within national boundaries (UK NEA, 2011; EME, 2012, 2014; NOU, 2013; de Knegt, 2014, 2019; Rugani et al., 2014).

In this paper, we present the Natural Capital Model (NC-Model), a spatially explicit set of models for quantifying and mapping ecosystem services within the Netherlands at the local, regional, and national level. The aim of the NC-Model is to support the integration of ecosystem services within spatial planning and policymaking within the Netherlands, contributing to the fulfilment of national and international environmental policy targets (EZ, 2013a; EC, 2011, 2013, 2019a; CBD, 2010). The model is continuously under development and improvement by a collaboration of Dutch knowledge institutes (i.e., National Institute of Public Health and the Environment, RIVM; Wageningen ENvironmental Research, WENR; Netherlands Environmental Assessment Agency; PBL), fostering knowledge exchange and reducing overlapping modelling efforts within national borders. The first models were originally translated from ecosystem service models developed for Flanders by the Belgian knowledge institute VITO (Staes et al., 2017; Jacobs et al., 2016) and applied to the Netherlands to develop maps for the Netherlands Atlas of Natural Capital (Remme et al., 2018; Paulin et al., 2019; https://atlasnaturalcapital.nl).

Key methodological advantages of the NC-Model include (i) its contribution to universal models for quantifying and mapping ecosystem services, and (ii) its consideration of spatial and thematic detail and heterogeneity relevant at the regional and local level (e.g., urban, rural). The NC-Model builds on existing process-based approaches for quantifying and mapping ecosystem services by introducing previously unexplored combinations of explanatory variables for modelling ecosystem functions and the socioeconomic benefits they accrue. All ecosystem service models make use of standard publicly-available input datasets and can be customised for their use by parties within or outside the Netherlands. Spatial detail is considered by making use of fine-detail local data, including population and remotely-sensed vegetation maps at a high resolution (10 x 10 m). Accounting for thematic detail requires considering scale-specific linkages between ecological and socioeconomic factors that determine the production and consumption of ecosystem services (Martínez-López et al., 2019; Keeler et al., 2019). The model realises this by assimilating quantitative relationships between ecological, social, and economic parameters within the Netherlands, established within empirical studies. To account for the heterogeneity that characterises ecosystem service production and consumption patterns across the urban-rural gradient, the NC-Model additionally comprises a subset of ecosystem service models tailored to the urban environment, namely the Urban Natural Capital Model (Urban NC-Model; Remme et al., 2018).

In this paper, we (i) describe the NC-Model, (ii) present six completed urban ecosystem service models, and (iii) demonstrate their application within the Municipality of Amsterdam. Section 3.2 describes the mechanism behind the NC-Model and delves into models for the ecosystem services: Air Quality Regulation, Physical Activity, Property Value, Urban Cooling, and Urban Health. Section 3.3 presents and analyses quantification and mapping results and expands on their potential use to support decision-making. Section 3.4 presents the concluding remarks.

3.2 Materials and methods

The NC-Model comprises an extensive set of models for quantifying and mapping ecosystem services, classified according to the Common International Classification of Ecosystem Services (CICES; Haines-Young & Potschin, 2018). These include the recreational potential provided by natural landscapes, natural pollination, natural pest control, and water purification, among others. Output (i.e., ecosystem service maps and total quantities) is produced by use of algorithms combining formulas and input data, including (i) a standardised set of spatial data and (ii) reference values obtained from empirical studies capturing linkages between variables. Reference values are incorporated into algorithms either directly by incorporating them into formulas, or through their prior integration into look-up tables. Detailed input data and model descriptions, including stepwise procedures for their direct replication, are found in the Supplementary Material (Appendices 2 – 1 and 2 – 2). All codes and model outputs are available from the authors upon request. As all input necessary for model implementation is readily available, key user requirements include thorough knowledge of spatial modelling and coding. Model customisation additionally requires a deep understanding of ecosystem functions and their relationships with socioeconomic parameters relevant for the model under customisation, as well as spatial data and reference values to substitute custom input. Customisation may be desirable for improving model inaccuracies, integrating novel insights and data, developing scenarios to support decision-making, or for translating models into different spatial contexts (Zulian et al., 2014; Martínez-López et al., 2019).

We present six models for mapping and quantifying ecosystem service supply and use in urban areas. Urban models are highly relevant in the Netherlands, where the population is expected to increase from roughly 17 million in 2018 to 18.4 million inhabitants in 2060, and where most of the population is concentrated in urban areas (https://opendata.cbs.nl/; Stoeldraijer, et al., 2017). Figure 3.1 presents a schematic diagram illustrating the interlinkages between supply and use indicators, as well as key input variables that influence their quantity and distribution. Supply proxy indicators capture the performance of ecosystem functions, such as the filtration of atmospheric concentrations of particulate matter by vegetation and water. Use indicators capture realised social or economic benefits that result from the performance of ecosystem functions. Social benefits include the contribution of green space to human health or to people's inclination to engage in outdoor physical activity, among others. Economic benefits reflect either the direct contribution of natural capital to economic markets (e.g., the effect of vegetation and water on property value) or the translation of social benefits into monetary units (e.g., economic gains from enhanced health due to the presence of green space). Descriptions for all indicators in Figure 3.1 are provided in Table 3.1.

To ensure consistency in model output for all applications, all ecosystem service models use a standard set of spatial data as input (Table 3.2). Most datasets are publicly available at standard international data repositories, such as the INSPIRE (https://inspire-geoportal.ec.europa.eu/) and ESRI geoportals (http://esrinl-content.maps.arcgis.com/), and all datasets can be found at governmental data registries. Since spatial datasets are difficult to attain for the same year, we included the most recent versions of datasets or versions that align well with the general dates of all datasets. In addition to land use data (*Ecosystem Unit Map, EUM*) and given its subjectivity at the local scale (Rabe et al., 2016), the NC-Model makes use of high-resolution (10 x 10 m) vegetation maps as intermediate input. Vegetation maps are derived using the national digital elevation model (*Actueel Hoogtebestand Nederland, AHN2*), based on LiDAR (Light Detection and Ranging), and high-resolution (25 x 25 cm) aerial photography (*Luchtfoto*). National digital elevation data is used to derive a layer with the height of objects. The red and infrared bands from the aerial photograph of the Netherlands are used to derive a Normalised Difference Vegetation Index (NDVI) layer, which distinguishes between vegetation and other objects (Huang et al., 2008). Next, an overlay operation is performed to obtain vegetation height. This results in three separate layers, including low vegetation (<1 m high), shrubs and bushes (1-2.5 m high), and trees (>2.5 m high). Another key intermediate input for various models is the high-resolution (10 x 10 m) population map, which captures the human component of such models and directly influences ecosystem service supply and use. The population map is derived by assigning neighbourhood-specific population statistics (*Wijk-en buurtkaart*) to housing units (*Basisregistratie Adressen en Gebouwen, BAG*). Within the Supplementary Material, we provide (i) additional information on the contents and availability of input datasets (Appendix 2 – 1), and (ii) a stepwise procedure for developing vegetation and population maps (Appendix 2 – 2).

Table 3.1: Main input and output indicators for six urban ecosystem services, based on Figure 3.1 (cell size = 10 x 10 m). PM_{10}= Particulate matter up to 10 micrograms, UHI= urban heat island, GP= general practitioner.

Indicator	Input/output	Unit	Description
Contribution to cycling (commuting)	Output (use)	minutes/ year	The contribution of green space surrounding a household to the time individuals spend cycling for commuting purposes
Contribution to property value	Output (use)	€	Contribution to residential property value by surrounding green areas and water
PM_{10} concentration	Input	µg/m^3/year	PM_{10} atmospheric PM_{10} concentration
PM_{10} retention	Output (supply)	µg/year	Amount of atmospheric PM_{10} retained by vegetation and water
Population	Input	Inhabitants	Number of inhabitants per cell
Reduced costs from reduced mortality	Output (use)	€/year	The economic benefit of avoided premature deaths from reduced all-cause mortality, based on the European default value of a statistical life (VSL; Kahlmeier et al., 2017)
Reduced health costs	Output (use)	€/year	Savings in health costs due to reduced years of lost life (YOLL) as an effect of reduced atmospheric PM_{10} concentrations (CE Delft, 2017)
Reduced mortality	Output (use)	avoided deaths/year	The number of avoided premature deaths from the reduced risk of all-cause-mortality from cycling
Reduced health costs	Output (use)	€/year	Reduced public health costs resulting from urban-green related health enhancements
Reduced labour costs	Output (use)	€/year	Reduced labour costs from urban-green related enhancements to the health of employees
Reduced rainwater in sewers	Output (supply)	m^3/year	The amount of water that is stored by vegetation and hence does not end up in the drainage system

Reduced UHI effect	Output (supply)	°C/year	The cooling effect of vegetation and water in the direct surroundings of a location (30 metres)
Reduced visits to GP	Output (use)	visits/year	Reduced number of visits to general practitioner per year as a result of the amount of green space surrounding an area
Reduced water treatment costs	Output (use)	€/year	The reduction in water treatment costs associated with reductions of rainwater in the drainage system
Vegetation and water	Input/ Output (supply)	Percentage cover	The percentage of a cell that is covered by vegetation (trees, bushes/shrubs, and low vegetation) or water
Wind speed	Input	m/s	Average wind speed at 100 m height

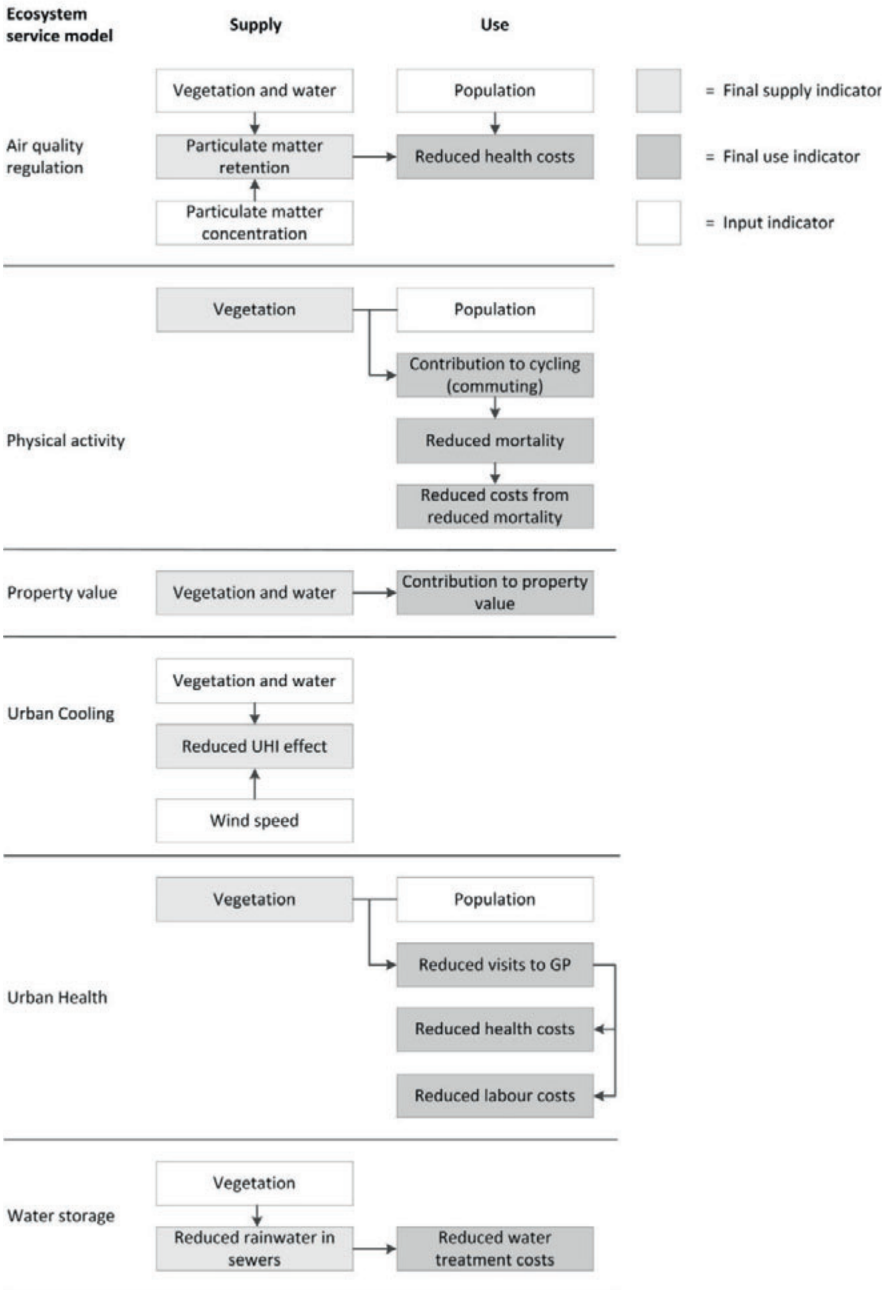

Figure 3.1: Schematic overview of six ecosystem service models from the Urban NC-Model. Indicators presented (white boxes) either directly or indirectly influence the final supply (light grey) or final use (dark grey) of ecosystem services

Table 3.2: Standard input spatial data for modelling six ecosystem services at the urban scale. Extensive details on dataset content, sources, and availability is found in the Supplementary Material (Appendix 2 – 1)

Dataset name	Description	Resolution	Year
Actueel Hoogtebestand Nederland (AHN2)	Elevation data	0.5 x 0.5 m	2015
Basisregistratie Adressen en Gebouwen (BAG)	Basic registry of addresses and buildings	10 x 10 m	2016
Basisregistratie Gewaspercelen (BRP)	Agricultural areas of the Netherlands	10 x 10 m	2017
Bevolkingskernen	Contour of populated areas	10 x 10 m	2011
Ecosystem Unit Map (EUM)	Land use map of the Netherlands	10 x 10 m	2017
Fijnstof 2017 (pm10)	Concentration of particulate matter up to 10 micrograms	50 x 50 m	2017
Luchtfoto	High resolution aerial photograph of the Netherlands	0.25 x 0.25 m	2017
Top10NL	Topographic land use map of the Netherlands	5 x 5 m	2017
Wijk- en buurtkaart	District and neighbourhood data	10 x 10 m	2017
Windsnelheden op 100m hoogte	Average wind speed at 100 m altitude	2.5 x 2.5 km	2015
WOZ-waarde	Real estate value	10 x 10 m	2016

To demonstrate the NC-Model's application and its use to support decision-making, all six models were applied to quantify and map ecosystem services for the Municipality of Amsterdam. Amsterdam is the capital of the Netherlands and its most populated city, with more than 850,000 inhabitants and a population density of roughly 5,200 inhabitants/km^2 (https://www.amsterdam.nl/ois; Figure 3.2). Its surface area covers 219 km^2, distributed among water bodies (24.9%), built-up areas (e.g., residential and commercial buildings; 35.8%), semi built-up areas (e.g., dump sites and cemeteries; 5.6%), agricultural areas (e.g., greenhouses; 11.6%), recreational areas (e.g., sport grounds and allotment gardens; 11.8%), woodland and nature (2.7%), and transport infrastructure (7.6%). We applied the urban models here presented to provide an overview of ecosystem service values generated by Amsterdam's green and blue infrastructure, to support the 'Quality Impulse Green' (*KwaliteitsImpuls Groen*; Amsterdam Municipality, 2017a). The Quality Impulse Green is a spatial plan developed by the Municipality of Amsterdam, which aims to strengthen green and blue infrastructure (i.e., vegetation and water) in alignment with the municipality's demographic trends and economic ambitions (Amsterdam Municipality, 2017a). Input maps showing the distribution of vegetation, water, and inhabitants, are displayed in Figure 3.3. The values for total tree, bushes/shrubs, and grass coverage in the municipality are 5%, 2%, and 20% respectively. Algorithms were written in Python programming language, using the PCRaster software to perform spatial calculations (https://www.python.org/; http://pcraster.geo.uu.nl/). All models are described in brief below and more extensively in the Supplementary Material (Appendix 2 – 2).

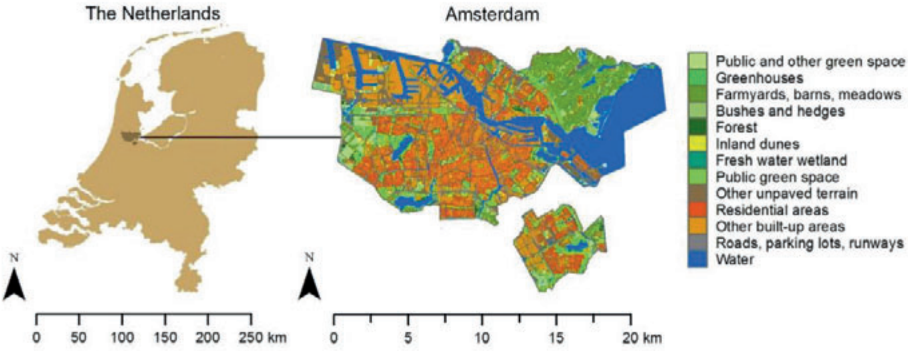

Figure 3.2: Dominant land cover in the Municipality of Amsterdam. Based on the 2017 Ecosystem Unit Map (EUM) of the Netherlands developed by Statistics Netherlands (CBS; Eden & Van Leeuwen, 2016).

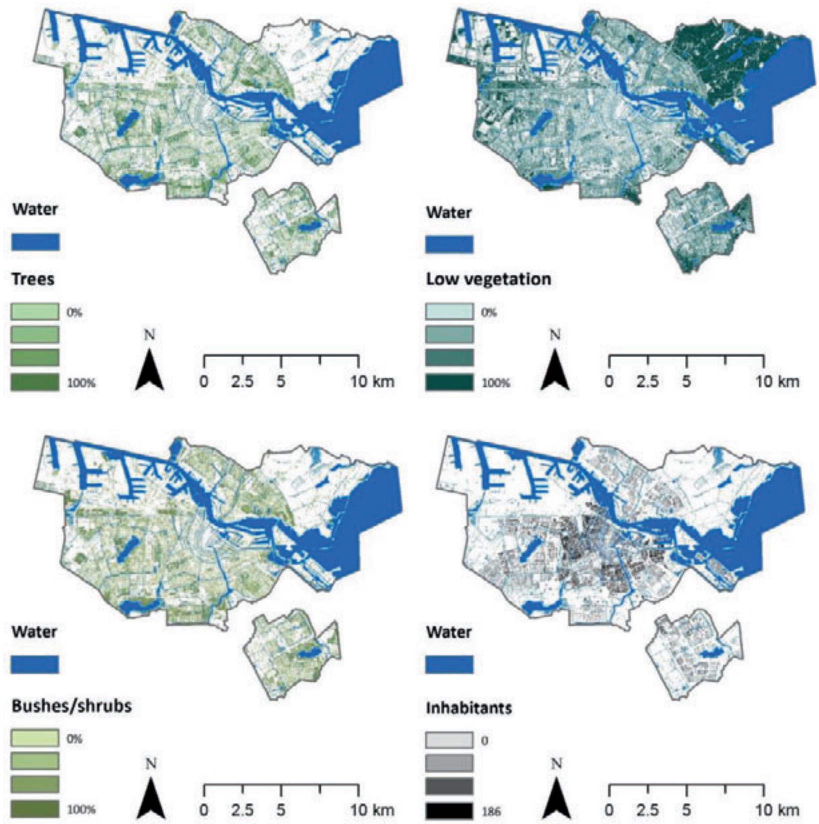

Figure 3.3: Vegetation cover (percentage of trees, shrubs/bushes, low vegetation) and number of inhabitants, per cell (10 x 10 m) within the Municipality of Amsterdam. For each map, legends show quantile values. All quantile thresholds values are presented in the Supplementary Material (Table A2 - 7, Appendix 2 – 3)

3.2.1 Air Quality Regulation

Air pollution is a common problem within cities, caused by factors such as traffic and industry. The most harmful component of air pollution for human health is particulate matter, which is associated with respiratory and cardiovascular diseases, as well as mortality (Derkzen et al., 2015; Santibañez et al., 2013). The ability of particles to enter the human body is determined by their diameter, with smaller particles entering the lungs and airways with more ease. Once entering the body, particulate matter with a diameter of up to 10 micrometres (PM_{10}) can cause cardiovascular disease (Cassee et al., 2013; Gerlofs-Nijland et al., 2019). Because of the roughness of their surface, different types of vegetation can contribute to capturing particulate matter (Remme et al., 2018; Janhäll, 2015). The urban Air Quality Regulation model estimates the contribution by water and different vegetation types to reductions in atmospheric PM_{10} concentrations in Dutch cities. It considers three important factors as determinants for atmospheric self-depuration: the deposition velocity and resuspension of suspended particles, and the total concentration of PM_{10} in the air. The deposition velocity is the speed with which particulate matter deposits to the natural surface (Chen et al., 2012). Resuspension occurs when deposited particles are re-emitted into the air due to various factors (e.g., physical characteristics of the contaminated surface, physicochemical nature of the contaminant, meteorological conditions), leading to the redistribution of particles (Gradoń, 2009). Additionally, the model can be implemented to calculate the effect of reductions (or increases) in PM_{10} concentrations on human health, calculated as the reduction (or increase) in health costs associated with reduced (or increased) mortality (CE-Delft, 2017).

3.2.2 Physical Activity

Exposure to green space can affect people's behaviour, including their inclination to engage in outdoor physical activity. Physical activity is beneficial to human health, promoting physical and mental health across lifespans (Staatsen et al., 2017; Klompmaker et al., 2018; WHO, 2016). Despite the potential risks associated with engaging in active transport (e.g., cycling, walking), such as exposure to air pollution or traffic accidents, recent reviews have shown that the benefits of engaging in outdoor physical activity generally outweigh the costs (Staatsen et al., 2017; Kelly et al., 2014). The urban Physical Activity model captures the effect of urban green space on physical activity, as well as the resulting health benefits and economic gains. Based on the work of Maas (2009), the model calculates the time cycled by individuals to-from work that can be attributed to the availability of green space in their surroundings. The health benefits and resulting economic gains of cycling are calculated based on the methodology underlying the Health Economic Assessment Tool (HEAT), a tool developed by the World Health Organization that calculates the health and economic benefits of walking and cycling (Kahlmeier et al., 2017). The tool translates the time cycled by individuals to reduced mortalities, based on empirically-established quantitative relationships. It then calculates the associated economic gains from reduced mortalities, based on the value of a statistical life. Custom reference values used by the tool were tailored to values relevant in the Dutch context. A detailed description of the Physical Activity model, including reference values and their origin, is found within the Supplementary Material (Appendix 2 – 2).

3.2.3 Property Value

Natural and semi-natural elements in cities, such as trees, parks, gardens, and water, increase the amenity of residential areas, which is reflected in property values (Czembrowski & Kronenberg, 2016; Franco & Macdonald, 2018). Studies in the Netherlands have shown that a positive relationship exists between property prices, and vegetation and open water cover (Daams et al., 2016; Ruijgrok & de Groot, 2006; Luttik & Zijlstra, 1997). Based on these studies, the urban Property Value model captures the contribution to property prices by vegetation and open water. The model takes into consideration the availability of green and blue elements and their proximity from people's households.

3.2.4 Urban Cooling

Cities recurrently experience higher temperatures than their surrounding rural areas, a phenomenon commonly referred to as the ‘urban heat island (UHI) effect’ (Rizwan et al., 2008). The UHI effect exacerbates heat extremes and is one of the leading causes of health hazards in cities (Lauwaet et al., 2018). The UHI effect is mainly a consequence of anthropogenic released heat (e.g., cars and industry) and of the heavy use of synthetic construction materials that store and re-radiate large amounts of heat (Rizwan et al., 2008). The roughness of infrastructure additionally reduces wind speed and hence its contribution to heat removal and transfer (Rizwan et al., 2008). Unsealed soil, vegetation, and surface water have a cooling effect during high temperatures. Vegetation increases the evaporation capacity of an area and provides shade, while soil releases heat more quickly than sealed areas (Akbari et al., 2001; Lauwaet et al., 2018). The cooling effect of vegetation has made planting vegetation the most widely adopted mitigation measure taken to tackle heat extremes in cities (Rizwan et al., 2008). Building on Lauwaet et al. (2018), the Urban Cooling model captures the reduction in the UHI effect by vegetation. The UHI effect is estimated as a function of vegetation cover, impervious cover, population density, and wind speed.

3.2.5 Urban Health

Vegetation influences human health in cities by mitigating pressures such as noise pollution, air pollution, and temperature extremes (Staatsen et al., 2017; Hartig et al., 2014; James et al., 2015). Evidence suggests that green space leads to improved health (e.g., improved cognitive function, improved psychological well-being, reduced prevalence of type 2 diabetes, reduced adverse pregnancy outcomes; Staatsen et al., 2017; Bratman, et al., 2019; Gascon et al., 2016) and reduced all-cause mortality (e.g., all-cause cardiovascular disease mortality; Staatsen et al., 2017; Kondo et al., 2018). The NC-Model captures the effect of urban green space on health and labour costs resulting from improved health conditions. Building on the TEEB-Stad Tool (https://www.teebstad.nl; KPMG, 2012; Maas, 2009) the contribution of green space to health is calculated as (i) the reduced costs associated with the incidence of seven disease categories (i.e., cardiovascular diseases, musculoskeletal diseases, mental diseases, respiratory diseases, neurological diseases, digestive diseases, and a miscellaneous category) and (ii) reduced number of visits paid to general practitioners. Inspired by the study ‘The Economics of Ecosystems and Biodiversity’ (Sukhdev & Kumar; 2008), the TEEB-Stad Tool enables the wider public to quantify the economic benefits of green and blue elements within cities in the Netherlands. The quantification of labour costs resulting from improved health conditions include reductions in costs associated with absenteeism, reduced labour productivity, and job losses (KPMG, 2012; Steenbeek et al., 2010).

3.2.6 Water Storage

Water storage by vegetation and soils is a crucial function in cities, as urban flooding around the world becomes more prominent and damaging in response to climate change (Van Herk et al., 2011). Urban flooding is closely linked to the expansion in impervious cover and reduction in vegetation cover that is required to build infrastructure for growing urban population (Wang et al., 2008). The replacement of vegetated cover by impervious surfaces decreases infiltration by compacting soils, decreases evaporation by reducing soil water volumes, and decreases interception through vegetation removal (Wang et al., 2008). The result is increased rainwater runoff that is charged with excess pollutants that are left unfiltered by vegetation and soils, and a higher risk of flooding. With 26% of its surface area under sea level, the Netherlands has developed cutting-edge technology and expertise that enabled it to become the best-protected delta in the world (PBL, 2014). Despite this advantage, 59% of the country is still under threat of flooding (PBL, 2014). Large infrastructure alone cannot meet the increasing challenges that climate change poses, calling for an integrated spatial planning approach that considers not only technological solutions but also nature-based solutions (Van Herk et al., 2011). The urban Water Storage model captures the avoided amount of rainwater in the drainage system due to water storage by vegetation, as well as the associated reduction in water treatment costs.

3.3 Results and discussion

Urban ecosystem service models were implemented to quantify and map indicators displayed in Figure 3.1 and Table 3.1. Total ecosystem service supply and use values are presented in Table 3.3. Maps for all six ecosystem services, each represented by one supply or use indicator, are presented in Figure 3.4.

Table 3.3: Output supply and use values of six ecosystem services for the Municipality of Amsterdam

Ecosystem Service	Supply/ Use	Indicator	Unit	Value
Air Quality Regulation	Supply	PM_{10} retention	thousand kg/yr	99
	Use	Reduced health costs	million €/yr	4.3
Physical Activity	Use	Contribution to cycling (commuting)	thousand hours/yr	50
	Use	Reduced mortality	lives/yr	18
	Use	Reduced costs from reduced mortality	million €/yr	38
Property Value	Use	Contribution to property value	billion €	6.2
Urban Cooling	Supply	Reduction in UHI effect	°C	1.8
Urban Health	Use	Reduced visits to GP	thousand visits/yr	21
	Use	Reduced health costs	million €/yr	18
	Use	Reduced labour costs	million €/yr	88
Water Storage	Use	Reduced rainwater in sewers	million m^3/yr	18
	Use	Reduced water treatment costs	million €/yr	14

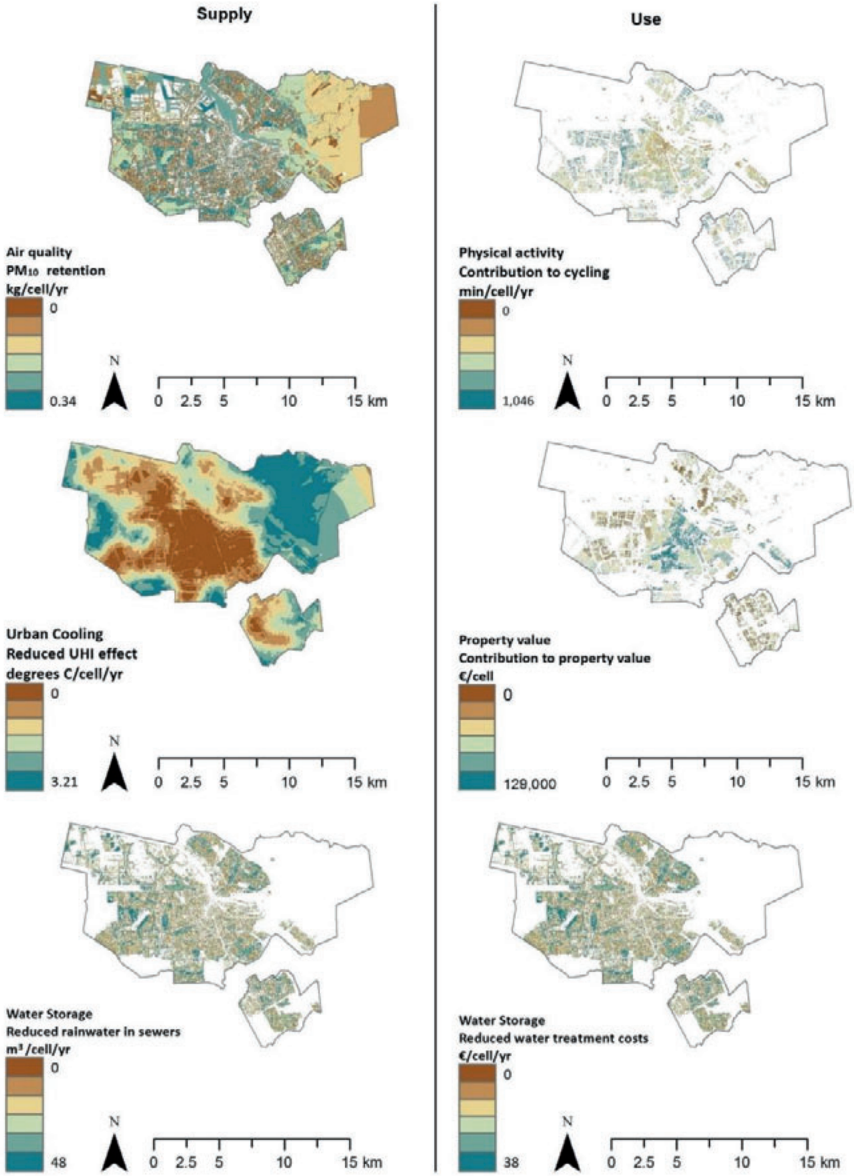

Figure 3.4: Output maps of six urban ecosystem services (cell=10 x 10 m). One supply or use indicator is represented per ecosystem service, based on the indicators presented in Figure 3.1 and Table 3.1. For each ecosystem service, legends show quantile values. All quantile thresholds values are presented in the Supplementary Material (Table A2 - 8, Appendix 2 – 3).

In Table 3.3, total ecosystem service supply and use values are expressed in biophysical, social, and economic units and, in some cases, through the use of more than one indicator. A frequent concern associated with quantifying indicators in various units is that it leads to an "adding-apples-and-oranges-situation", obstructing their comparability and potential aggregation (Satz et al., 2013). Despite these disadvantages, considering multiple indicators and in multiple units is central for the holistic assessment of ecosystem services. First, it is not always possible to quantify the supply and use of an ecosystem service in identical units (Alam et al., 2016), yet these two components of ecosystem service delivery are closely interlinked. Ecosystem service supply is linked to the provision of ecosystem functions (Syrbe & Walz, 2012), which is best represented through the use of biophysical indicators. Ecosystem service use reflects the socioeconomic benefits that these ecosystem functions generate for people (Syrbe & Walz, 2012), hence best represented by social and economic indicators. For instance, vegetation and water lead to the yearly reduction of 99,000 kg of atmospheric PM_{10} in Amsterdam, yet its value is only cultivated if this reduction leads to social or economic gains, in this case valued at €4.3 million/year. This brings us to the second point: different indicators speak to different audiences (Satz et al., 2013). A decision-maker that prioritises the contribution of natural capital to the economy may be interested in the effect of green and blue infrastructure on property value (€6.2 billion) or on reduced labour costs (€88 million/year). A decision-maker focused on enhancing human well-being may be interested on the contribution of green space to human health (e.g., reduction of 21,000 yearly visits to general practitioners). A policymaker dealing with climate change may prioritise natural capital's contribution to the reduction of the UHI effect (1.8 °C). The matter of prioritisation brings us to the third point: expressing all ecosystem service use values in monetary units can be subjective and misleading (Satz et al., 2013). The aforementioned examples show how prioritising monetary values when assessing the utility of green and blue infrastructure will undeniably shift the priority to property values, while other benefits are perhaps more critically needed in cities like Amsterdam, where only a few get to benefit from increased property values. When dealing with such a complex system, variety in choice of indicators may bring perspective, yet at the cost of simplicity.

3

Aggregation of ecosystem service indicators is a common practice within ecosystem service assessments, which requires commensurability among aggregated indicators. This can be done, for instance, by expressing ecosystem service values in monetary terms or transforming them into dimensionless values (Alam et al., 2016; Satz et al., 2013). Aggregation can be adopted to provide information on the extent and magnitude of ecosystem service bundles, and for quantifying composite indicators that enable the assessment of trade-offs and synergies among variables (Alam et al., 2016). Despite these advantages, we refrain from aggregating ecosystem service indicators, as it may lead to the overestimation or underestimation of relative ecosystem service values, thus hampering the objectivity of an assessment. Over- or underestimation of ecosystem service values can occur (i) if monetary values are disproportionately higher or lower than those of other ecosystem services, (ii) if several or no monetary values are available for an ecosystem service, or (iii) if double-counting takes place. Disproportionate variations in ecosystem service monetary values result from market imperfections (Bunse et al., 2015). For instance, property values are often subject to property bubbles, which highly affect property values and hence the attributed contribution by vegetation and water. Another example occurs with common-good (i.e., rivalrous, non-excludable) and public-good (i.e., non-rivalrous, non-excludable) ecosystem services, which are often free and non-marketed (Fisher et al., 2009; Bunse et al., 2015), so people often lack awareness of the role they perform in their everyday lives. For instance, PM_{10} retention by vegetation is freely accessible to everyone (non-excludable) yet is limited by the availability of vegetation (rivalrous). Its non-marketed and, in this case, invisible nature make this ecosystem function and its benefits to humans difficult to perceive. Moreover, aggregation of monetary values may lead to overestimation if more indicators can be aggregated for one ecosystem service than for others, and to underestimation if no monetary indicator is available for an ecosystem service. One last problem with aggregation is that it may lead to double-counting. This may occur if indicators overlap, which is often the case due to the abstract nature of ecosystem functions and their benefits, and to the strong interlinkages among them (Gunton et al., 2017). For instance, overlaps may occur between the indicators for reduced mortalities from increased cycling (Physical Activity), reduced health costs due to the reduction of seven types of diseases (Urban Health), and reduced health costs from reduced atmospheric PM_{10} concentrations (Air Quality Regulation). There may even be overlaps between different indicators for a single ecosystem service. Due to all abovementioned factors, aggregation is discouraged.

Supply maps in Figure 3.4, including Air Quality Regulation, Urban Cooling, and Water Storage, show the complexity with which green and blue infrastructure perform ecosystem functions. Within the Air Quality map, the capture of PM_{10} relies on two main factors: the type of vegetation and the total concentration of PM_{10} in an area. Trees and water have the highest capacity for PM_{10} retention, followed by shrubs and low vegetation (in descending order). Densely populated areas with a high degree of human activity often experience relatively high concentrations of particulate matter and thereby experience greater PM_{10} uptake where vegetation or water is present. However, vegetation cover is less prominent in densely populated areas, where infrastructure is predominant. This explains the high degree of fragmentation in PM_{10} uptake visible in the most densely populated parts of the city. The north-eastern part of Amsterdam seems to experience lower PM_{10} uptake compared to densely populated areas. This occurs since population density in the northeast is substantially low, resulting in lower overall atmospheric PM_{10} concentrations. Additionally, these areas are characterised by a predominant low vegetation cover, which retains less PM_{10} than water, trees, and shrubs and bushes. Within the Urban Cooling map, the UHI effect is a function of three main variables: soil sealing (including built-up areas), population density, and wind speed. The UHI effect is most prominent in areas where population density is highest and where impervious cover is predominant. Areas where the reduction of the UHI is highest encompass larger extents of semi-natural and agricultural land, with where low population densities and impervious cover predominate. Within the Water Storage model, the reduced amount of rainwater in sewers relies on two main factors: vegetated cover, which determines the amount of rainwater stored, and population concentrations, which act as an indicator for the presence of extensive sewage systems. Hence, water storage is correlated with the percentage of vegetated cover in Amsterdam and is visible in the populated fraction of the municipality.

Use maps in Figure 3.4, including Physical Activity, Property Value, and Water Storage, show the close relationship that exists between ecosystem service use and the distribution of ecosystem service beneficiaries (population). The Physical Activity map shows the total amount of minutes cycled per cell that can be attributed to the availability of green space in an area. The Property Value map shows the contribution to property value that can be attributed to green and blue elements. The Water Storage map shows the monetary contribution of water storage by vegetated surfaces to reduced water treatment costs. At a first glance, the Physical Activity and Property Value maps show strong similarities. This is the case since both the number of individuals benefitting from increased cycling and the property value are linked to the distribution of housing units. However, taking a closer look will reveal that the distribution of ecosystem service values strongly differs between both maps, with the highest and lowest amounts of benefits taking place in different areas. The Physical Activity map relies mainly on the distribution of inhabitants and the amount of green space surrounding an area. This is why the most densely populated and vegetated areas experience the highest benefits. The Property Value map is closely linked to property values; hence ecosystem service use values are highest in neighbourhoods with the highest property prices, even when green and blue elements are not predominant. The Water Storage use map shows a direct translation of the reduced rainwater in sewers from the Water Storage supply map into reduced water treatment costs. Hence, there is a full correlation between the Water Storage supply and use maps. Given that extensive sewage systems are linked to densely populated areas, the map also shows a close relationship to population distribution. However, ecosystem service values are more closely linked to the percentage of vegetated cover, which ensures water storage.

Output generated by use of the NC-Model provides useful insights on ecosystem functions, how their performance is affected by the distribution of natural elements, and how this in turn affects human well-being. However, models possess drawbacks that limit their objectivity, and which should be considered when using model output to support decision-making. Society and ecological systems are extremely complex and are influenced by a perhaps infinite number of variables in a continuously changing fashion. As such, it becomes difficult to capture all relevant factors that determine the supply or use of an ecosystem service at the desired level of accuracy within models. For instance, the supply of Air Quality Regulation fails to capture the negative effect of trees within street canyons (Janhäll, 2015) and the Urban Cooling model does not consider the cooling effect of soils in cities, as a consequence of the lack of empirical findings necessary to integrate these factors. Another limitation hindering model accuracy comes from the spatial extrapolation of data, which requires making generalisations that do not always align with reality. Extrapolation inaccuracies are found both within input data obtained from various sources and within model output. For instance, the Physical Activity model provides information on the additional time people spend cycling due to the presence of green space in their surroundings. While this extrapolation is based on empirical findings linking green spaces and cycling behaviour in the Netherlands (Maas, 2009), the distribution of additional minutes cycled presented in supply maps is based on a simplified reality, hence meant to be viewed as an indicator for the benefits provided by green infrastructure. Validation of ecosystem services could serve as a potential solution to assess model accuracy. However, this is not possible for most ecosystem services due to privacy concerns associated with the relevant indicator (e.g., the reduction of seven disease groups; the amount of time cycled by individuals) or due to their subjectivity (e.g., the contribution of natural elements to property value). In cases where validation is possible (e.g., PM_{10} retention by vegetation and water; water storage), it is often time consuming and expensive.

Despite these drawbacks, the NC-Model takes a step towards nationally harmonised mapping and quantification of ecosystem services. First, detailed maps are needed for meeting national and international environmental policy targets (EZ, 2013a; EC, 2011, 2013, 2019a; CBD, 2010), and for supporting decision-making at the regional and local level (Hauck et al., 2013). The NC-Model makes use of the best available spatial data, accepted and endorsed by Dutch national and local governments, to map and quantify ecosystem services at a high resolution. The diversity of indicators to quantify ecosystem services offered speaks to different audiences and suits different contexts. This presents an opportunity for decision-makers to make choices in alignment with different priorities and circumstances. Second, the combination of quantification and high resolution mapping of ecosystem services is a powerful communication tool to inform local decision-makers and spatial planners concerned with the optimal allocation of natural elements to endorse the realisation of socioeconomic gains. Supply maps communicate the complexity with which ecosystem functions take place, revealing how the design and choice of green and blue infrastructure can affect the overall supply of ecosystem services (Janhäll, 2015). Use maps are an effective way of displaying the distribution of ecosystem service benefits, which is central to addressing the issue of unequitable distribution of ecosystem services (Potschin & Haines-Young, 2011). The juxtaposition of supply and use maps tells the story behind the nature with which ecosystem functions take place (supply maps), and how these ultimately lead to socioeconomic gains (use maps). Third, publicly-available models can be customised to improve model inaccuracies, integrate novel insights and data, develop scenarios, or translate models to different geographical locations and extents (for an example of scenario development using the NC-Model to support spatial planning, see Paulin et al., 2019). Customisation can be done by replacing custom with place-specific input datasets and reference values. This may prove difficult in situations where these inputs are not readily-available yet can be corrected for by assimilating similar datasets and reference values (perhaps more) relevant at different spatial contexts. Input deficiency may also bring attention to the need for site-specific data and reference values necessary for developing parallel national and subnational ecosystem service assessment approaches.

3.4 Conclusions

International environmental policy-targets call for nationally harmonised approaches for quantifying and mapping ecosystem services (EC, 2011, 2013, 2019; EZ, 2013a; CBD, 2010). This paper presented the NC-Model, a Dutch approach for quantifying and mapping ecosystem services within national boundaries. The model contributes to national harmonisation efforts by synthesising the knowledge of experts from national research institutes and integrating best-available datasets endorsed by national and local governments. Mapping national ecosystem services and integrating them into policymaking requires user-friendly, high-resolution maps that meet the needs of local decision-makers (Hauck et al., 2013; Martínez-López et al., 2019). High-resolution ecosystem supply and use maps in the NC-Model provide detailed spatial information useful for supporting spatial planners and decision-makers who wish to optimise the allocation of natural elements while supporting the needs of citizens. They paint a picture on the interlinkages that exist between natural elements, ecosystem functions, and socioeconomic well-being in a friendly manner, tailored to various audiences with differing priorities. The open-access nature of models enables their customisation, supporting the sharing of knowledge and data to endorse ecosystem service modelling efforts by external parties within and outside the Netherlands.

A key limitation of the NC-Model concerns its inability to capture all relevant factors that contribute to the supply and use of ecosystem services, affecting model accuracy and hence the objectivity of assessments. This is problem is not unique to the NC-Model, as it is virtually impossible for any model to capture all factors in socioecological systems that affect the production and consumption of ecosystem services. To improve accuracy in output from spatially explicit ecosystem service models, we recommend conducting empirical research capturing the relationship between available spatial data, and proxy indicators for ecosystem functions and socioeconomic well-being. Such research should be conducted at different scales and locations, capturing spatial and thematic heterogeneity across geographical locations and extents. However, they should make use of similar techniques, ensuring the harmonised integration of reference values into models at different scales and locations. This will facilitate the comparability of results across space and the substitutability of reference values to suit different scales and locations. The integration of quantitative scale-specific empirical evidence on the relationships between ecological, social, and economic parameters within assessment tools will support a more accurate depiction of reality, endorsing higher objectivity in assessments. Capturing spatial and thematic detail at various scales and locations will additionally provide the choice of integrating variables into models based on their relevance in particular contexts, suiting the needs and expectations of decision-makers at different levels (Martínez-López et al., 2019; Hauck et al., 2013).

This paper demonstrates how the NC-Model can be implemented to quantify and map ecosystem services in the Dutch context for informing decision-makers and spatial planners. For a more thorough assessment of ecosystem services, this approach could be accompanied by a systematic assessment of trade-offs and synergies, hotspots and coldspots, or an analysis of correlations among ecosystem service input and output maps (Wang et al., 2017; Rabe et al., 2016; Li et al., 2016). The NC-Model may be adapted for use in other contexts by adaptation of its open access models to local data-availability and reference values. Models are under constant improvement by developing parties and open to recommendations from interested external parties. In the future, they may be expanded and integrated with similar models that are under development by national research institutes (Remme et al., 2018), such as the Netherlands Natural Capital Accounts, under development by WENR and CBS (Graveland et al., 2018).

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4

CHAPTER 4

Application of the Natural Capital Model to assess changes in ecosystem services from changes in green infrastructure in Amsterdam, the Netherlands

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Abstract

This paper demonstrates the utility of local models for assessing ecosystem services to support urban planning. It does so by application of the NC-Model, a spatially-explicit set of models for assessing ecosystem services in the Netherlands, to assess changes in ecosystem services in the Municipality of Amsterdam given the implementation of strategies from the Green Quality Impulse. The Green Quality Impulse is a spatial plan that envisions the development of Amsterdam's green infrastructure by 2025 to support the needs of Amsterdam's growing population. The NC-Model was implemented to spatially quantify six ecosystem services within a 'business-as-usual' scenario (only residential and population expansion considered) and three scenarios that capture changes in green infrastructure from the implementation of strategies from the Green Quality Impulse. Incorporation of local knowledge and data enabled quantification of ecosystem services at a high spatial resolution and identification of key factors that influence ecosystem service delivery. Such an approach can support urban planners who wish to better-understand the mechanism by which green infrastructure generates value for urban dwellers, to develop scientifically-sound spatial strategies that optimise ecosystem service supply and use, and to further communicate this information to decision-makers, investors, and local inhabitants in an accessible manner.

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4.1 Introduction

The future of the world's population is urban (UN, 2019). Urban areas are commonly associated with economic growth, poverty reduction, and human development, behaving as centres for high-skilled labour, business, and knowledge exchange (UN, 2019). Despite these benefits, urbanisation also leads to an increase in the occurrence of environmental hazards and health risks. For instance, the substantial replacement of vegetated cover by impervious cover to support infrastructure expansion has led to prominent flooding around the world (Van Herk et al., 2011; Wang et al., 2008). Air pollution released by traffic and industry increases the risk of respiratory and cardiovascular diseases, as well as mortality (Derkzen et al., 2015; Santibañez et al., 2013). Anthropogenic heat release (e.g., from cars, industry, houses) and the inability of heat absorption by synthetic construction materials contribute to the Urban Heat Island (UHI) effect, a leading cause of health hazards in cities (Rizwan et al., 2008; Lauwaet et al., 2018). Addressing the challenges posed by urbanisation requires promoting a better understanding of the interconnected nature of green infrastructure and socioeconomic human well-being, supporting evidence-based urban planning (Haase et al., 2014; Keeler et al., 2019; Luederitz et al., 2015).

Green infrastructure (GI; i.e., soil, vegetation, and water; sometimes denoted as green and blue infrastructure) provides urban dwellers with valuable 'ecosystem services', or the contributions by natural capital (i.e., Earth's ecosystems and underpinning geo-physical systems) to human well-being (Haines-Young & Potschin, 2018). More specifically, ecosystem services encompass functional ecological structures and processes (ESP) and the socioeconomic benefits that they generate for humans. By mitigating pressures such as noise pollution, air pollution, and heatwaves, GI contributes to improved physical and mental health (Kruize et al., 2019; Staatsen et al., 2017), as well as reduced all-cause mortality (Staatsen et al., 2017; Kondo et al., 2018). GI mitigates the magnitude of peak runoff from precipitation events by redirecting or absorbing precipitation or by providing retention space for surplus water (Gunnell et al., 2019). Uncovered soil releases heat quicker than sealed areas, while vegetation increases an area's evaporation capacity and provides shade, together generating a cooling effect during heat extremes (Akbari et al., 2001; Lauwaet et al., 2018). Trees, parks, gardens, canals, and other GI increase the amenity of residential areas, which reflects in higher property values (Czembrowski & Kronenberg 2016; Franco & Macdonald, 2018). A variety of GI typologies (e.g., urban and peri-urban forests, tree-lined streets, peri-urban agriculture, brownfields) act as basis for nature-based recreational activities that support physical activity, social interactions, empowerment, and social cohesion (Cortinovis et al., 2018). The growing awareness of the many benefits that GI generates has led to an upsurge in the number of initiatives endorsing the integration of ecosystem services into urban planning (EC, 2013, 2019a; https://www.c40.org/; https://www.iclei.org/; https://www.nature4cities.eu). Despite these initiatives, the translation of the ecosystem services concept from discourse to practice in the urban context remains limited (Hansen et al., 2015; Haase et al., 2014).

Integrating ecosystem services into urban planning requires readily-available ecosystem service assessment approaches that capture spatial and thematic detail relevant at various urban contexts (Paulin et al., 2020b; Keeler et al., 2019; Luederitz et al., 2015). GI distribution is grounded in sociocultural influences, such as histories of land development or evolving ideas about leisure and recreation (Wolch et al., 2014). In general, the pathway by which GI leads to ecosystem service delivery is highly contextual and hence not uniform across space (Luederitz et al., 2015). The same size, configuration, and composition of GI can lead to differential ecosystem service distribution, influenced by local ecological and socioeconomic characteristics (Keeler et al., 2019; Grafius et al., 2018). From an ecological perspective, the distribution of ESP is influenced by factors such as climate and landscape in heterogeneous spatial and temporal gradients (Paulin et al., 2020b). From a socioeconomic perspective, the realisation of benefits supported by ESP is determined by factors such as accessibility and safety in green spaces, as well as sociodemographic characteristics (e.g., income, education, age, gender, health status, cultural background, individual perceptions, institutional perceptions; Kruize et al., 2019; Murali et al., 2019; Luederitz et al., 2015). The degree of spatial and thematic heterogeneity that characterises the urban landscape suggests that a universal toolkit for assessing the value of urban nature is unlikely to occur (Keeler et al., 2019). This calls for location-specific ecosystem service assessment approaches that capture ecological and socioeconomic detail relevant at the urban level (Luederitz et al., 2015; Keeler et al., 2019).

Such an approach is offered by the Natural Capital Model (NC-Model), a spatially explicit set of models for quantifying and mapping ecosystem services and accrued socioeconomic benefits within the Netherlands (Paulin et al., 2020b; Remme et al., 2018). To account for heterogeneity that characterises the urban environment, the NC-Model comprises a set of urban ecosystem service models. These models capture spatial detail by incorporating best-available local data, including population and remotely-sensed vegetation maps at a high resolution (10 x 10 m; Paulin et al., 2020b; Remme et al., 2018). Thematic detail is captured through assimilation of quantitative relationships between ecological and socioeconomic parameters respective to the Netherlands (Paulin et al., 2020b). Urban ecosystem services research is often conducted from an ecological perspective, resulting in a lack of full engagement with all aspects of the ecosystem services cascade (Gómez-Baggethun & Barton, 2013; cascade model available in Potschin & Haines-Young, 2016). This limits the understanding necessary for ecosystem services management and integration into sustainable urban planning (Gómez-Baggethun & Barton, 2013). Urban ecosystem service assessment approaches should address ecological, economic, and also societal issues that determine the distribution and final use of ecosystem services (Gómez-Baggethun & Barton, 2013). The NC-Model captures ecological and socioeconomic factors contributing to ecosystem service delivery by spatially quantifying ESP underpinned by GI, and the social and economic benefits ESP support.

This paper presents an application of the NC-Model to assess the effect of changes in GI on ecosystem services and human well-being in the Municipality of Amsterdam. Amsterdam's population, alongside its number of residential units, is expected to increase substantially within upcoming years (Amsterdam Municipality, 2017b). To meet the socioeconomic needs of its inhabitants in the face of rapid urbanisation, the Municipality has developed a number of policy initiatives, seeking to maintain and enhance the quality of public spaces (Amsterdam Municipality, 2010, 2015a, 2015b, 2017a, 2018, 2019a, 2019b). This assessment was performed to support the Green Quality Impulse (*KwaliteitsImpuls Groen*; Amsterdam Municipality, 2017b), a spatial plan for the expansion and improvement of Amsterdam's GI by the year 2025. The Green Quality Impulse envisions Amsterdam's expansion as a transition into a sustainable, climate-proof, and socially attractive city, in alignment with its demographic trends and economic ambitions (Amsterdam Municipality, 2017a). The aim of this study is to demonstrate the utility of the NC-Model for assessing urban ecosystem services to support urban planning. This paper builds on Paulin et al. (2020b), which presents the first complete set of urban ecosystem service models available in the NC-model. Assessment results disclose information on how enhancements in GI may lead to changes in the distribution, relative performance, and overall performance of ESP and ecosystem services. This information is instrumental to support urban planning in the context of (i) communication and awareness raising; (ii) strategic planning and priority setting; and (ii) economic accounting and incentive design (Gómez-Baggethun & Barton, 2013; Haase et al., 2014).

4.2 Materials and methods

4.2.1 Study site

In the Netherlands, 92% of the population is concentrated in urban areas (UN, 2019). Amsterdam, its most populous municipality, is home to more than 850,000 inhabitants (https://data.amsterdam.nl/), confined to an area of 219 km^2 (Figure 4.1). More than a third of the municipality's surface area consists of infrastructure and other built-up areas, a fourth comprises water bodies, and the rest consists of mainly semi-natural areas. Characterised by its complex canal structure, historically and culturally rich architecture, and thriving economy, Amsterdam is a hotspot for tourism and an attractive destination for local and international people aspiring for a place to live. By 2025, Amsterdam's population is expected to increase by roughly 70,000 (Paulin et al., 2019), which will be made possible by the creation of around 5,000 residential units per year (Amsterdam Municipality, 2017b). The expansion in grey infrastructure (i.e., built-up and paved areas) necessary to support the municipality's growing needs exerts ever increasing pressure on GI, the ESP it supports, and the essential benefits it provides to urban dwellers.

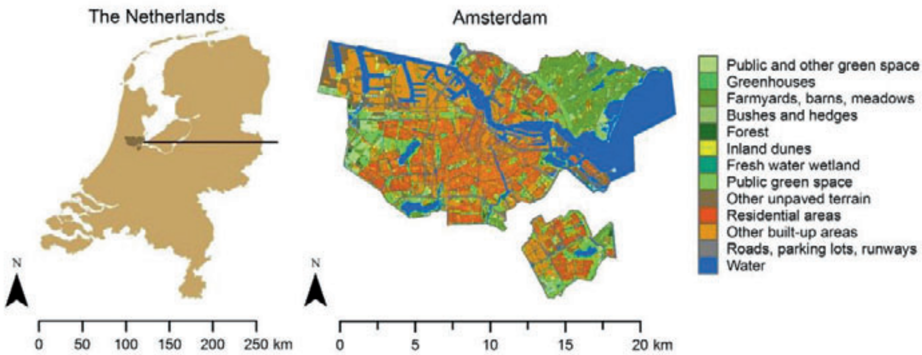

Figure 4.1: Dominant types of land cover in the Municipality of Amsterdam (Paulin et al., 2020b)

4.2.2 Assessment approach

To evaluate the effect of changes in ecosystem services resulting from the implementation of GI strategies, an assessment was conducted, consisting of four main stages. The first stage consisted of a workshop with decision-makers from the Municipality of Amsterdam, facilitated by the National Institute for Public Health and the Environment (RIVM) and De Urbanisten, an innovative consultancy firm for urban research and landscape design based in the Netherlands (http://www.urbanisten.nl/). To support the translation of objectives in the Green Quality Impulse into realistic spatial strategies, decision-makers were asked (i) to state their expectations regarding the introduction of new GI; and (ii) to share their knowledge on relevant trends and potential limitations for the formulation of spatial strategies. During the workshop, ecosystem service supply and use indicators were selected for their assessment. On the second stage of the assessment, urban design firm De Urbanisten developed GI spatial scenarios for the year 2025, capturing strategies from the GI plan and considering requirements established during the first workshop. In addition to changes in the municipality's GI, scenarios capture expected changes in residential infrastructure, as well as a projected population increase of roughly 70,000 inhabitants. On the third stage of the assessment, executed by RIVM, ecosystem service supply and use were quantified and mapped for three GI scenarios and for a reference scenario, where no alterations to GI take place. This enabled the comparison of values from GI scenarios with those of the reference scenario to estimate the effectiveness of different strategies. Changes considered encompass (i) expected changes in the performance and spatial distribution of ESP given the application of GI strategies, and (ii) the effects of such changes on the value and distribution of socioeconomic benefits across the city. The fourth stage of the assessment consisted of a workshop, facilitated by RIVM and De Urbanisten, where GI scenarios and expected associated changes in ecological and socioeconomic factors were presented to members of the Municipality of Amsterdam. In this final stage, workshop participants were provided the opportunity to express their views and opinions regarding results and the way were developed and communicated.

4.2.3 Modelling approach

Ecosystem services were assessed by use of the NC-Model. The model combines formulas and input data (i.e., spatial data and reference values) into algorithms to quantify and map ecosystem service supply and use for any given scenario (provided that input data requirements are met). Ecosystem service supply captures the distribution and total performance of final ESP that contribute to human well-being (Paulin et al., 2020b). Ecosystem service use captures the distribution and total value of realised socioeconomic benefits underpinned by ESP (Paulin et al., 2020b). As input, all models make use of a standard set of spatial data (details specified in Appendix 3 – 1, Supplementary Material). Key intermediate input for all urban ecosystem service models includes detailed maps (10 x 10 m resolution) showing the distribution of vegetation and population across urban areas (stepwise procedure for creating vegetation and population maps available in Paulin et al., 2020b). In order to develop scenarios, input spatial data can be adapted to reflect changes that are expected to occur in each scenario (e.g., changes in the configuration or composition of vegetation). Ecosystem services were assessed for the year 2025, when Amsterdam's GI plan will reach its completion phase. Throughout the assessment process, only changes in GI that could be reflected as changes in model input, were included. Hence, some changes that could potentially affect ecosystem service delivery but are not included in the NC-Model, such as enhancements in the quality of GI, were not considered for this assessment.

Ecosystem services were quantified and mapped by use of six urban ecosystem service models comprised within the NC-Model (Paulin et al., 2020b), namely the Air Quality Regulation, Physical Activity, Property Value, Urban Cooling, Urban Health, and Water Storage models. While the NC-Model captures ecosystem service and benefit indicators in a way that is compatible with the Common International Classification of Ecosystem Services (CICES; Haines-Young & Potschin, 2018; https://cices.eu), the names ascribed to models comprise user-friendly terms that are instrumental when involving stakeholders and decision-makers perhaps less knowledgeable on ecosystem services terminology and concepts. Table 4.1 presents supply and use proxy indicators that were spatially quantified by use of each ecosystem service model. Classification of all proxy indicators according to CICES (version 5.1) is indicated in the Supplementary Material (Table A3 - 2, Appendix 3 – 2). Figure 4.2 illustrates the relationship between ecosystem service supply and use as considered in urban ecosystem service models, in alignment with the ecosystem services cascade (Potschin & Haines-Young, 2016). For some models, supply indicators are not available. This occurs since ecosystem service models often capture the way ecological structures (as opposed to processes) contribute to human well-being. Benefit-generating structures (e.g., vegetation and water) often comprise key model input, making their quantification redundant. The Urban Cooling model was only used to assess ecosystem service supply (i.e., reduction of the UHI effect), as there was no readily available approach for translating GI's contribution to urban cooling into socioeconomic benefits (e.g., enhanced health conditions, reduced health costs, enhanced labour productivity). All models are described in detail in Paulin et al. (2020b), which also comprises an extensive 'Supplementary Materials' section providing stepwise procedures for the implementation of every model.

Table 4.1: Descriptions of quantified and mapped ecosystem service supply and use proxy indicators (Paulin et al., 2020b)

Ecosystem service model	Supply/ use	Indicator	Description
Air Quality Regulation	Supply	PM_{10} retention	Reduction in atmospheric PM_{10} concentrations by vegetation and water
	Use	Reduced health costs	Reduction in health costs from avoided PM_{10} related mortalities
Physical activity	Use	Contribution to cycling (commuting)	Contribution to time cycled by individuals for commuting purposes that can be attributed to the availability of green space in their surroundings
	Use	Reduced mortality	Avoided all-cause mortalities from enhanced health benefits due to the contribution to cycling (commuting)
	Use	Reduced costs from reduced mortality	Economic gains from reduced all-cause mortalities, based on the value of a statistical life
Property value	Use	Contribution to property value	Contribution by vegetation and open water to property prices
Urban cooling	Supply	Reduction in UHI effect	Contribution by vegetation and water to mitigation of the UHI effect
Urban health	Use	Reduced health costs	Reduction in health costs linked to the contribution by green space to mitigating the incidence of seven disease categories (i.e., cardiovascular diseases, musculoskeletal diseases, mental diseases, respiratory diseases, neurological diseases, digestive diseases, and a miscellaneous category)
	Use	Reduced visits to general practitioner	Avoided visits to general practitioners linked to the contribution of green space to improved health conditions
	Use	Reduced labour costs	Reduction in costs of absenteeism, reduced labour productivity, and job losses, linked to the contribution of green space to improved health conditions
Water storage	Supply	Reduced rainwater in sewers	Avoided rainwater in the drainage system due to water storage by vegetation
	Use	Reduced water treatment costs	Reduction in water treatment costs from avoided rainwater in the drainage system

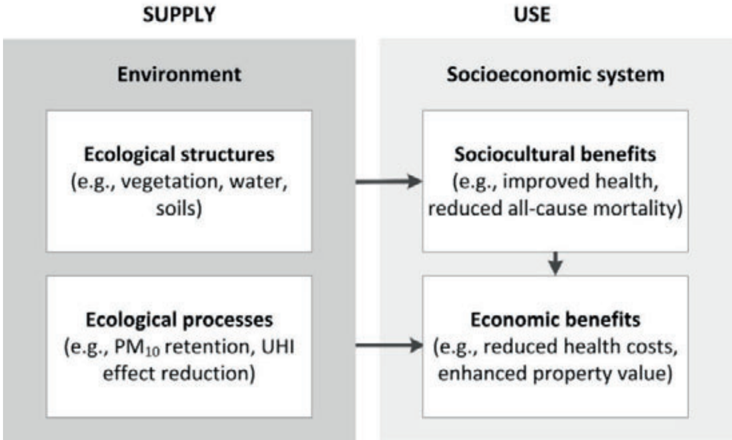

Figure 4.2: Schematic diagram of relationship between ecosystem service supply and use within urban ecosystem service models. Supply captures ecosystem functions and use captures the realised contributions of ecosystem functions to society and the economy. Sociocultural benefits capture the direct contribution of ecological structures and processes to human well-being. Economic benefits capture either (i) the direct contribution of ecological structures and processes to the economy or (ii) the translation of accrued sociocultural benefits into monetary units.

4.2.4 Scenarios

The Business-As-Usual scenario, or reference scenario, portrays a situation, where no changes other than expected population and planned residential expansion occur. Changes in the distribution of infrastructure were based on data from the 'Housing Plans Map' (*Woningbouwplannenkaart*) from the Municipality of Amsterdam (https://maps.amsterdam.nl/). The dataset shows areas where new residential plans have been made for the upcoming years, including the number of housing units that are expected to be built. Plans that fell within the phases 'investment decision taken' and 'in construction' with planned completion in the period 2018-2025', were included. Neighbourhood statistics from the input layer *Basisregistratie Adressen en Gebouwen (BAG*; Kadaster, n.d.) were used to develop a spatially disaggregated map displaying the number of inhabitants that will reside in each new housing unit in the year 2025. Figure 4.3 presents maps showing the distribution of inhabitants and three types of vegetation (i.e., trees, shrubs/bushes, low vegetation), which were developed for the BAU scenario.

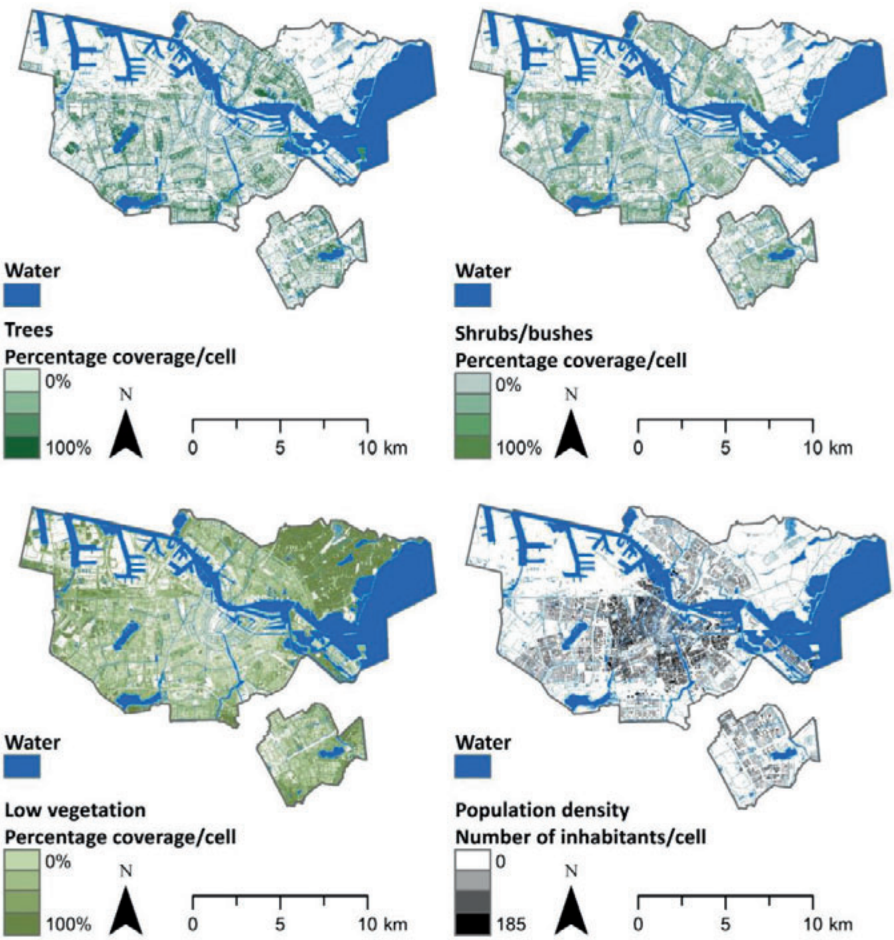

Figure 4.3: Vegetation cover (percentage of trees, shrubs/bushes, low vegetation) and population (number of inhabitants) per cell (10 x 10 m) within the Business-As-Usual scenario (year = 2025). For each map, legends show quantile values. All quantile thresholds values are presented in the Supplementary Material (Table A3 - 3, Appendix 3 – 3).

Ecosystem services were assessed for three GI scenarios simulated for the year 2025, described hereunder (extensive scenario descriptions available in Paulin et al., 2019). Within each scenario, new vegetation types are introduced (Figure 4.4). Table 4.2 presents the total change in spatial area for each vegetation type in each scenario. In some instances, a decrease in spatial area is seen, comprising transformations from current vegetation types to different typologies (e.g., low vegetation to shrubs/bushes or trees).

- **Green Neighbourhoods**: This scenario comprises a substantial increase in vegetation in areas that currently comprise little to no green. The main modifications to GI in this scenario include substantial conversions of parking spaces into green surfaces and grey roofs into green roofs.
- **Green Network**: This scenario envisions a strengthened ecological and recreational (e.g., cycling, sports, hiking trails) network within Amsterdam. The main modifications to GI include completing the main tree network connecting green areas, and the transformation of current vegetation to different typologies (e.g., converting different low vegetation typologies into shrub typologies).
- **Urban Parks**: This scenario integrates objectives from the Green Quality Impulse regarding the enhancement of urban parks for recreational use. The main changes captured within this scenario include the creation of new parks, expansion of existing parks, and increased net abundance of vegetation in existing parks.

Table 4.2: Total vegetation (i.e., trees, shrubs, bushes, low vegetation) coverage for one Business-As-Usual scenario, and total change in coverage (increase/decrease) for three GI scenarios (= ha GI scenario - ha Business-As-Usual scenario)

GI element	**Unit**	**Total cover**	**Change in cover**		
		Business-As-Usual	**Green Neighbourhoods**	**Green Network**	**Urban Parks**
Trees	ha	1,173	-	454	258
Bushes/shrubs	ha	441	-	526	-79
Low vegetation	ha	6,010	249	-570	139
Total	**ha**	**7,623**	**249**	**410**	**318**

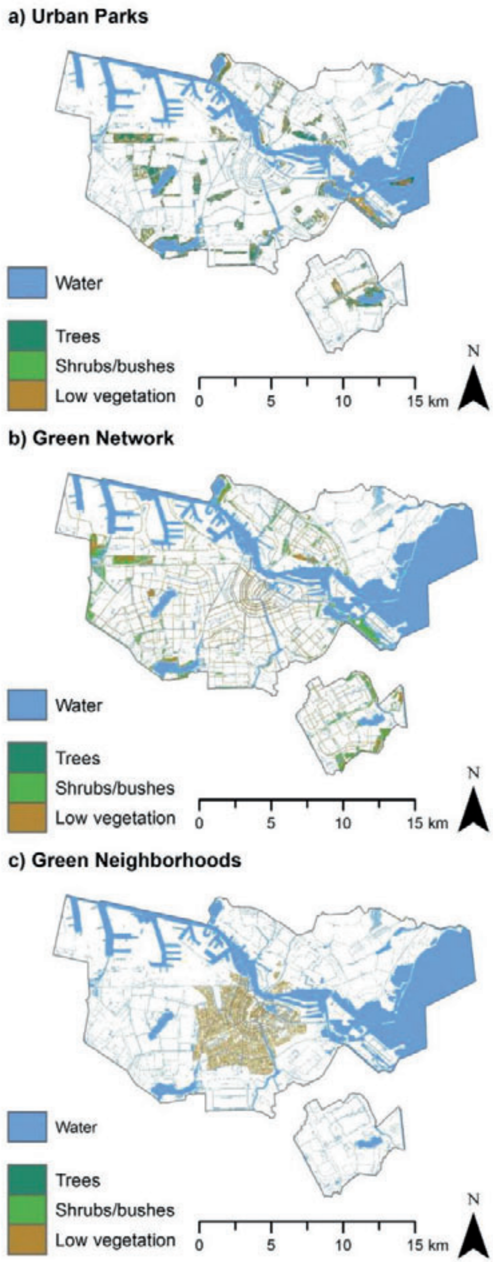

Figure 4.4: Areas where new vegetation cover (i.e., trees, shrubs/bushes, low vegetation) is introduced for the Urban Parks, Green Network, and Green Neighbourhood scenarios. Unshaded areas comprise areas where no value has been assigned.

4.3 Results and discussion

4.3.1 Changes in ecosystem services: Total values

In Table 4.3, data and model outputs are presented, reflecting how changes in GI in each scenario affect total ecosystem service supply and use. First, an increase in value for nearly all ecosystem service indicators across all GI scenarios is expected. This is primarily the case since all scenarios comprise an increase in the extent of vegetated cover. Since ecosystem service models capture ESP and accrued socioeconomic benefits that are underpinned by GI, improvements in Amsterdam's GI generally lead to improvements in overall ecosystem service delivery. Second, the highest improvement in performance for most ecosystem service indicators assessed (7 out of 12) is expected in the Green Neighbourhoods scenario. This is a somewhat unexpected outcome since the scenario (i) comprises the lowest net expansion in vegetated cover (249 ha), and (ii) comprises no transformation from herbaceous to woody vegetation, which leads to a higher contribution to ecosystem delivery within several models (e.g., Air Quality Regulation, Property Value, Urban Cooling). Substantial improvements in ecosystem service use in the Green Neighbourhoods scenario occur since, in addition to the size and composition, the configuration of introduced GI (i.e., location and distribution) plays a key role in ecosystem service performance (Keeler et al., 2019; Grafius et al., 2018). Within the Green Neighbourhoods scenario, GI is introduced in areas (i) where built-up infrastructure predominates (see Figure 4.1 and Figure 4.4) and (ii) which are densely populated. Introducing GI in areas where built-up infrastructure predominates leads to a higher marginal increase in ecosystem service performance than in areas where vegetation is already predominant. This is also the case when GI is introduced in densely populated areas, as they comprise a high concentration of potential ecosystem service beneficiaries (Vallecillo et al., 2018). Third, the Green Network scenario reveals the highest improvement in performance for Air Quality Regulation and Water Storage (supply and use) proxy indicators. Increases in PM_{10} retention and accrued health benefits can be attributed to the substantial expansion of tree cover (454 ha), as trees bear the highest capacity for PM_{10} retention out of all vegetation types covered by the Air Quality Regulation model. The Water Storage model captures the direct relationship between the areal extent of vegetated cover and its capacity for rainwater storage, as well as accrued economic benefits. Hence, substantial improvements in water storage and the economic benefits in the Green Network scenario can be attributed to the substantial net expansion in vegetated cover (410 ha) in the scenario. Finally, the Urban Parks scenario revealed the most substantial improvement in performance for Urban Cooling proxy indicators. This occurs since this scenario entails a substantial increase in tree cover (258 ha) confined to relatively small areas (e.g., parks). In the Urban Cooling model, two central factors contributing to the reduction of the UHI effect are the vegetation typology and vegetation density, where trees in high densities lead to a higher reduction than other vegetation typologies in lower densities.

Table 4.3: Total ecosystem service values for a Business-As-Usual scenario (year = 2025) and differences in values for three GI scenarios in reference to the Business-As-Usual scenario. Green shaded cells encase the highest increases in ecosystem service values per GI scenario per proxy indicator considered (i.e., supply or use).

Ecosystem service model	Supply/ Use	Indicator	Unit	Total value	Difference in value (BAU as reference)		
				Business-As-Usual	Green Neighb.	Green Network	Urban Parks
Air Quality Regulation	Supply	PM_{10} retention	thousand kg/yr	98.6	2.9	8.1	3.3
	Use	Reduced health costs	million €/yr	4.8	0.1	0.4	0.2
Physical Activity	Use	Contribution to cycling (commuting)	thousand hours/yr	54	8	5	2
	Use	Reduced mortality	lives/yr	19	3	2	1
	Use	Reduced costs from reduced mortality	million €/yr	42	6	4	2
Property Value	Use	Contribution to property value	billion €	6.36	0.10	0.08	0.02
Urban Cooling	Supply	Reduction in UHI effect	°C	1.78	0.00	0.01	0.04
Urban Health	Use	Reduced visits to general practitioner	thousand visits/yr	23	3	2	1
	Use	Reduced health costs	million €/yr	20	3	2	1
	Use	Reduced labour costs	million €/yr	96	13	9	4
Water Storage	Supply	Reduced rainwater in sewers	million m^3/yr	17.6	1.2	1.4	0.8
	Use	Reduced water treatment costs	million €/yr	13.8	0.9	1.1	0.6

4

4.3.2 Changes in ecosystem services: Spatial distribution

To better understand how changes in GI affect ESP and accrued socioeconomic benefits, it is useful to juxtapose quantitative results with maps displaying the distribution of changes in ecosystem service performance (Crossman et al., 2012). Figure 4.5 presents maps displaying the distribution of changes in the performance of four ecosystem services, each represented by one supply or use proxy indicator. Notable improvements in ecosystem service performance are visible in areas where GI is introduced (see Figure 4.4), an anticipated result as GI underpins the delivery of ecosystem service supply and use. A strong resemblance is visible between the distribution of changes in ecosystem service use estimated by use of the Physical Activity model (Figure 4.5a) and the distribution of inhabitants (see Population density map, Figure 4.3). This occurs since the distribution of inhabitants serves as a proxy for the distribution of potential ecosystem service beneficiaries. This does not necessarily imply that the spatial distribution of ecosystem service use is always correlated with population distribution, as the mechanisms leading to the realisation of ecosystem service benefits vary in nature. For instance, within the Water Storage model, water retention is directly related to the spatial extent of vegetated cover in an area. Ecosystem service use captures reductions in water treatment costs associated with water stored by vegetated areas in areas with extensive sewage systems. Hence, a strong resemblance is visible between the distribution of changes in ecosystem service use estimated by use of the Water Storage model (Figure 4.5b), and the distribution of introduced vegetation (see Figure 4.4a). Changes in the distribution of ecosystem service supply (Figure 4.5c and d) occur primarily in areas where GI is introduced (Figure 4.4a and b), yet the distribution pattern of such changes varies substantially. This accentuates the complexity with which ESP take place. While no resemblance is visible between population distribution and changes in ecosystem service supply, human populations may play an indirect role in the distribution of ESP (e.g., by manipulating the distribution of GI and exerting ecological pressures that require mitigation).

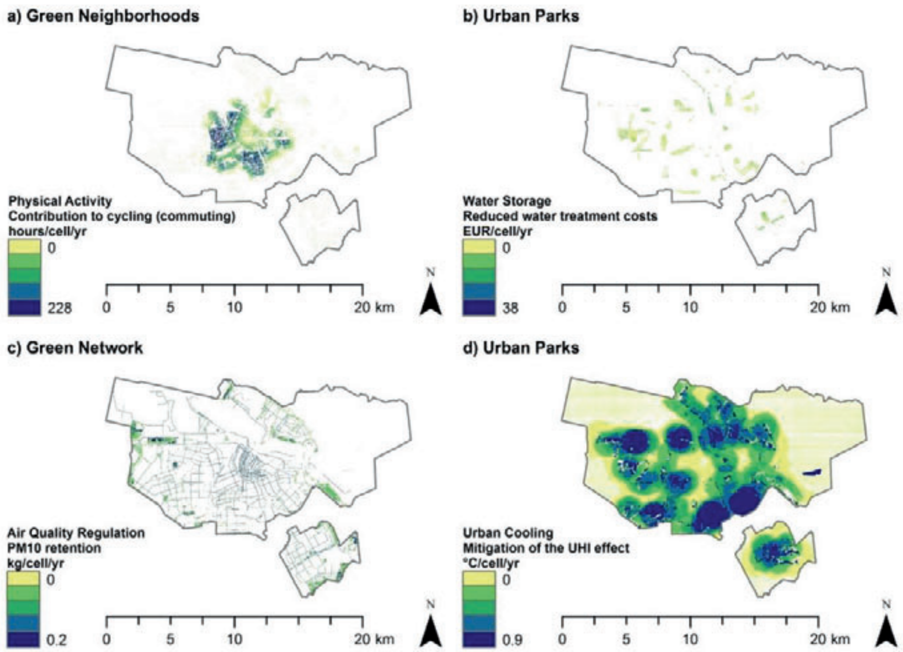

Figure 4.5: Changes in the performance of four ecosystem service supply and use for different scenarios in reference to the Business-As-Usual scenario (cell size = 10 x 10 m). For each map, legends show quantile values. All quantile thresholds values are presented in the Supplementary Material (Table A3 - 4, Appendix 3 – 3). Unshaded areas comprise areas where no value has been assigned. Additional maps displaying changes in the distribution of ecosystem service supply and use across scenarios, see Supplementary Material (Appendix 3 – 4).

4.3.3 Changes in ecosystem services: Relative values

Assessing ecosystem service delivery by use of various indicators expressed in various units is useful since (i) ecosystem service supply and use are intricately interconnected yet cannot always be expressed in identical units (Alam et al., 2016), and as (ii) it enables the communication of ecosystem service values to audiences with different backgrounds and preferences (Satz et al., 2013). Despite these advantages, incommensurability of ecosystem service indicators also restricts their comparability and potential aggregation. Comparability enables the assessment of trade-offs and synergies among ecosystem services, contributing to (i) instrumental decision-making based on information on ecosystem service gains and losses, and (ii) conceptual discussions that shape the way decision-makers and stakeholders think about ecosystem service policies (Wright et al., 2017). Aggregation is instrumental for estimating the total value of ecosystem service bundles and comparing them across time and space (Yang et al., 2019). However, it can also lead to double-counting and over- or underestimation of individual ecosystem service values, hampering the objectivity of results (Paulin et al., 2020b). Because of this, we refrain from aggregating ecosystem services in this study. Instead, commensurability is achieved by calculating the percentage change in value of each ecosystem service proxy indicator for each GI scenario in reference to the Business-As-Usual scenario. For comparability, results are visualised in a radar plot (Figure 4.6), a common approach for illustrating relative values of ecosystem service indicators within bundles and across scenarios (Demestihas et al., 2019).

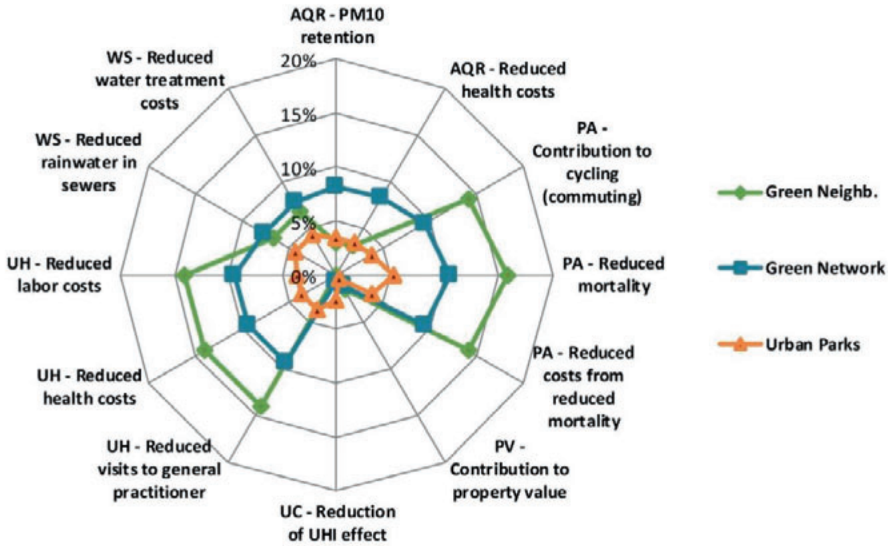

Figure 4.6: Total relative change (percentage increase) in ecosystem service values per GI scenario in reference to the Business-As-Usual scenario. AQR = Air Quality Regulation; PA = Physical Activity; PV = Property Value; UC = Urban Cooling; UH = Urban Health; WS = Water Storage. Detailed statistics on percentage changes per indicator and heterogeneity across scenarios, are presented in the Supplementary Material (Table A3 - 5, Appendix 3 – 3).

Evaluating relative changes in ecosystem service performance across scenarios enables the assessment of total changes in individual performance indicator values and total changes in ecosystem service bundles across scenarios. It also enables the assessment of heterogeneity within scenarios by providing information on overall changes (i.e., mean) against individual changes (i.e., SD) of performance indicator values. In a nutshell, five main conclusions can be drawn based on the assessment of relative values in this case example:

i) **Highest relative increase.** The highest relative increase in ecosystem service performance indicator values is seen for indicators modelled by use of the Physical Activity and Urban Health models, given the application of strategies in the Green Neighbourhoods scenario.

ii) **Lowest relative increase.** Ecosystem service performance indicators modelled by use of the Property Value model reveal the lowest relative increase given the implementation of GI strategies (0-2%).

iii) **Green Neighbourhoods.** Implementation of strategies in this scenario lead to the highest relative increase in individual indicator performance, with a 14-16% increase in six performance indicators. All performance indicators considered, this scenario reveals the highest general increase in proxy indicator performance and highest heterogeneity in changes (mean = 9%, SD = 6%).

iv) **Urban Parks.** Implementation of strategies in this scenario lead to a relatively low increase in individual indicator performance (0-5%). All performance indicators considered, this scenario reveals the lowest general increase in proxy indicator performance and lowest heterogeneity in changes (mean = 3.3%, SD = 1%).

v) **Green Network.** Implementation of strategies in this scenario lead to a moderate increase in individual indicator performance, with all but two performance indicators revealing an 8-10% increase. All performance indicators considered, this scenario reveals a moderate in proxy indicator performance and moderate heterogeneity in changes (mean = 8%, SD = 3%).

4.3.4 Urban planning

This study's application of the NC-Model provides information on how the implementation of GI strategies can lead to changes in the distribution, relative performance, and overall performance of ecosystem service supply and use. Assessment results are instrumental to support urban planning in the context of (i) communication and awareness raising; (ii) strategic planning and priority setting; and (iii) economic accounting and incentive design (Gómez-Baggethun & Barton, 2013; Haase et al., 2014). We expand on these points below.

i) **Communication and awareness raising**. This assessment provided quantitative and illustrative information on how changes in GI can affect ecosystem service delivery and hence the socioeconomic well-being of urban dwellers. The juxtaposition of high-resolution maps and quantitative values revealed that changes in the size, configuration, and typology of GI are key determinants to changes in ecosystem service delivery. Displaying relative changes in ecosystem service values across scenarios by use of radar plots enabled comparison of highly complex information in a clear and user-friendly manner. This kind of information is instrumental for urban planners who wish to better understand the mechanism by which GI contributes to the urban quality of life and to further disseminate this information to the public. Urban planners that participated in this study's final workshop (i.e., presentation of results) exhibited a higher interest in the mechanism by which GI supports ESP and socioeconomic well-being, than in the economic value that GI generates, providing a glimpse of their preferences and priorities in the context of urban planning. For some participants, the many ecosystem services that GI generates and the process leading to their delivery were virtually unknown prior to the workshop, emphasising the value of the approach for awareness-raising. By obtaining results in various formats (tables, maps, radar plots, indicators expressed in various units), urban planners were equipped with tools to further disseminate results for communication and decision-making purposes, in a way that speaks to various target groups (e.g., decision-makers, investors, local inhabitants).

ii) **Strategic planning and priority setting.** Assessment results can be adopted by urban planners to develop, rethink, and prioritise GI strategies in alignment with local objectives, based on scientifically-sound information. In Amsterdam, a number of policy initiatives (Amsterdam Municipality, 2010, 2015a, 2015b, 2017b, 2018, 2019a, 2019b) epitomise the desire to improve the quality of public spaces to enhance the quality of life of Amsterdam's rapidly growing population. To make this desire a reality, the Green Quality Impulse (Amsterdam Municipality, 2017a) will lay out the implementation plan for expanding and redesigning Amsterdam's GI to support its transition into a sustainable, climate-proof, and socially attractive city. This study's results revealed that expected changes in the distribution and performance of ecosystem services, resulting from changes in GI, are not homogeneous across space. This can be explained by the complex and diverse mechanisms that underpin the delivery of each ecosystem service. In better understanding the factors that influence ecosystem service delivery (e.g., distribution and composition of GI, population distribution), as well as the trade-offs that exist among ecosystem services, urban planners in the Municipality of Amsterdam can rethink and prioritise GI strategies to target objectives from the Green Quality Impulse (e.g., emphasising the reduction of the UHI effect and enhancement of water storage for climate resilience), as well as areas where societal challenges (e.g., heat stress, flood risk, low income) overlap.

iii) **Economic accounting and incentive design.** The lack of awareness of the many benefits that GI can generate in urban areas often results in conversion of urban nature into built infrastructure, resulting in ecosystem service loss (Gómez-Baggethun & Barton, 2013). Building a case for investments in GI requires transparency regarding capital and operational costs, as well as socioeconomic benefits, that investments in GI entail (Schäffler & Swilling, 2013; Maes et al., 2015b). In doing so, GI can be viewed from a more positive light, not solely as a source of cost but also as an investment opportunity (Ernst, 2020). In this study, the costs associated with investments in GI were not considered, as the Green Quality Impulse is in its development phase, so this information is not readily-available. A complete assessment of the expected efficacy of each strategy for meeting local objectives would require juxtaposition of the (capital and operational) costs of implementation, with associated improvements in the performance of ecosystem services. For instance, implementation of the Green Neighbourhoods strategy reveals the highest increase in value for most indicators. This was a somewhat surprising finding for urban planners, given the lower expansion in vegetated cover relative to other strategies, and as the strategy envisions no transformation from herbaceous to woody vegetation. The expected improvement in ecosystem service delivery occurs since GI is introduced in densely populated areas with predominant built-up cover, which often host a high abundance of beneficiaries and ecological pressures that require mitigation. Despite these benefits, introducing GI in densely populated areas is often costly due to previous removal or degradation of green space in ways that are difficult to reverse (Kruize et al., 2019; Vallecillo et al., 2018). In Amsterdam, densely populated areas include post-war residential neighbourhoods, comprising relatively small housing units with no front yard and small backyards. Introducing GI in sealed areas would entail high costs due to the presence of infrastructure (e.g., sewers, gas, water, electricity, internet cables). This accentuates (i) the need to consider the costs of particular GI strategies when evaluating their accrued benefits (Kruize et al., 2019), and (ii) the significance of considering GI as a fundamental aspect of urban development and not just as an end-of-the-pipe solution.

4.3.5 Limitations

The assessment approach presented in this paper can be useful to inform urban planners on the complex nature by which GI generates value for urban dwellers. However, it is important that all limitations associated with such an assessment approach are elucidated to receptors of results for their unbiased interpretation and appropriate consideration within urban planning. First, models adopted for quantifying and mapping ecosystem services provide a simplification of real systems, as it is not objective (nor possible) to consider all aspects that characterise inherently complex coupled socioecological systems (Paulin et al., 2020b). Hence, only the most relevant factors affecting ecosystem service performance, for which instrumental knowledge and data is available, are considered. For instance, trees in street canyons can lead to an increase atmospheric PM_{10} concentrations (Janhäll, 2015). This factor is currently not considered in the Air Quality Regulation model due to limitations in knowledge and data. In turn, this may lead to an overestimation of the contribution by trees to PM_{10} retention in the Green Network scenario, where trees are added along streets. Second, a diverse yet limited selection of ecosystem service performance indicators is assessed. This results from limitations in data and empirical research required to assess a more comprehensive, locally-relevant suite of ecosystem services. This may lead to an overestimation of assessed ecosystem services and an underestimation of omitted ecosystem services (Paulin et al., 2020b). Improving model objectivity would require the development of periodically-repeated empirical research capturing relationships between ecological and socioeconomic parameters relevant at the urban scale in the Netherlands. The continuous assessment of empirical relationships relevant in the Netherlands could provide valuable input for continuously developing and calibrating the current suite of models in the NC-Model.

4.4 Conclusions

Through its incorporation of best-available local knowledge and data (e.g., Dutch spatial datasets and empirically-established relationships linking ecological and socioeconomic parameters), the NC-Model enables the spatial quantification of Dutch urban ecosystem services at high spatial and thematic detail. In this study, the model was successfully implemented to spatially quantify ecosystem services in the Municipality of Amsterdam at a high resolution, considering locally-relevant environmental and socioeconomic characteristics (e.g., PM_{10} concentrations that require mitigation, GI typologies whose local value reflects in property values). Implementation of the NC-Model across scenarios that capture GI strategies from Amsterdam's Green Quality Impulse, provided detailed insights on how the implementation of each strategy may influence ecosystem service delivery. Changes in GI from the application of each strategy are expected to heterogeneously affect the total performance and distribution of different ecosystem services, accentuating the complexity of the mechanism that underpins ESP and accrued socioeconomic benefits. In general, the distribution and composition of GI, as well as population distribution, were identified as key factors affecting ecosystem service delivery. Changes in ecosystem service supply were primarily affected by changes in the distribution and composition of GI, while changes in ecosystem service use were significant where population densification is prominent. In capturing ecosystem services and (ecological and socioeconomic) factors that influence their performance in fine detail, such an approach can foster a better understanding among urban planners on the mechanism by which GI generates value for urban dwellers. This is instrumental for urban planners who wish to develop strategies that optimise ecosystem service delivery in alignment with local objectives, and to further communicate this information to decision-makers, investors, and local inhabitants in simple but scientifically-sound manner.

The availability of input data and knowledge required to model urban ecosystem services in fine detail, capturing relevant local characteristics, varies significantly across different geographical locations. Where local spatial data is absent or poor in quality, incorporation of existing datasets, produced at the regional and global scales, is endorsed for modelling ESP. In the absence of local empirically-established relationships between ecological parameters, incorporation of empirically-established relationships, obtained in regions with similar environmental characteristics, is endorsed. In the absence of local empirically-established relationships between ecological and socioeconomic parameters, consideration of empirically-established relationships, obtained in regions with similar socioeconomic characteristics, is endorsed. Even though data and benefit transfer may reduce the level of accuracy and hence the objectivity of results, it can provide an opportunity for endowing urban planners with valuable information on the value of GI and its contribution to human well-being. Given the adequate communication of uncertainties, local approaches for assessing urban ecosystem services can endorse the optimal allocation of GI to mitigate the pressures of urbanisation and to promote the fair distribution of ecosystem services.

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Cooperatie Collectief Hoeksche Waard (CCHW), Land en Tuinbouw Organisatie Nederland (LTO Nederland), Stichting Rietgors, Staatsbosbeheer, and Natuurmonumenten.

5

CHAPTER 5

Integration of local knowledge and data for spatially quantifying ecosystem services in the Hoeksche Waard, the Netherlands

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Abstract

Enhancing consideration of the ecosystem services concept within decision-making calls for integration of local knowledge and data within assessments to capture context-specific spatial and thematic detail. This paper demonstrates how local knowledge and (biophysical, sociocultural, economic) data can be integrated within ecosystem service assessments by use of well-established spatial quantification methods. We hypothesise that ecosystem services can be spatially quantified at high spatial and thematic detail by integration of local knowledge and data within models, contributing to identification of key factors that influence their delivery. We demonstrate this by making use of local knowledge and data to assess ecosystem services in the Hoeksche Waard, a Dutch municipality characterised by historically-rich cultural landscapes and predominant agriculture. Ecosystem services assessed include crop production, air quality regulation, human health, pest control, soil biodiversity, sociocultural values, property value. Quantification methods were selected based on their suitability for modelling ecosystem services given particular resource endowments (i.e., time, data, knowledge). Methods implemented include look-up tables, causal relationships, expert elicitation, primary data extrapolation, and regression models. Maps displaying the distribution of ecosystem services at high spatial resolution (10 x 10 m) enabled identification of factors that influence their delivery, including the distribution and typology of natural elements, ecological pressures that require mitigation, and the distribution of inhabitants that act as ecosystem service beneficiaries. For instance, the distribution and typology of field margins plays a key role in the suppression of pests (aphids) by natural enemies (hoverflies, carabids, coccinellids). Air quality regulation (i.e., particulate matter retention) is highest in the northeast sector of the municipality given higher concentrations of particulate matter that require mitigation due to the area’s proximity to the cities of Rotterdam and Dordrecht. Contributions by natural elements to human health and property value are prominent in villages, where most inhabitants and thus built-up property are concentrated.

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5.1 Introduction

Ecosystem services are the direct and indirect benefits that ecosystems provide humans with (Costanza et al., 2017; MA, 2005). According to the Common International Classification of Ecosystem Services (CICES; Haines-Young & Potschin, 2018), ecosystem services can be classified as provisioning, regulation and maintenance (also regulating; MA, 2005; TEEB, 2010), and cultural services. Provisioning services are the material contributions that natural capital endows humans with, such as crop and groundwater production for human consumption and animal feed (MA, 2005). Regulating services are ecological processes that directly or indirectly contribute to human well-being (MA, 2005). Examples include the retention of atmospheric concentrations of harmful pollutants by vegetation and water (Remme et al., 2018; Janhäll, 2015), the potential contribution of biological pest control to final crop yield and ecosystem resilience (Tschumi et al., 2016), and climate regulation through carbon sequestration by vegetation and soils (Díaz et al., 2009; Breure et al., 2018). Cultural services are non-material benefits that ecosystems provide for humans, such as the spiritual, recreational, or intrinsic value people assign to natural elements and landscapes (MA, 2005).

The ecosystem services concept provides an opportunity for incorporating scientific knowledge within spatial management (Haase et al., 2014; Walsh et al., 2015). It can serve as a language through which science communicates to society, shedding light on the various mechanisms by which ecosystems generate value for humans (Potschin & Haines-Young, 2016; Breure et al., 2012). Recognising the concept's potential, recent years have seen an upsurge in the number of initiatives calling for integration of ecosystem services within environmental decision-making (Díaz et al., 2015b; EC, 2011; CBD, 2010). To meet their common objectives, a number of standardised modelling approaches have been developed (e.g., InVEST, ARIES, ESTIMAP; Tallis and Polaski, 2011; Villa et al., 2014; Zulian et al., 2014; Maes et al., 2016). Their aim is to contribute to the standardisation and harmonisation of the ecosystem service spatial quantification process, thereby facilitating the comparability of results across space and time. Despite these initiatives, consideration of the ecosystem services concept within decision-making remains limited (Haase et al., 2014; Walsh et al., 2015). This partly results from a limited integration of spatial and thematic detail central to particular socioecological systems within assessments (Derkzen et al., 2015; Martínez-López et al., 2019), as well as challenges in the communication of complex models and their outputs to non-scientist end users (Villa et al., 2014). This limits confidence in assessment output and thereby its uptake to support spatial planning (Lilburne & Tarantola, 2009; Schuwirth et al., 2019).

To address this challenge, enhanced integration of local knowledge and data within ecosystem service models is needed (Martínez-López et al., 2019; Orchard-Webb et al., 2016). Consideration of local knowledge and data (where available) is instrumental for capturing site-specific sociocultural and ecological factors that influence the access to and use of ecosystem services by beneficiaries (Díaz et al., 2018; Paulin et al., 2020a). It also contributes to the democratisation and legitimisation of the assessment process, supporting the integration of final results within decision-making (Orchard-Webb et al., 2016). Despite these benefits, standardised ecosystem service assessment models often incorporate non-site specific knowledge and data as input (Petz et al., 2017). This leads to the underrepresentation of fundamental characteristics of particular socioecological systems at the local scale (Martínez-López et al., 2019). In general, ecosystem services research is often performed within the field of ecology (Luederitz et al., 2015), carried out separately from economic studies and decision-science (Rasmussen et al., 2016; Haase et al., 2014). In addition, local knowledge is often excluded from assessments, oversimplifying the central role that culture plays in defining human-nature interactions (Díaz et al., 2018).

Incorporation of local knowledge and data within ecosystem service assessments often requires deviation from the use of standard ecosystem service models that are unsuited for capturing the spatial and thematic detail that define real-world management situations (Villa et al., 2014; Martínez-López et al., 2019). In these instances, the choice of suitable methods for quantifying and mapping local ecosystem services is complicated by the abundance of available methods (Seppelt et al., 2011). Methods refer to the way in which data sources are used to quantify and map ecosystem services (Martínez-Harms &

Balvanera, 2012). The suitability of these methods may vary depending on various factors. These include (but are not limited to) the type of ecosystem service under consideration, the expertise of assessors, the type of data available, and time constraints (Eppink et al., 2012; Schröter et al., 2015). The aim of this study is to demonstrate how local knowledge and data (i.e., biophysical, sociocultural, economic data) can be included within ecosystem service assessments by integration of well-established spatial quantification methods.

Conceptualisations of ecosystem service spatial quantification methods vary throughout the literature. For clarity and consistency, a number of studies reviewed publications mapping ecosystem services in order to identify and conceptualise commonly implemented spatial quantification methods (Martínez-Harms & Balvanera, 2012; Grêt-Regamey et al., 2017; Lavorel et al., 2017). We demonstrate how well-established methods can be integrated to spatially quantify seven ecosystem services in the Hoeksche Waard, incorporating local knowledge and data (Section 5.3). The Hoeksche Waard is a Dutch municipality characterised by historically-rich cultural landscapes and predominant agriculture. Covering 40% of Earth's terrestrial ecosystems, agricultural landscapes comprise a range of ecosystem services and stakeholders, making them some of the most interesting to analyse (Foley et al., 2011; Montoya et al., 2020). The Hoeksche Waard is a particularly interesting study site, given notable interest by local stakeholders to collaborate to meet their common objectives. Spatial quantification methods that were implemented in this study were conceptualised based on the seminal work of Martínez-Harms & Balvanera (2012) and Schröter et al. (2015), who conducted extensive literature reviews to identify spatial quantification methods that are commonly implemented to map ecosystem services. Implemented methods include causal relationships, expert elicitation, primary data extrapolation, regression models, and look-up tables. We elaborate on these methods and how they can be integrated to spatially quantify ecosystem services given different resource endowments (i.e., knowledge, data, time; Section 5.2). A key feature of this study's methodology is that all models incorporate local knowledge and data. We hypothesise that ecosystem services can be spatially quantified at high spatial and thematic detail by integration of local knowledge and data within models, contributing to identification of key factors that influence their delivery.

5.2 Materials and methods

5.2.1 Study area

The Hoeksche Waard is a municipality with a surface area of approximately 324 km^2, distributed across croplands (49%), grasslands (13%), nature areas (13%), small and large water bodies (17%), and built-up and paved areas (9%; Figure 5.1). It has a population of 86,656 inhabitants (https://opendata.cbs.nl/) and its economy is primarily agricultural, focusing on the production of sugar beet and potato, in rotation with a variety of other crops. The spatial configuration of the area's natural elements is attributed to a long history of reclamation and cultivation. The creation of the dike structure as we know it started in 1538 and the island reached its current extent in 1653. Today, the landscape contains different reclamation structures reflecting its history, including vast polders, large water bodies, and protected natural areas. Over the years, the desire to protect the landscape has given rise to a collaborative network between farmers, local and regional governments, and NGOs (http://hwodka.nl/; www.rietgorsinfo.nl/home/; https://www.cchw.eu/). In 2004, the project Functional Agrobiodiversity Hoeksche Waard (FAB-HW; Van Alebeek & Cleveling, 2005) came to place, aiming to increase biodiversity to enhance biological pest control, thereby reducing the application of chemical pesticides (Van Rijn et al., 2008). This initiative has led to the large-scale implementation (>460 km) of field margins, and grassy and flower-rich strips of land, which surround parcels that provide habitat for plants, birds, and beneficial insects (e.g., pollinators, insects that prey upon pest organisms; www.hwl.nl/; Van Rijn et al., 2008; Wratten et al., 2012). In 2005, given its varied and valued cultural and ecological background, the Hoeksche Waard was assigned as one of 20 Dutch National Landscapes. This status is granted to areas characterised by unique cultural, historical, and natural elements (https://nationalelandschappen.nl/).

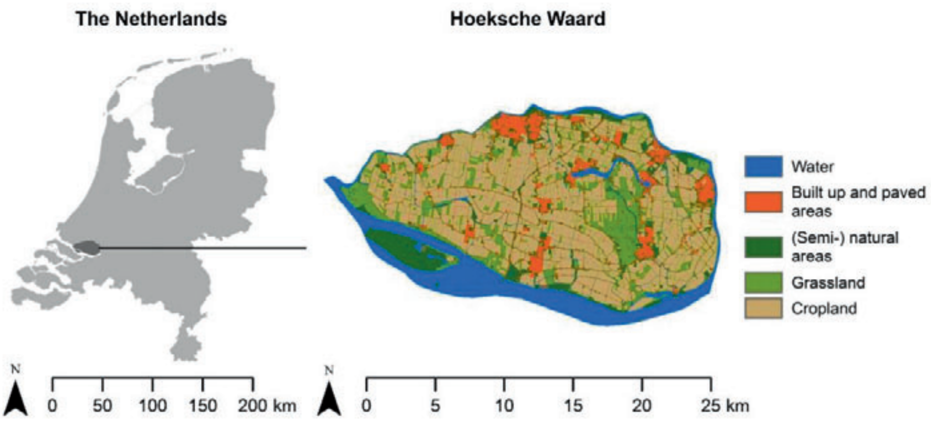

Figure 5.1: Dominant types of land cover in the Hoeksche Waard. Based on the crop parcels map (BRP; https://www.rvo.nl/) and the land use map (Ecosystem Unit Map, EUM; Van Leeuwen et al., 2017).

5.2.2 Scoping

In the scoping phase of this assessment, stakeholders and ecosystem services relevant to the Hoeksche Waard were identified. The scoping consisted of two stages. During the first stage, a literature review was performed with two main purposes. The first purpose was to obtain information on ecosystem services that are relevant in the area given current land use patterns. The second purpose was to identify stakeholder groups that influence (manage) or benefit from the provision of ecosystem services in the area. During the second stage, semi-structured interviews were carried out with representatives from each identified stakeholder group with two main purposes. The first purpose was to obtain information on relevant ecosystem services. The second purpose was to identify stakeholder groups that may have been overlooked by only assessing the literature. Interviewed stakeholder representatives included farmers, members of local governments, nature organisations, the local water board, and environmental organisations. Environmental organisations are entities that focus on the achievement of objectives shared by different stakeholder groups, in this case related to nature management. This stage was key for understanding stakeholder objectives and for identifying site-specific ecosystem services. The Supplementary Material (Appendix 4 – 1) lists identified stakeholder objectives and provides details on interviewed parties.

5.2.3 Modelling approach

Based on the literature review and semi-structured interviews conducted during the scoping phase of the study, seven ecosystem services were identified for their assessment (Table 5.1). The names ascribed to ecosystem services comprise user-friendly terms that are instrumental when involving stakeholders and decision-makers perhaps less knowledgeable on ecosystem services terminology and concepts (Paulin et al., 2020a). Ecosystem services were operationalised into supply and use indicators, capturing the biophysical and the socioeconomic aspects of ecosystem service delivery (Figure 5.2). Supply captures the distribution and performance of ecosystem functions, or the ecological structures and processes that contribute to human well-being (Burkard et al., 2012; Syrbe & Walz, 2012). Hence, it is best represented by use of biophysical indicators (Martín-López et al., 2014; Castro et al., 2014; Vigl et al., 2017). Use captures the distribution and performance of the realised socioeconomic benefits that

Table 5.1: Descriptions of ecosystem service supply and use indicators considered within seven ecosystem service models

Ecosystem service model	Supply/ use	Indicator	Description	Unit
Provisioning services				
Crop Production	Supply	Volume of harvested crops	Physical volume of harvested crops	kg/yr
	Use	Net output of harvested crops	Total revenue of harvested crops	€/yr
	Use	Value added of harvested crops	Net output minus intermediate inputs (costs of labour and capital)	€/yr
Regulation and maintenance services				
Air Quality Regulation	Supply	PM_{10} retention	Reduction in atmospheric PM_{10} concentrations by vegetation and water	kg/yr
	Use	Reduced health costs	Reduction in health costs from avoided PM_{10} related mortalities	€/yr
Human Health	Use	Reduced health costs	Reduction in health costs linked to the contribution by green space to mitigating the incidence of seven disease categories (i.e., cardiovascular diseases, musculoskeletal diseases, mental diseases, respiratory diseases, neurological diseases, digestive diseases, and a miscellaneous category)	€/yr
	Use	Reduced labour costs	Reduction in costs of absenteeism, reduced labour productivity, and job losses, linked to the contribution of green space to improved health conditions	€/yr
	Use	Reduced visits to general practitioners	Avoided visits to general practitioners linked to the contribution of green space to improved health conditions	visits/ yr
Pest Control	Supply	Yearly effective pest control	Effectiveness level of suppression of aphids by natural enemy populations (i.e., hoverflies, coccinellids, carabids) inhabiting natural elements	score 0-1.5
Soil Biodiversity	Use	Performance of soil biodiversity	State of populations of species that support soil quality, based on soil, environmental, and management characteristics	score 1-3
Cultural services				
Sociocultural Values	Supply	Natural landscape elements	Croplands, large water bodies, grassland and shrubs, polder structures, ditches, creeks, trees, hedges and wood walls	area (ha)
	Use	Cultural identity and heritage	The perceived contribution of landscape natural elements to the cultural identity and heritage of the area	score 0-5

	Use	Educational and scientific knowledge	The perceived contribution of landscape natural elements to educational and scientific knowledge	score 0-5
	Use	Habitability	The perceived contribution of landscape natural elements to the habitability of the area	score 0-5
	Use	Intrinsic values	The perceived contribution of landscape natural elements to intrinsic values	score 0-5
	Use	Landscape aesthetics	The perceived contribution of landscape natural elements to landscape aesthetics	score 0-5
	Use	Recreational potential	The perceived contribution of landscape natural elements to an area's recreational potential	score 0-5
Property Value	Use	Contribution to property value	Contribution by vegetation and open water to property prices due to the value people assign to these elements	€

ecosystem functions underpin (Wolff et al., 2015; Syrbe & Walz, 2012). Hence, it can be best represented by use of sociocultural and economic indicators (Martín-López et al., 2014; Castro et al., 2014; Vigl et al., 2017). The Supplementary Material provides an overview of the operationalisation of all supply and use indicators modelled according to CICES (version 5.1; Appendix 4 – 2).

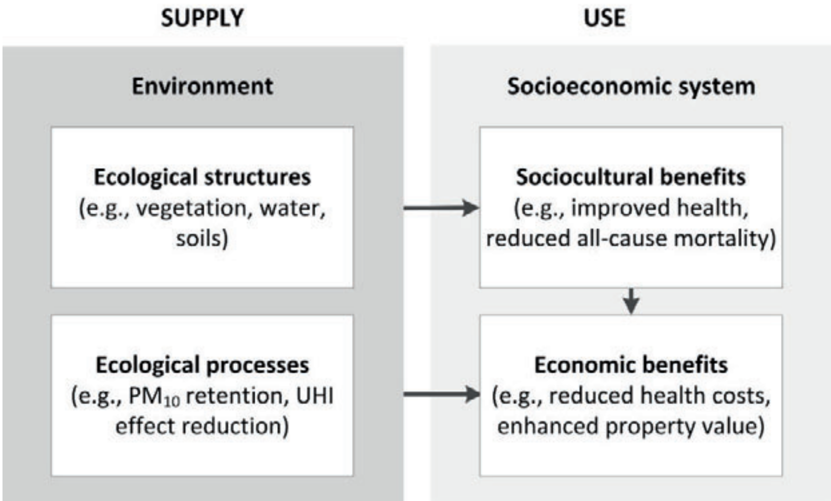

Figure 5.2: Visual representation of relationship between ecosystem service supply and use (source: Paulin et al., 2020a)

Ecosystem services were modelled by use of social-ecological assessment models. In social-ecological assessment models, the relationship amongst measurable biophysical (e.g., land-cover, remote- sensed data, spatially extrapolated field observations) and socioeconomic variables (e.g., population, survey data, statistical data) is modelled to spatially quantify ecosystem service proxy indicators (Martinez-Harms & Balvanera, 2012). All ecosystem service models integrate local knowledge and data, which enabled the development of ecosystem service maps at a high spatial resolution (10 x 10 m) and the identification of key factors that influence their delivery. To couple local knowledge and data, various spatial quantification methods are integrated within models. Spatial quantification methods applied in this study are presented and described in Table 5.2. These methods are conceptualised based on the work of Martinez-Harms & Balvanera (2012) and Schröter et al. (2015).

The selection of methods that are implemented within each model can be based on their suitability given particular resource endowments (i.e., knowledge, data, time). Models based on 'causal relationships' are suitable in situations where the use of readily-available knowledge and data is sufficient to adequately capture local ecosystem service delivery. Where literature-based and data-based causal relationships are insufficient to adequately capture ecosystem service supply and use, 'regression models' can be implemented. In doing so, it may be possible to identify causal relationships amongst biophysical and socioeconomic variables. However, this method is considerably more time-consuming compared to the use of readily-available causal relationships. 'Expert elicitation' provides a practical solution where causal relationships are insufficient to adequately capture local supply and use, and where our mechanistic understanding of the system is insufficient (Jacobs & Burkhard, 2017). It is also key for understanding local stakeholder preferences (e.g., motivations, perceptions, knowledge, principles, virtues) that determine which ecosystem services are valued in an area (Santos-Martín et al., 2017). Information based on causal relationships can be directly linked to spatial variable categories to capture ecosystem service delivery through their incorporation within 'aggregated statistics look-up tables' (LUT). Information that has been obtained through expert elicitation can be linked to spatial classification typologies through their incorporation within ‘qualitative LUT’. ‘Multiple layer LUT’ are useful for linking various LUT, which may include a combination of binary, qualitative, and aggregated statistics LUT. In this study, ‘primary data extrapolation’ took place through incorporation of primary data within LUT. In addition, the models Air Quality Regulation, Human Health, and Property Value incorporate results from regression models that have been published in peer reviewed literature (Paulin et al., 2020b).

Ecosystem services were modelled by combining spatial data and reference values that capture relationships between variables within algorithms. Reference values can be obtained from various sources, such as statistical databases, regression models, expert elicitation, and published empirical studies. Algorithms were written in Python programming language (https://www.python.org/) using the PCRaster software to perform spatial calculations (http://pcraster.geo.uu.nl/). All codes and model outputs are available from the authors upon request. A number of ecosystem services were modelled by use of local models from the Natural Capital Model (NC-Model), a spatially-explicit set of models for quantifying and mapping ecosystem services within the Netherlands at various scales (Paulin et al., 2020b; Remme et al., 2018). Models from the NC-Model implemented in this study include the Air Quality Regulation, Human Health, Pest Control, and Property Value models. Spatial datasets that have been used as input throughout the modelling process are presented in Table 5.3. Selected input datasets include their most recent available version, or versions that align well with the general dates of all datasets. The Supplementary Material provides detailed descriptions of the spatial data that were used as input, as well as stepwise procedures for the direct replication of all models (Appendices 4 – 3 and 4 – 4). For all output maps, descriptive statistics (i.e., minimum, maximum, mean, standard deviation) and Pearson correlation coefficients were calculated by use of the ArcGIS 10.6.1 geospatial processing program (https://www.arcgis.com/; Supplementary Material, Appendix 4 – 5).

Table 5.2: Ecosystem service spatial quantification methods implemented in this study. In this context, an expert is any individual who may provide key information necessary for understanding and modelling the system under assessment (e.g., scientist, technician, stakeholder; Jacobs & Burkhard, 2017).

Method	Definition	Source
Causal relationships	Incorporation of quantitative knowledge (e.g., from statistical databases, regression models) about the relationship between biophysical and socioeconomic variables within ecosystem service models	Martinez-Harms & Balvanera (2012); Schröter et al. (2015)
Expert elicitation	Mobilisation of experts to assign values to biophysical and socioeconomic variable categories (e.g., through ranking, rating, assigning, weights, photo elicitation) based on their knowledge	Martinez-Harms & Balvanera (2012)
Primary data extrapolation	Extrapolation of biophysical or socioeconomic values (e.g., land-cover class, pH value, social value) from primary data (e.g., field data, survey data) to the area under assessment	Martinez-Harms & Balvanera (2012)
Regression models	Model the relationship among biophysical and socioeconomic variables by means of statistical regressions (e.g., relationship between proxy indicators representing green space and proxy indicators representing human health)	Martinez-Harms & Balvanera (2012); Schröter et al. (2015)
Look-up table(LUT)	The use of a constant value to link measurable biophysical and socioeconomic variables	Martinez-Harms & Balvanera (2012)
Binary LUT	LUT that define the presence or the absence of spatial variable category (e.g., trees, bushes, shrubs)	Schröter et al. (2015)
Qualitative LUT	LUT that assign weights, ranks, or values to spatial variable categories (e.g., aesthetic value of particular natural elements) based on expert knowledge	Schröter et al. (2015)
Aggregated statistics LUT	LUT that assign values to spatial variable categories (e.g., PM_{10} retention capacity by vegetation typologies) based on reported statistics or quantitative research findings (e.g., regression models))	Schröter et al. (2015)
Multiple layer LUT	LUT that assign values to biophysical and socioeconomic variables categories based on cross tabulation of different LUT	Schröter et al. (2015)

Table 5.3: Spatial data used as input for modelling seven ecosystem services in the Hoeksche Waard

Original dataset name	Description	Resolution	Year
Actueel Hoogtebestand Nederland (AHN2)	Elevation map of the Netherlands	5 x 5 m	2007-2012
Akkerranden Hoeksche Waard 2017	Field margins of the Hoeksche Waard	10 x 10 m	2017
Basisregistratie Adressen en Gebouwen (BAG)	Basic registry of addresses and buildings	2.5 x 2.5 m	2016
Basisregistratie Gewaspercelen (BRP)	Agricultural crop parcels of the Netherlands	25 x 25 m	2018
Bodem Biologische indicator (BoBi)	Soil (biological, physical, and chemical) characteristics	100 x 100 m	2006-2012
Bodemfysische Eenhedenkaart (BOFEK2012)	Soil biophysical units	50 x 50 m	2012
Ecosystem Unit Map (EUM)	Land use map of the Netherlands	10 x 10 m	2013
Fijnstof 2017 (PM10)	Concentration of particulate matter up to 10 micrograms (PM_{10})	25 x 25 m	2017
Population	Distribution of inhabitants in the Netherlands	10 x 10 m	2017
Top10NL	Topographic land use map of the Netherlands	5 x 5 m	2017
Vegetation	Distribution of vegetation	10 x 10 m	2017
Wijk- en buurtkaart	Key statistical figures of neighbourhoods, districts, and municipalities	160 x 160 m	2018
WOZ-Waarde	Real estate value	10 x 10 m	2016

As all codes, data, and steps required for the implementation of models are publicly available, their direct replication principally requires a thorough understanding of spatial modelling and coding techniques. Customisation of models for their application at different sites is possible provided (i) that spatial data used as input for models is available at the location under assessment, and (ii) that reference values (e.g., obtained from previous studies or through expert elicitation) are tailored to values that are relevant at the particular ecological and socioeconomic context. In particular, implementation of the Sociocultural Values model at a different site would require executing surveys with stakeholders, which is a time demanding process.

5.2.4 Ecosystem services

5.2.4.1 Crop Production

The Crop Production model captures the yearly volume of harvested crops per spatial unit, as well as its associated monetary value. Data on average harvested volumes of different crops per spatial unit (kg/ha), average output per unit of volume (€/kg), and average value added per unit of volume (€/kg), were obtained from the central databank of the Netherlands (https://opendata.cbs.nl/). To model ecosystem service supply and use, spatial data providing information on the distribution of different crop types (agricultural crop parcels map) was reclassified by use of aggregated statistics LUT. LUT link different crop types to their average production volumes and their equivalent monetary units (i.e., net output and value added per spatial unit). Figure 5.3 presents a schematic diagram displaying how input data was modelled by use of two spatial quantification methods to quantify and map ecosystem service supply and use.

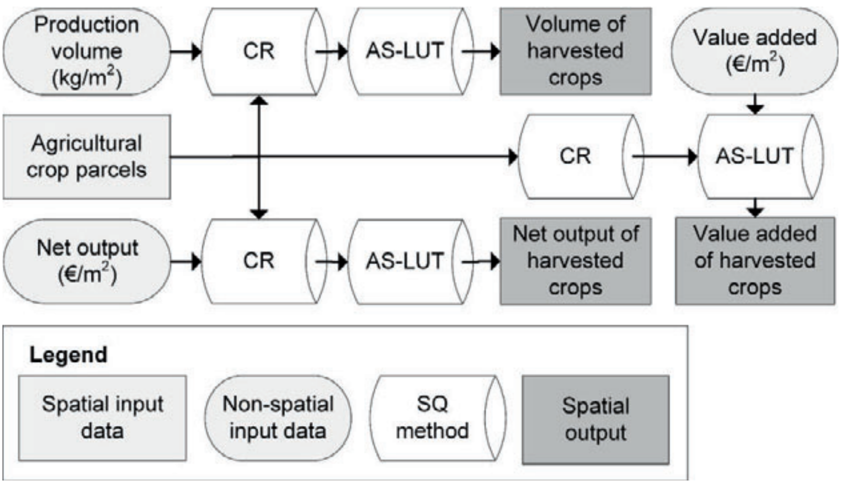

Figure 5.3: Schematic diagram displaying the modelling procedure for the ecosystem service 'Crop Production'. CR = causal relationships; AS-LUT = aggregated statistics look-up table; SQ method = spatial quantification method.

5.2.4.2 Air Quality Regulation

Particulate matter (e.g., from agriculture, industry, cars) is one of the most harmful components of air pollution for human health. It is associated with respiratory and cardiovascular diseases, as well as mortality (Derkzen et al., 2015; Santibañez et al., 2013). Due to the roughness of their surface, different vegetation types contribute to the retention of particulate matter (Remme et al., 2018; Janhäll, 2015). In this paper, the Natural Capital Model (NC-Model; Paulin et al., 2020b) was implemented to model the contribution by different vegetation types to reductions in atmospheric PM_{10} (i.e., particulate matter with a diameter of up to 10 μm) concentrations, as well as associated contributions to human health. Ecosystem service supply displays actual PM_{10} retention by vegetation. It is modelled by reclassifying spatial data displaying the distribution of natural elements (land use, vegetation maps) by use of aggregated statistics LUT. LUT link different vegetation typologies with their capacity for PM_{10} retention. Subsequently, an overlay is performed with spatial data displaying the actual distribution of atmospheric PM_{10} (PM_{10} concentration map). Ecosystem service use links PM_{10} reductions to associated reductions in health costs from avoided PM_{10} related mortalities. Ecosystem service use is modelled by combining the output layer displaying distribution of PM_{10} retention with a layer displaying the distribution of inhabitants (population map), as well as with causal reference values obtained from the literature. Reference values integrate regression model results linking PM_{10} concentrations to mortality with national statistics to calculate associated reductions in health costs (CE-Delft, 2017).

5.2.4.3 Human Health

Through its mitigation of environmental pressures (e.g., noise pollution, air pollution, and temperature extremes), green space (i.e., vegetation abundance) contributes to improvements in overall human health and reductions in all-cause mortality (Staatsen et al., 2017; Hartig et al., 2014; Kondo et al., 2018). To model the contributions of green space to human health in the Hoeksche Waard, the NC-Model (Paulin et al., 2020b) was implemented. The model captures the relationship between vegetation abundance surrounding households and its contribution to the health of household inhabitants. Contributions to human health reflect in reduced visits to the general practitioners, reduced health costs (i.e., from reductions in the incidence of nine disease groups), and reduced labour costs (i.e., from reduced absenteeism, increased labour productivity, and avoided job losses). It does so by combining spatial information on the distribution of natural elements (vegetation map) and the distribution of inhabitants (population map), with causal reference values obtained from the literature (Remme et al., 2018). Reference values include regression model results that link the percentage vegetated cover surrounding households (1 km buffer) to reductions in individual health costs, visits to general practitioners, absenteeism, and job losses, as well as increased labour productivity (KPMG, 2012; Maas, 2009). Regression model results on absenteeism, job losses, and labour productivity are combined with national statistics to calculate reduced labour costs associated with green space availability (KPMG, 2012).

5.2.4.4 Pest Control

As the negative side effects of widespread pesticide use becomes increasingly apparent (e.g., increased resistance of pests to pesticide treatment, suppression of non-target wildlife and beneficial insects, surface water contamination, deteriorated human health; Köhler & Triebskorn, 2013; Gooijer et al., 2019), policies are emerging calling for drastic reductions in pesticide use in order to protect humans and the environment (EZ, 2013b; EP, 2009). This can be achieved by implementing agricultural measures that protect and enhance populations of pest suppressing organisms (e.g., creation of field margins; Van Rijn et al., 2008). Pest suppression by natural enemy populations (i.e., insects that suppress pest populations) was modelled by use of the NC-Model (Paulin et al., 2020b; Remme et al., 2018). The main pest population considered in this paper consists of aphids. Natural enemy populations considered include hoverflies, coccinellids, and carabids. As field margins constitute a key habitat for natural enemy populations, the model was customised to include a local spatial dataset capturing the distribution of field margins in the Hoeksche Waard (field margins map). To model ecosystem service supply, spatial data displaying the distribution of natural elements (crop parcels, field margins, vegetation maps) is reclassified by use of binary LUT. LUT are used to determine which vegetation types may act as potential habitat for individual pest and natural enemy insect populations. The distribution of insect populations is not only determined by the distribution of their habitat, but also by insect mobility. Based on the distribution of pest and natural enemy populations, a score determined by experts (i.e., scientists) is assigned to each cell, capturing the effectiveness of pest control by biological agents (0 = no effectiveness, 1.5 = high effectiveness). Within each spatial unit, the effectiveness of pest control is determined by potential interactions between predator and prey, as well as potential intraguild interactions (complementarity or predation) amongst biological control agents (Hindayana et al., 2001; Alhmedi et al., 2010). Figure 5.4 presents a schematic diagram displaying how input data was modelled by use of various methods to spatially quantify ecosystem service supply.

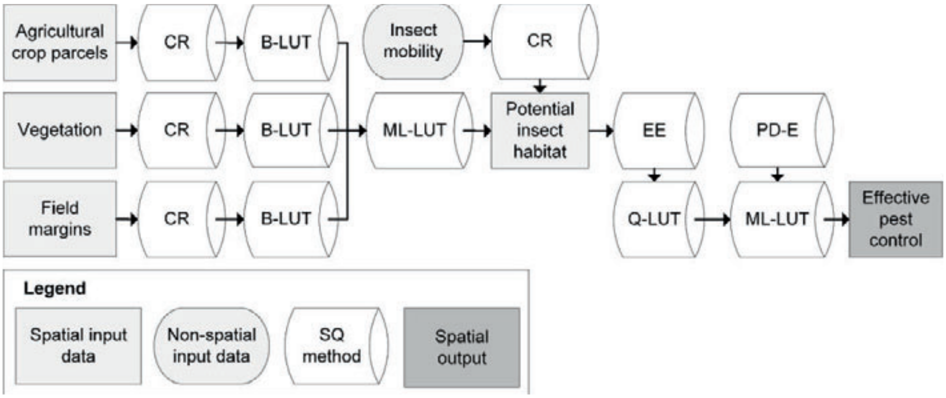

Figure 5.4: Schematic diagram displaying the modelling procedure for the ecosystem service 'Pest Control'. CR = causal relationships; B-LUT = binary look-up table; ML-LUT = multiple layer look-up table; AS-LUT = aggregated statistics look-up table; Q-LUT = qualitative look-up table; EE = expert elicitation; PD-E = primary data extrapolation; SQ method = spatial quantification method.

5.2.4.5 Soil Biodiversity

Soil biodiversity underpins vital soil functions, such as nutrient cycling, moderation of greenhouse gas emissions, carbon sequestration, water storage and water purification (Wall et al., 2012; Montanarella et al. 2015). This paper modelled the performance of soil biodiversity by adapting the Soil Biodiversity model from the Soil Navigator (http://www.soilnavigator.eu/) into a spatial model integrating local reference values and spatial data. Developed as part of the Horizon 2020 project LANDMARK (http://landmark2020.eu/), the Soil Navigator is an expert-based decision-tree approach that incorporates physical, chemical, and biological attributes (i.e., measurable data) for modelling the performance of soil functions in agricultural areas (Van Leeuwen et al., 2019). It does so by classifying attributes that define system characteristics (i.e., soil, environmental, management characteristics) into overarching classes at various tier levels. Attribute values are initially assigned performance scores (e.g., poor, moderate, good) based on thresholds assigned by technical (scientific) experts. Attributes are then grouped into overarching classes at higher tier levels and assigned performance levels, based on expert-based qualitative LUTs. Overall soil biodiversity performance (i.e., poor, moderate, good performance) for a particular test site is determined at the highest tier level. The Soil Navigator comprises two models for quantifying the performance of soil biodiversity – one for grasslands and one for croplands (Van Leeuwen et al., 2019). These models were adapted into a spatial model by replacing field measurements for individual attributes (e.g., pH, bulk density, earthworm abundance) with spatial data displaying the distribution of said attributes (soil characteristics, soil biophysical units maps), where available. Where spatial data was unavailable or inadequate (e.g., due to low spatial heterogeneity), attributes were assigned reference scores (e.g., overall poor, moderate, good performance) by experts. The Supplementary Material provides a detailed description of the way the model was adapted (Appendix 4, Section A4 – 4.5). It also presents maps displaying the distribution of soil biotic and abiotic attributes, which were used as input for the model (Appendix 4 – 6).

5.2.4.6 Sociocultural Values

The Sociocultural Values model captures the perceived importance of sociocultural values by local stakeholders in the Hoeksche Waard, and how they experience these values through their interaction with natural landscape elements. Natural landscape elements and sociocultural values were identified during the scoping phase of this study, based on the literature and semi-structured interviews conducted with stakeholders. Identified sociocultural values that are considered in this model include cultural identity and heritage, habitability, intrinsic values, landscape aesthetics, and recreation. A survey was subsequently developed and distributed by interviewed local representatives (number of respondents = 87). The survey enabled local stakeholders to assign a score to sociocultural values based on their perceived level of importance (0 = not important, 5 = very important). A low standard deviation was considered a robust choice for consideration of a proxy indicator (Rutgers et al., 2012). Respondents were then requested to link each sociocultural value to natural landscape elements to which they have been exposed. Linkages were made based on stakeholders' perceived importance of natural elements for experiencing each sociocultural value. Respondents were allowed to choose a maximum number of three elements per sociocultural value. Maps displaying the distribution of natural landscape elements typologies were developed by use of multiple layer LUT. LUT combine spatial datasets displaying the distribution of natural landscape elements (vegetation, land use, agricultural crop parcels maps) into individual layers displaying the distribution of each natural landscape element typology considered for this ecosystem service. For each sociocultural value, natural landscape elements were assigned a score. Each score is based on the average level of importance assigned to the sociocultural value and the average number of times each element was linked to it. These scores were integrated within qualitative LUT linking natural landscape elements to sociocultural values (Table 5.4). All calculations are available in the Supplementary Material (Appendix 4, Section A4 – 4.6).

Table 5.4: Weighted scores assigned to natural landscape element typologies for seven sociocultural value indicators. In bold, highest valued element for each indicator.

Sociocultural value indicator	Trees, hedges, wood walls	Grass-lands and shrubs	Large water bodies	Ditches and creeks	Agricul-tural fields	Polder struc-tures
Cultural history and identity	0.7	0.5	1.7	1.8	**4.3**	3.1
Educational and scientific knowledge	0.9	0.1	0.1	0.6	**4.4**	0.5
Habitability	0.8	1.1	**3.6**	1.1	1.4	3.3
Intrinsic values	3.9	4.1	3.4	**4.7**	0.5	2.1
Landscape aesthetics	3.4	**4.7**	4.2	3.5	3.2	**3.5**
Recreational potential	**4.6**	3.3	1.5	1.7	0.7	3.0

5.2.4.7 Property Value

Natural elements increase the amenity of residential areas, which reflects in property values (Czembrowski & Kronenberg 2016; Franco & Macdonald, 2018). The contribution of (the configuration and composition of) natural elements to property values was modelled by use of the NC-Model (Paulin et al., 2020b). The model combines spatial information on average property prices in different areas (key statistical figures map) and on the distribution of natural elements (vegetation, topographic land use maps), with causal reference values obtained from the literature (Remme et al., 2018). Reference values include regression model results linking different natural element typologies (e.g., park, open water, tree line) and their configurations (i.e., size, distance from property) with property prices (Daams et al., 2016; Ruijgrok & de Groot, 2006; Luttik & Zijlstra, 1997).

5.3 Results

For a comprehensive assessment of results, maps displaying the spatial distribution of ecosystem services (Figure 5.5) are analysed alongside total supply and use values calculated for the municipality of the Hoeksche Waard (Table 5.5). Maps display one supply or use indicator for each ecosystem service, based on the indicators presented in Table 5.1. In the Supplementary Material, maps displaying the distribution of indicators modelled by use of the Sociocultural Values model are found (Appendix 4 – 7), as well as descriptive statistics and correlations for all output maps (Appendix 4 – 5).

Results from the Crop Production model reveal production volumes that are substantially fragmented and heterogeneous across space, due to the presence of crop rotations that include a variety of cover crops. While the net output of harvested crops is valued at approximately €107 million/year (€6,800/year/productive ha), the value added of harvested crops (net output once labour and capital costs have been deducted) is estimated at less than 70% of net output. Results from the Air Quality Regulation model reveal substantially higher PM_{10} retention levels in the northeast of the Hoeksche Waard, an area adjacent to the cities of Rotterdam and Dordrecht. Benefits that were modelled by use of the Human Health and Property Value models are mainly visible within villages, where most of the municipality's inhabitants and built-up property are concentrated (see 'built-up and paved areas' in Figure 5.1). Results from the Pest Control model illustrate the degree of suppression of pests by natural enemy populations. Areas in yellow (score = 0) comprise areas where pests are present, but no biological pest control takes place, while areas in dark blue (score = 1.5) comprise areas where pest control is most effective. Results from the Soil Biodiversity model disclose that, on average, soil biodiversity performs at moderate to good levels. Results from the Sociocultural Values model capture the importance of natural landscape elements for the delivery of five locally-relevant cultural services. In general, average values for the entire area are often relatively low for indicators whose units are based on score systems (Pest Control, Sociocultural Values), as they bundle up high and low values across the entire area. An exception includes results from the Soil Biodiversity model, where the area's average score was high, as the performance of soil biodiversity is relatively high across the evaluated extent. On instances where averaging ecosystem service delivery scores leads to bias towards lower values, maps provide more indicative results. As maps display the distribution of ecosystem service supply and use, they facilitate the comparison of relative values across space.

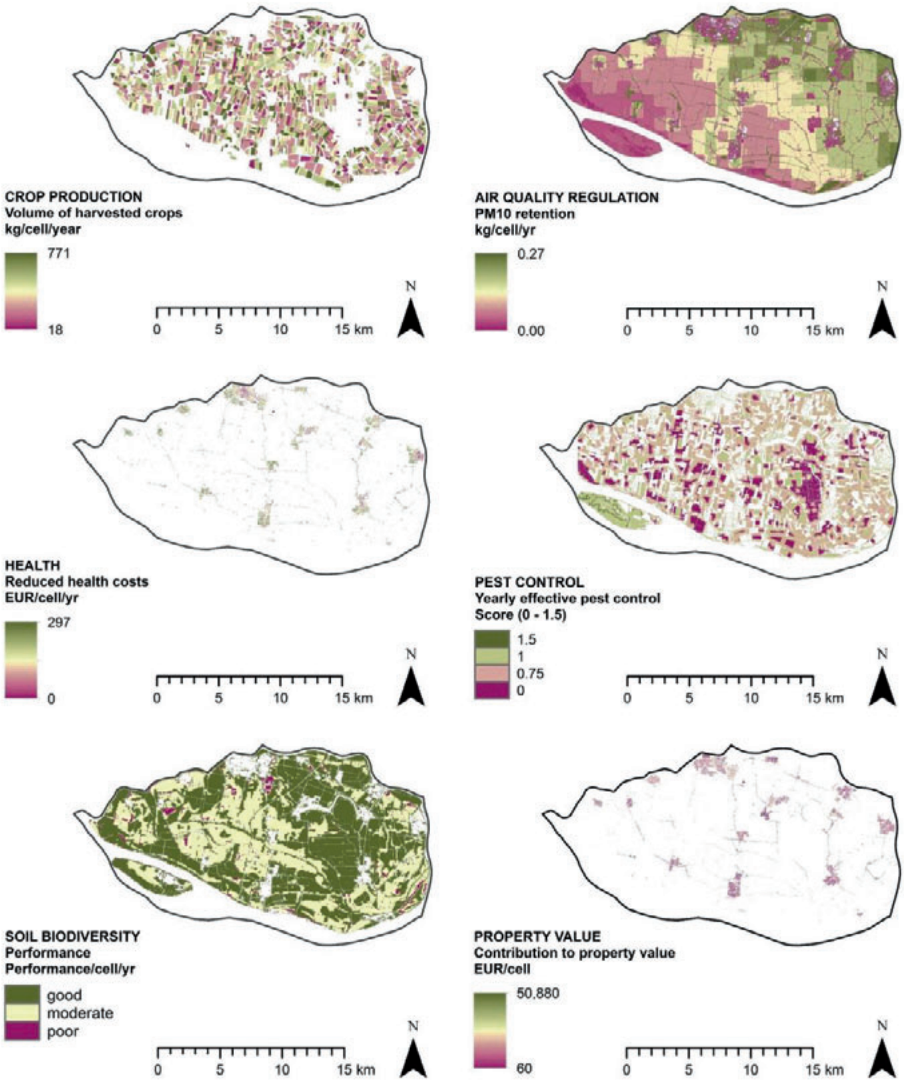

Figure 5.5: Output maps for six ecosystem services (cell size = 10 × 10 m) in the Municipality Hoeksche Waard. Unshaded areas comprise areas where no value has been assigned.

5

Table 5.5: Supply and use values of seven ecosystem services for the Municipality Hoeksche Waard

Ecosystem service model	Supply/ use	Indicator	Unit	Total value
Provisioning services				
Crop Production	Supply	Volume of harvested crops	million kg/yr	459
	Use	Net output of harvested crops	million €/yr	107
	Use	Value added of harvested crops	million €/yr	72
Regulation and maintenance services				
Air Quality	Supply	PM_{10} retention	thousand kg/yr	140
Regulation	Use	Reduced health costs	million €/yr	6
Health	Use	Reduced health costs	million €/yr	3.4
	Use	Reduced labour costs	million €/yr	17
	Use	Reduced visits to general practitioners	thousand visits/yr	4
Pest Control	Supply	Yearly effective pest control	average score 0-1.5	0.7
Soil Biodiversity	Supply	Average performance of soil biodiversity	performance classes	3
Cultural services				
Sociocultural	Supply	Croplands	thousand ha	19.4
Values	Supply	Large water bodies	thousand ha	1.8
	Supply	Grasslands and shrubs	thousand ha	7.6
	Supply	Polder structures	thousand ha	0.7
	Supply	Ditches and creeks	thousand ha	5.0
	Supply	Trees, hedges and wood walls	thousand ha	2.6
	Use	Cultural identity and heritage	average score 0-5	3.4
	Use	Educational and scientific knowledge	average score 0-5	3.2
	Use	Habitability	average score 0-5	1.5
	Use	Intrinsic value	average score 0-5	2.3
	Use	Landscape aesthetics	average score 0-5	3.7
	Use	Recreational potential	average score 0-5	1.7
Property Value	Use	Contribution to property value	million €	920

5.4 Discussion

5.4.1 Ecosystem services

5.4.1.1 Crop Production

Within the Crop Production model, production volumes and accrued monetary values are influenced by the type of crop under production. For instance, on average, the production (kg/ha) of sugar beets is higher than the production of brown beans by a factor of 40 (https://opendata.cbs.nl/). However, the market price of brown beans is higher than the price of sugar beets by a factor of 10 (https://agrimatie.nl/). In addition, a number of cover crops (e.g., English ryegrass, red and white clover) are less profitable but support the maintenance of soil quality within crop rotations (Munkholm et al., 2013). All these factors ultimately influence the net output (€/ha) for each crop type. Since a variety of crops with different profitability levels are combined within rotation systems, the net output for all crops produced (€107 million/year) is a more adequate indication of the profitability of farming systems than net output per crop type. An even more adequate measure of the sector's profitability is the value added of harvested crops (€72 million/year), as it considers net output once labour and capital costs have been deducted.

5.4.1.2 Air Quality Regulation

Within the Air Quality Regulation model, PM_{10} retention is influenced by the vegetation typology and its configuration, as well as the distribution of atmospheric concentrations of PM_{10}. Since land cover is relatively homogeneous across the Hoeksche Waard (i.e., mainly croplands and grasslands), the capacity for PM_{10} retention does not vary significantly. Instead, overall PM_{10} retention is mainly determined by atmospheric concentrations of PM_{10} (PM_{10} concentration map displayed in Appendix 4 – 8). This explains why significant uptake takes place in the northeast sector of the municipality, given its proximity to the cities of Rotterdam and Dordrecht. It also reveals that, while agriculture may act as a source of PM_{10} emissions (Lagerwerf et al., 2019), it may also behave as a sink. In this case, natural elements in the Hoeksche Waard act as sinks for urban pollution spill-overs.

5.4.1.3 Human Health

In the Hoeksche Waard, total reductions in health costs and health-related labour costs amount to €3.4 million and €17 million respectively. These values are substantially lower than values obtained by application of the NC-Model in the municipality of Amsterdam (health costs = €18.2 million, health-related labour costs = €90 million; Paulin et al., 2020a). This can be explained by the fact that Amsterdam's population is larger by a factor of 10, comprising a larger number of beneficiaries compared to the Hoeksche Waard. Despite this considerable difference, the total per capita contribution of green space to reductions in health costs and health related-labour costs is higher in the Hoeksche Waard than in Amsterdam by a factor of two. This can be explained by the abundance of vegetation that characterises predominant agricultural areas relative to predominantly urban areas. In general, human health maps share a weak correlation with other output maps (r = 0 - 0.1), with the exception of the use map modelled by use of the Property Value model (r = 0.5). This is likely the case since the contributions to human health and property value by natural elements are restricted to areas where inhabitants and built-up property are concentrated, which is not the case for other indicators considered.

5.4.1.4 Pest Control

Within the Pest Control model, pest suppression by natural enemy populations (i.e., insects that suppress pest populations) is determined by (i) the distribution of natural elements that act as habitat for pest organisms (i.e., aphids) and natural enemy populations (i.e., hoverflies, carabids, coccinellids), (ii) intraguild interactions (predation or complementarity) between natural enemy populations, and (iii) the mobility of individual pest or natural enemy populations. In the Hoeksche Waard, the landscape is predominantly fragmented into crop fields, which often serve as a habitat for aphids (Langoya & Van Rijn, 2008). Crop parcels are commonly surrounded by field margins and wood walls, which provide habitat for aphids and their natural enemies in various combinations (Van Rijn, 2014). Hence, effective pest control mainly takes place inside and in the proximity of field margins and wood walls surrounding crop parcels. Areas where no effective pest control takes place (score = 0) mainly comprise extensive grasslands or croplands that are too remote from woody elements and field margins to be reached by natural enemy populations. Pest control seems to be most effective in the south-eastern island of Tiengemeten, which mainly comprises natural protected areas. As such, it comprises a higher diversity of vegetation, hosting a more diverse distribution of pest and natural enemy populations than the predominantly agricultural landscape.

5.4.1.5 Soil Biodiversity

On average, soils in the Hoeksche Waard receive a performance score of 'good', the highest achievable score that can be obtained by implementation of the Soil Navigator. In general, soil biodiversity seems to be moderately correlated with crop productivity (r = 0.5). This may result in part from the harmonised management style that defines the area, as well as the many initiatives that seek to enhance soil biodiversity and overall quality. High performance is most prominent in grasslands and non-cropland natural areas (see Figure 5.1), which comprise a number of protected areas managed by nature organisations and farmer's collectives (e.g., Natuurmonumenten, Staatsbosbeheer, Rietgors Foundation; https://www.natuurmonumenten.nl/; https://www.staatsbosbeheer.nl/; www.rietgorsinfo.nl/).

5.4.1.6 Sociocultural Values

On average, the perceived contribution of the Hoeksche Waard's natural landscape elements is highest for the sociocultural value 'Landscape aesthetics', which receives a score of 3.7 (Table 5.5) on a scale of 0 to 5 (unimportant to very important). This occurs since a strong perceived linkage was identified between the sociocultural value and the grasslands and shrubs typology (score = 4.4; Table 5.4), and as all natural landscape element typologies considered received a score no lower than 3.2 (Table 5.4). Hence, the score assigned to grasslands and shrubs (26% of evaluated areas) and other natural landscape elements led to an average score of 3.7. Meanwhile, the average perceived contribution of the Hoeksche Waard's natural landscape elements is lowest for the sociocultural value 'Habitability' (score = 1.5; Table 5.5). This is the case since the highest valued natural element typology (i.e., large water bodies) receives an intermediate score of 3.6 (Table 5.4) and has a relatively low spatial extent (6% of evaluated areas), while all other typologies received lower scores, leading to a relatively low average score for evaluated areas. These observations shed light on the fact that average values may overlook natural element typologies that receive high scores, but which do not cover large spatial extents. For instance, for the 'Intrinsic values' indicator, ditches and creeks (17% of evaluated areas) are perceived as important (score = 4.7; Table 5.4), while for 'Recreational potential', trees, hedges, and woodwalls (9% of evaluated areas) are perceived as important (score = 4.6; Table 5.4). This accentuates the need to evaluate not only average values, but also the spatial distribution of these values across the landscape.

5.4.1.7 Property Value

The contribution of natural elements to property value is estimated at €920 million. This value is substantially lower than the value estimated by application of the NC-Model in the municipality of Amsterdam (€6 billion; Paulin et al., 2020a). This may occur in part due to Amsterdam's larger population size and consequently higher density of built-up property. However, the contribution of natural elements to property values is additionally influenced by the typology, configuration, and proximity of these elements to households. In Dutch cities, households are often concentrated in areas which are highly sealed and surrounded by substantially lower vegetation abundance, while villages in rural areas are commonly surrounded by abundant nature. This explains why, even though the population size in the Hoeksche Waard is smaller than that of Amsterdam by a factor of 10, the contribution of its natural elements to property value is smaller by a factor of six. In general, the use map produced by implementation of the Property Value model is weakly correlated with other output maps (r = 0 - 0.1), with the exception of maps modelled by use of the Human Health model (r = 0.5). As with results obtained by implementation of the Human Health model, this may occur in part since the contribution to property value by natural elements is restricted to areas where built-up property is concentrated, which is not the case for other indicators considered.

5.4.2 Local relevance

In the Hoeksche Waard, stakeholders share an intimate relationship with the natural landscape. Farmers seek to become more innovative to maintain and enhance the quality of the soils that sustain their crops, while enjoying the cultural and aesthetic value of the landscape. The local water board aims for safe and clean waterways (https://www.wshd.nl/). Influenced by different drivers, local governments (EZ, 2013b; South Holland Province, 2013) and environmental organisations (https://www.natuurmonumenten.nl/; https://www.staatsbosbeheer.nl/; www.hwl.nl/) strive to maintain and enhance the ecological and cultural value of the landscape. Locals value the aesthetics of grasslands and find educational and scientific potential in agricultural fields. These interests are epitomised in a history of collaboration between stakeholder groups that has led to a continuous search for innovative and multifunctional agriculture (Van Alebeek & Cleveling, 2005; Van Rijn et al., 2008). Agriculture defines and fragments the landscape into agricultural parcels, driving the economy of the area but threatening ecosystem resilience (e.g., soil compaction by use of heavy machinery; water quality, soil quality, and biodiversity degradation from pesticide use). To address this threat, the large-scale implementation of field margins seeks to suppress the use of pesticides and its negative impacts.

Results from this study reveal effectiveness in suppression of local pests due to the presence of field margins, in alignment with empirical research (Van Rijn & Wäckers, 2016). It also emphasises the importance of the configuration and composition of field margins and other natural enemy habitats for effective pest suppression. This may partially explain why, despite substantial agriculture taking place in the area, soils continue to enjoy moderate to good performance of soil biodiversity, in alignment with virtually all stakeholder objectives (Supplementary Material, Appendix 4 – 1). Villages enjoy enhanced property values and improved health due to the abundance of vegetation in their near surroundings. Meanwhile, this abundance of vegetation acts as a sink for atmospheric PM_{10} released in urban areas at the northeast of the municipality. Moreover, the assessment of sociocultural values reveals that locals value natural elements differently based on the particular sociocultural value under consideration. These values are maximised when highly-valued natural landscape elements comprise large extents, accentuating the spatial trade-offs that exist amongst sociocultural values in the area.

5.4.3 Methodological relevance

Limited consideration of ecosystem service assessment results within decision-making often results from the use of standardised models that are ill-suited for capturing context-specific spatial and thematic detail (Villa et al., 2014). To advance consideration of the ecosystem services concept within decision-making, there is a need for integrating local knowledge and data within assessments to capture context-specific spatial and thematic detail (Martínez-López et al., 2019). This study provided insights on how local knowledge and data can be incorporated to assess local ecosystem services. Assessing a diverse suite of locally-relevant ecosystem services, while integrating local knowledge and data, called for integration of various well-established spatial quantification methods. These methods vary in suitability given particular resource endowments. A key methodological advantage of considering local knowledge and data within assessments is that it enables the spatial quantification of ecosystem services at high spatial and thematic detail.

Flexibility in the selection of spatial quantification methods is instrumental, as knowledge and data availability vary across spatiotemporal gradients and may come in various forms, which requires adaptability. For a number of ecosystem services (i.e., Air Quality Regulation, Human Health, Property Value), models from the NC-Model were available for their quantification, making use of best-available local spatial data and reference values obtained from regression models. For remaining ecosystem services, readily-available spatial data, statistical data, and knowledge from empirical studies, were analysed to determine how spatial quantification methods could be integrated for their assessment. For the ecosystem service Crop Production, it was possible to develop a model based on readily-available local spatial data and statistics on crop production volumes and prices. The abundance of spatial data on soil characteristics and the aid of technical experts (i.e., scientists) made it possible to adapt the expert-based model underlying the Soil Navigator to model the performance of soil biodiversity. For the Pest Control model, it was difficult to obtain reference values based on regression models due to time and data constraints. Hence, with the aid of technical experts, a model was developed capturing the interactions between aphids and natural enemy populations based on the distribution of their habitats. Recognising that sociocultural values are largely context-dependent, proxy indicators for the Sociocultural Values model were identified and modelled through semi-structured interviews and participatory exercises with stakeholders.

From a thematic perspective, incorporating local knowledge within assessments can generate a better understanding of the socioeconomic and ecological context under assessment. This facilitates the process of selecting ecosystem services that will be assessed in a way that aligns with the needs and preferences of local stakeholders, as well as with local environmental characteristics. The consideration of socioeconomic and ecological characteristics that define spatiotemporal gradients within models can serve as a means to enhance the legitimacy of assessment results, supporting their uptake within decision-making (Orchard-Webb et al., 2016). In this study, ecosystem services were not selected a priori but were rather identified by conducting semi-structured interviews with local stakeholders and by reviewing local literature sources. This led to a selection and assessment of ecosystem services of central importance in the Hoeksche Waard, including site specific cultural services.

From a spatial perspective, the integration of local data at a high spatial resolution (where available) within ecosystem service models can enable their quantification at high spatial detail. This is useful for visualising the heterogeneity that characterises the distribution of ecosystem services at the local scale. Such level of detail is often difficult to obtain by implementing large scale standardised models that link secondary data to land use/land cover. These models often contribute to a binary view of what are in fact continuous and heterogeneous landscapes (Malinga et al., 2015), sacrificing the level of spatial detail that is necessary to manage ecosystem services at the local level (Derkzen et al., 2015; Martínez-López et al., 2019). Visualising the spatial distribution of ecosystem services at a high resolution enables the identification of spatial matches and mismatches between supply and use, as well as potential factors that influence their distribution (Paulin et al., 2020a). In this study, all models implemented made use of best available spatial data that is accepted and endorsed by Dutch local governments, enabling the assessment of ecosystem services at a high spatial resolution (10 x 10 m). This was key for identifying potential drivers that influence the distribution of assessed ecosystem services. Drivers include (but are not limited to) pressures that require mitigation (e.g., PM_{10}), as well as the distribution of ecosystem service beneficiaries.

5.4.4 Limitations

In this study, we demonstrated how local knowledge and data can be incorporated within ecosystem service models by use of well-established spatial quantification methods, thereby capturing locally-relevant spatial and thematic detail. While instrumental, a number of limitations should be articulated to users of the approach and receptors of its results.

First, coupled human-natural systems comprise a range of ecosystem services that interact with each other in a synergistic and antagonistic fashion. However, the difficulty of assessing multiple ecosystem services given limited resource endowments may lead to over- or underestimation of ecosystem service values. For instance, assessed ecosystem services in this study comprise only those which were prioritised based on the literature review and semi-structured interviews performed during this study's scoping phase. This may lead to an underestimation of ecosystem services not considered in this study (e.g., climate regulation by vegetation, runoff reduction by field margins) by potential end-users.

Second, ecosystem services are not always produced and consumed in situ (Syrbe & Walz, 2012), which may lead to over- or underestimation of ecosystem service production and use taking place outside of the assessed domain. For instance, results obtained by implementation of the Air Quality Regulation model revealed that the Hoeksche Waard acts as a sink for urban pollution spill-overs. Hence, the contribution by natural elements to health is likely to be higher than the estimated value, as beneficiaries residing outside of the municipality were not considered. In general, it is not possible to consider all elements of a system and their interactions (Paulin et al., 2020b). Instead, ecosystem service models can be implemented to portray a depiction of reality.

Third, this study does not deal with model uncertainty and validation. Uncertainty has often been overlooked within ecosystem service modelling studies (Haase et al., 2014) due to difficulty in its examination when considering large spatial datasets (Schuwirth et al., 2019; Abily et al., 2016; Lilburne & Tarantola, 2009). Moreover, validation of ecosystem service indicators is often challenging due to difficulties in obtaining observations. In this study, obtaining observations for some ecosystem service indicators was limited due to privacy concerns (e.g., the reduction of seven disease groups; the amount of time cycled by individuals) or due to their subjectivity (e.g., the contribution of natural elements to property value). In other instances, obtaining the observations necessary for validation would have entailed a time-consuming and expensive process (e.g., PM_{10} retention by vegetation and water; water storage). For these reasons, uncertainty analysis and validation were not considered in this study. Despite this limitation, the act of including local data and perceptions within the ecosystem service modelling process supports its veracity and adds legitimacy to the assessment.

5.5 Conclusions

In this paper, an integrated approach for the spatially-explicit quantification of local ecosystem services was presented. We demonstrated how six well-established ecosystem service assessment methods can be integrated to quantify and map locally-relevant ecosystem services, enabling incorporation of best-available local knowledge and data. In integrating various ecosystem service assessment methods, we successfully quantified and mapped a diverse suite of ecosystem service supply and use indicators that span across different value domains (i.e., ecological, economic, social domains), even in situations where models, knowledge, or data were limited. High resolution maps integrating local knowledge and data enabled identification of potential factors that influence ecosystem service delivery. These factors include (but are not limited to) the typology and configuration of natural elements, the presence of ecological pressures that require mitigation (e.g., PM_{10} concentrations generated in urban areas), and the distribution of villages that are home to ecosystem service beneficiaries. Total values quantified provided a glimpse of the magnitude of ecosystem service delivery in the area in units that speak to end users from various knowledge backgrounds. In addition, they emphasised the importance of the extent of natural elements for total ecosystem service delivery (e.g., the effect of the extent of particular natural elements on the total delivery of sociocultural values) and shed light on trade-offs that exist amongst ecosystem services (e.g., trade-offs amongst sociocultural values due to their reliance on different natural landscape elements).

Information obtained from implementation of such an approach is useful for a number of reasons. It facilitates communication of the complex mechanism by which ecosystems generate value for humans to decision-makers. In doing so, it can provide decision-makers with the tools to further communicate this information to other stakeholders. This information can empower stakeholders who wish to partake in the process of shaping the landscape in a way that considers their preferences and objectives. By considering these preferences and objectives, decision-makers can set priority areas to implement measures that optimise ecosystem service delivery in an inclusive manner. Decision-makers can additionally set ecosystem service targets for supply and use indicators, given the appropriate assessment of uncertainties associated with each model. To assess potential changes in ecosystem service delivery given the realisation of plausible futures (e.g., changes in natural element distribution from implementation of spatial strategies, population growth), input data can be altered to reflect changes and then incorporated within models. While the potential of ecosystem service modelling for integration of scientific knowledge into decision-making is latent, it is necessary to continue to produce biophysical and socioeconomic data relevant at various scales for their integration within models. This is instrumental for increasing the accuracy of assessments and systematically articulating model uncertainties. In doing so, it may enhance the effectiveness of model output to support spatial planning, thereby stimulating confidence in model response and hence uptake.

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6

CHAPTER 6

Synthesis

6.1 Introduction to synthesis

During the past 50 years, Earth's ecosystems have been altered at rates unprecedented in human history (Díaz et al., 2019). Halting their collapse to safeguard human well-being requires a paradigm shift in the way we embrace our entangled relationship with natural systems. The ecosystem services concept provides an opportunity for reifying nature's contributions to people. This enables their consideration within conventional management schemes and facilitates communication of their value to decision-makers at different levels. Despite its instrumental value, consideration of the concept within decision-making remains limited (Haase et al., 2014; Walsh et al., 2015). This partly results from issues related to (i) the ambiguous operationalisation of the delivery process, (ii) limited consideration of distinct value domains, and (iii) limited consideration of spatial and thematic detail within ecosystem service assessments. Focusing on these issues, this thesis aimed to develop approaches that facilitate the spatial quantification of ecosystem services at the regional and local level. It was hypothesised that tackling these issues is instrumental for informing decision-makers who wish to optimise the distribution of natural capital to support human well-being. To address the aim of this thesis, four research questions (RQ) were formulated.

RQ 1) How can existing approaches for operationalising the delivery process be harmonised to endorse clarity and consistency within ecosystem service assessments?

RQ 2) How can distinct value domains (i.e., biophysical, sociocultural, economic) be integrated within ecosystem service assessments?

RQ 3) How can ecosystem services be quantified at high spatial and thematic detail and across distinct value domains?

RQ 4) How can approaches for spatially quantifying ecosystem services be implemented to inform decision-making at the subnational (i.e., local, regional) level?

Section 6.2 provides a brief overview on the methodological approach implemented to address these RQ and the key findings of this thesis. Sections 6.3 to 6.6 further expand on these findings, providing a detailed account of how each RQ was addressed throughout Chapters 2 to 5. Section 6.7 expands on the methodological advances of this thesis and their relevance in a decision-making context. In Section 6.8, the final conclusions of this thesis are presented. Section 6.9 discusses remaining challenges for integration of ecosystem services within decision-making and provides recommendations for further research.

6.2 In a nutshell: Methodological approach and key findings of this thesis

In Chapter 2, a thorough review of multifunctional mapping studies revealed substantial ambiguity among well-established approaches for operationalising the ecosystem service delivery process. To endorse consistency of assessment results across space and time, a harmonised operationalisation framework was developed, linking delivery components formulated under identified approaches (i.e., supply and demand components, cascade components). The developed framework was then implemented to systematically assess the current consideration of diverse value domains (i.e., biophysical, sociocultural, economic) within multifunctional mapping studies. Despite numerous calls for better integration of anthropogenic aspects of delivery within assessments (i.e., demand side), it was found that mapping studies continue to overemphasise biophysical aspects of delivery (i.e., supply side), mapping mainly regulation and maintenance services. This calls for better integration of biophysical and anthropogenic aspects of delivery within assessments, given their inextricable relationship. A lack of synergy was additionally identified between the operationalisation of the delivery process into delivery components and the operationalisation of ecosystem services into sections (i.e., provisioning, regulation and maintenance, cultural services). This further adds to inconsistency and reduced comparability of assessment results across mapping studies. In doing so, it confuses application and interpretation of the ecosystem services concept and raises questions regarding the usefulness of operationalising ecosystem services into sections.

In Chapters 3 and 5, approaches were developed for spatially quantifying ecosystem services across the urban-rural gradient and at a high-resolution, incorporating best-available local knowledge and data. Their aim is to support the consideration of spatial and thematic detail within assessments, including diverse value domains, in order to meet the needs and expectations of decision-makers at local and regional (subnational) levels. To demonstrate their application, developed approaches were implemented to assess ecosystem services in the Municipality of Amsterdam (urban) and the Municipality of the Hoeksche Waard (rural) in Chapters 4 and 5. Incorporation of local knowledge and data enabled the assessment of ecosystem services at high thematic and spatial detail (10 x 10 m). This was useful for providing end users (e.g., decision-makers, stakeholders) with finely detailed information on the distribution and total value of ecosystem service supply and use, painting a picture of the complex mechanisms through which ecosystems benefit humans. This level of detail enabled identification of factors that influence the distribution of supply and use, which is instrumental for decision-makers who wish to manage natural capital in an optimal way (e.g., equitable, sustainable, efficient; Schröter et al., 2017). Information provided to decision-makers in various formats (e.g., diverse indicators, maps, total quantities, radar plots) can be used to further disseminate assessment results to the public and to investors, emphasising not just the costs but also the sociocultural and monetary benefits that the maintenance and enhancement of natural capital can generate for society.

6.3 Approaches for operationalising the delivery process

Since the publication of the Millennium Ecosystem Assessment (MA, 2005), several approaches for operationalising the delivery process have been developed, aiming to facilitate the integration of the ecosystem services concept within decision-making. Despite substantial progress in this respect, the plurality of approaches developed complicates application of the concept by practitioners (i.e., assessors, end users). The various interpretations of the delivery process within these approaches additionally limit the consistency of assessment results across space and time, limiting their comparability. This reduces credibility in assessment output and thereby its use to support decision-making (Villamagna et al., 2013; Luederitz et al., 2015; Wei et al., 2017). To address this issue, the following research question was posed:

RQ 1) *How can existing approaches for operationalising the delivery process be harmonised to endorse clarity and consistency within ecosystem service assessments?*

This RQ was addressed in Chapter 2, which sought to provide clarity and consistency regarding the operationalisation of the delivery process across value domains. To provide clarity, two well-established approaches for operationalising the delivery process were reviewed and analysed. The first approach, the ecosystem services cascade, operationalises the delivery process based on the different stages by which ecosystems generate value for humans. Here, ecosystem service delivery begins with the delivery of natural resources and processes by ecosystems, many of which serve a beneficial purpose for human well-being. These functional natural resources and processes, or ecosystem functions, provide benefits and values to society. The second approach operationalises the delivery process into delivery components that constitute the supply side and the demand side of delivery. Potential supply, actual supply, capacity, flow, demand, preferences, and social values are some of the many terms adopted to refer to supply and demand components across the literature. In general, this approach for operationalising the delivery process has been subject to a substantially wider range of interpretations across the scientific literature. Here, a single term is adopted to refer to diverse concepts and diverse terms are often adopted to refer to a single concept. Delivery components in both approaches have been formulated with enough flexibility to enable their varied interpretations, tailored to the case-specific needs of practitioners. However, vagueness and inconsistencies in their conceptualisation across approaches may confuse practitioners when attempting to implement these concepts in practice (Costanza et al., 2017; Potschin & Haines-Young, 2016).

To provide consistency, an operationalisation framework was developed, synthesising cascade and supply and demand components within one harmonised system. Ecosystem service supply was operationalised into two delivery components: potential supply and actual supply. Potential supply captures the full extent of natural resources and processes that ecosystems deliver, whereas actual supply captures only those which effectively contribute to human well-being (i.e., ecosystem functions). Ecosystem service demand was also operationalised into two components: use and societal demand. Use captures the socioeconomic benefits that people enjoy from their usage of ecosystem functions. Societal demand captures the amount of an ecosystem service that is desired by society, irrespective of whether it is met by its supply and actual use. Often neglected within assessments, the societal demand is central for optimising ecosystem service delivery in a way that meets the needs of current and future generations. In general, actual ecosystem service use does not necessarily reflect people's needs and preferences, as people's consumption possibilities are often limited by circumstantial factors (e.g., time, money, freedom). The developed framework is flexible in character, as delivery components can be spatially quantified by use of various modelling approaches based on case-specific resource endowments (i.e., knowledge, data, time). Despite its flexible character, delivery components were conceptualised in enough detail as to avoid ambiguity in their interpretation and subsequent use in practice. To ensure consistency, case examples of the implementation of the framework to operationalise the delivery process were presented.

6.4 State of the art: Ecosystem services mapping across value domains

Ecosystem services and the delivery process span across different value domains. Ecosystem functions and the benefits they generate for humans can be expressed in biophysical, sociocultural, or economic terms. Given the diverse forms in which ecosystem services manifest themselves, consideration of different sections within assessments is often viewed as necessary. Given the inextricable relationship between biophysical and socioeconomic aspects of delivery, a need for consideration of the demand and supply sides of delivery within assessments has also been established. Despite these needs, literature reviews conducted in recent years revealed that, in practice, anthropogenic aspects of delivery are often overshadowed by biophysical aspects within mapping studies (Maes et al., 2012; Martinez-Harms & Balvanera et al., 2012; Crossman et al., 2013; Luederitz et al., 2015; Malinga et al., 2015; Lautenbach et al., 2019). To evaluate progress towards a more holistic consideration of distinct value domains within assessments, the following RQ was posed:

RQ 2) *How can distinct value domains (i.e., biophysical, sociocultural, economic) be integrated within ecosystem service assessments?*

To address this RQ, a systematic literature review was performed in Chapter 2. Its aim was to evaluate current consideration of diverse value domains within multifunctional mapping studies (i.e., at least three ecosystem services mapped). A total number of 123 case studies comprised within 111 publications published between the years 2017 and 2019 were analysed to determine the frequency with which sections and delivery components are commonly mapped. Indicators mapped in the literature were operationalised into sections based on the Common International Classification of Ecosystem Services (CICES; Haines-Young & Potschin, 2011; https://cices.eu/) and into delivery components based on the developed operationalisation framework. A systematic analysis of the literature revealed continued bias in the representation of distinct value domains across mapping studies. In particular, a vast majority of mapped ecosystem service indicators have been constricted to the supply side of delivery, mainly expressed in biophysical terms and capturing regulation and maintenance services. Despite this clear bias, the number of cultural services mapped within assessments has been gradually increasing. This may partly explain the growing uptake of sociocultural indicators, generally measured by use of sociocultural valuation methods, as a means to express ecosystem service values. Furthermore, ecosystem service indicators have been scarcely expressed in monetary values. This is likely a consequence of long-standing critique on methodological and ethical aspects attributed to the economic valuation of nature (Scholte et al., 2015; Chee, 2004; Gómez-Baggethun & Ruiz-Pérez, 2011). It may also partly explain the declining rate at which provisioning services have been mapped across the literature. Given their tangible nature, provisioning services are often better represented within conventional markets and, as such, have often been subject to monetary valuation. For a more holistic consideration of value domains, a better representation of the demand side is needed within ecosystem service assessments (Haines-Young & Potschin-Young, 2018; Wei et al., 2017; Villamagna et al., 2013).

Furthermore, the systematic analysis of the literature raised questions regarding the practicality of classifying ecosystem services into sections for two main reasons. First, a single ecosystem service could constitute more than one section based on the stage of the delivery process under assessment (i.e., midpoint vs. endpoint). Regulation and maintenance services comprise ecological processes, which is why they will indisputably fall under the supply side of delivery. Ecological processes can benefit humans directly or indirectly by underpinning provisioning and cultural services. Cultural services are most often expressed by use of sociocultural indicators, which is why they often fall under the demand side of delivery. Provisioning services can fall in the supply side if their amount is expressed in biophysical units, or in the demand side if their value is expressed in sociocultural or monetary units. For instance, pollination is often classified as a regulation and maintenance service. Pollination may contribute to the production of crops (i.e., provisioning service), and to the availability of flowers that shape the aesthetic value of a landscape (i.e., cultural service). In this example, two delivery processes leading to two final benefits (i.e., food consumption, aesthetic value enjoyment) are viewed as constituting three ecosystem services (i.e., pollination, food production, aesthetic value) belonging to three different sections. Second, a single ecosystem service could constitute more than one section based on the indicator implemented for its reification. For instance, the supply of wild mushrooms can constitute a cultural service if related to mushroom picking as a recreational activity, and it can constitute a provisioning service if related to food production. This brings us back to RQ 1. Namely, to provide consistency within ecosystem service assessments, it is perhaps more practical to assess the delivery process that underpins ecosystem service endpoints (i.e., use) instead of constricting individual ecosystem services to a single value domain, as sections often do. In considering different aspects of the production chain, diverse value domains are considered, depicting the inextricable relationship that links the natural and socioeconomic systems.

6.5 Approaches for spatially quantifying ecosystem services at high spatial and thematic detail and across value domains

Despite their instrumental value at relatively-large scales, standardised models for spatially quantifying ecosystem services often make use of generalised spatial and non-spatial data as input. This leads to an oversimplification of the cross-scale heterogeneity that defines spatiotemporal gradients, sacrificing the spatial and thematic detail needed to support the needs and expectations of decision-makers at local and regional levels (Martínez-López et al., 2019; Hauck et al., 2013; Derkzen et al., 2015). To address this issue, the following RQ was posed:

RQ 3) How can ecosystem services be spatially quantified at high spatial and thematic detail and across distinct value domains?

This RQ was addressed in Chapter 3, which presented the Natural Capital Model (NC-Model), a spatially explicit set of models for quantifying and mapping ecosystem services within the Netherlands across the urban-rural gradient. The aim of the NC-Model is to support the integration of ecosystem services within policymaking in the Netherlands, contributing to the fulfilment of national and international environmental policy targets (EZ, 2013a; EC, 2011, 2013, 2019a; CBD, 2010). To account for spatial detail, models in the NC-Model can be implemented to map ecosystem services at a high spatial resolution (10 x 10 m), making use of best-available spatial data as input, accepted and endorsed by Dutch local governments. To account for thematic detail, models incorporate relationships among ecological and socioeconomic parameters specific to the Netherlands (where available), which have been established within empirical studies. To account for thematic heterogeneity across space, the NC-Model comprises a subset of ecosystem service models tailored to the urban environment, namely the Urban Natural Capital Model (Urban NC-Model; Remme et al., 2018). To account for thematic heterogeneity across time, the model is continuously under development and improvement by a collaboration of Dutch knowledge institutes (i.e., National Institute of Public Health and the Environment, RIVM; Wageningen ENvironmental Research, WENR; Netherlands Environmental Assessment Agency; PBL). Models enable the spatial quantification of indicators capturing different stages of the delivery process (i.e., supply and use), which allows for consideration of diverse value domains. Ecological and socioeconomic aspects of delivery are captured by use of a diversity of (biophysical, sociocultural, economic) indicators that reify ecosystem functions and the socioeconomic benefits they generate.

RQ 3 was further addressed in Chapter 5, which demonstrated how local knowledge and data can be included within assessments to spatially quantify ecosystem services across value domains capturing locally-relevant spatial and thematic detail. In general, standardised ecosystem service models often incorporate generalised knowledge and data as input (Petz et al., 2017), sacrificing the spatial and thematic detail needed to inform decision-makers at local and regional levels. Incorporation of local knowledge and data (where feasible) within assessments is instrumental for informing local decision-making as it can contribute to the spatial quantification of ecosystem services at high spatial and thematic detail, while adding legitimacy to the assessment process (Orchard-Webb et al., 2016). In Chapter 5, local knowledge and data were used to assess locally-relevant ecosystem services in the Dutch Municipality of the Hoeksche Waard. Incorporation of local knowledge and data within ecosystem service models was achieved by use of well-established spatial quantification methods. These included causal relationships, expert elicitation, primary data extrapolation, regression models, and look-up tables (Martínez-Harms & Balvanera, 2012; Schröter et al., 2015). Due to variations in resource availability across spatiotemporal gradients (e.g., knowledge, data, time), integrating local knowledge and data within ecosystem service models calls for flexibility in the selection of spatial quantification methods. Some ecosystem services were assessed by implementation of the NC-Model, which already incorporates local knowledge and data as input and thereby ensures a high degree of spatial and thematic detail. Given the absence of readily-available models for assessing remaining ecosystem services in high detail, additional models were developed. These models incorporated local, readily-available spatial data at a high resolution, statistical data, and knowledge from empirical studies. Where this information was insufficient to model ecosystem services, models incorporating local knowledge obtained through expert elicitation (i.e., surveys, semi-structured interviews), namely with scientists and stakeholders, were developed. The approach enabled assessment of the supply and use of locally-relevant ecosystem services, even in situations where models, knowledge, or data were limited.

6.6 Spatially quantifying ecosystem services for decision-support

This thesis developed approaches that facilitate the spatial quantification of ecosystem services. Developed approaches are expected to tackle issues that limit the consideration of assessment output within sustainable decision-making. Limiting factors addressed in this thesis concern ambiguity in the operationalisation of the delivery process, as well as limited consideration of distinct value domains and spatial and thematic detail within assessments. To address the practicality of developed approaches to inform decision-making, the following RQ was posed:

RQ 4) How can approaches for spatially quantifying ecosystem services be implemented to inform decision-making at the subnational (i.e., local, regional) level?

This RQ was addressed in Chapter 4, which demonstrated the practicality of the NC-Model to inform urban planning. In particular, the NC-Model was implemented to assess the effect of changes in green infrastructure (GI) on ecosystem services in the Municipality of Amsterdam. The assessment was performed to inform decision-makers from the Municipality involved in the development of the Green Quality Impulse (*KwaliteitsImpuls Groen*; Amsterdam Municipality, 2017b), a spatial plan for the expansion and improvement of Amsterdam's GI by the year 2025. The NC-Model was implemented to quantify and map ecosystem services within three GI strategy scenarios and one 'Business-As-Usual' scenario. Strategy scenarios captured changes in the distribution of GI that would result from implementation of different spatial strategies formulated under the Green Quality Impulse. The 'Business-As-Usual' scenario served as a benchmark to evaluate changes in GI in each strategy scenario and how these changes would affect the distribution, relative performance, and overall performance of ecosystem service supply and use. Application of the NC-Model enabled the spatial quantification of urban ecosystem service supply and use indicators at a high spatial resolution. Changes in ecosystem services were expressed in a diversity of indicators spanning across different value domains. In addition, results were presented to urban planners in various formats (e.g., total values, maps, radar plots). This was useful for raising awareness among them on the complex mechanism by which GI influences the well-being of Amsterdam's residents. It also endowed them with tools to further disseminate results in a way that speaks to target groups (e.g., decision-makers, investors, local inhabitants) with diverse interests and knowledge backgrounds. High-resolution maps enabled identification of key factors that affect ecosystem service distribution in Amsterdam (e.g., GI configuration and typology; population density). In better understanding how changes in GI can affect urban well-being, urban planners can develop, rethink, and prioritise GI strategies in alignment with local objectives, based on scientifically-sound information. Equipped with the right tools, they can additionally build a case for investments in GI, emphasising the maintenance and enhancement of GI as a potential investment opportunity rather than merely as a cost (Toxopeus, 2019).

RQ 4 was additionally addressed in Chapter 5, where local knowledge and data were integrated within ecosystem service models by use of various well-established spatial quantification methods in order to assess locally-relevant ecosystem services in the Dutch Municipality of the Hoeksche Waard. In the Hoeksche Waard, a predominantly agricultural area defined by unique cultural, historical, and natural characteristics, a common interest has been shown by local stakeholders to collaborate to meet their common objectives. Ecosystem services were assessed by incorporation of various spatial quantification methods based on case-specific resource endowments. From a thematic perspective, incorporation of local knowledge enabled the selection and assessment of ecosystem services of central importance in the Hoeksche Waard, aligning with local needs, preferences, and ecological characteristics. Assessed ecosystem services were not selected a priori by but were rather selected as a result of semi-structured interviews with local stakeholders and by reviewing local literature sources. From a spatial perspective, quantification of ecosystem services in high spatial detail was instrumental for visualising the heterogeneity that defines the distribution of ecosystem service supply and use, as well as identifying potential factors that influence their distribution. As in Chapter 4, identified factors influencing the distribution of ecosystem service supply and use included the configuration and typology of natural elements, as well as the distribution of inhabitants. In particular, the distribution of inhabitants acts as a proxy for the distribution of ecosystem service beneficiaries and is often linked to ecological pressures that require mitigation (e.g., pollution in cities), influencing the societal demand for ecosystem services. Results obtained are useful for raising awareness among decision-makers regarding the complex mechanism that underpins ecosystem service delivery. The integration of local views to identify and assess ecosystem services can additionally add legitimacy to the assessment process by empowering stakeholders who wish to partake in the process of shaping the landscape.

6.7 Towards integration of ecosystem services within sustainable decision-making

6.7.1 Current developments and challenges

Today, a growing number of international initiatives call for integration of ecosystem services within decision-making, such as the Convention on Biological Diversity (CBD; 2010), Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES; Pascual et al., 2017), UN Sustainable Development Goals (SDGs; UN, 2017), and the EU 2020 Biodiversity Strategy (EC, 2011). This has resulted in various initiatives aiming to facilitate this task, such as MAES (Mapping and Assessment of Ecosystems and their Services; Maes et al., 2015a), ESMERALDA (Enhancing ecoSysteM sERvices mApping for poLicy and Decision mAking; Vihervaara et al., 2019), MAIA (Mapping and Assessment for Integrated ecosystem Accounting; https://maiaportal.eu/), and the System of Environmental-Economic Accounting (SEEA; UN, 2014). It has also resulted in growing efforts by governments to incorporate ecosystem services within decision-making at the national and subnational levels (e.g., UK NEA, 2011; EME, 2012, 2014; NOU, 2013; Rugani et al., 2014; EZ, 2013a; 2019). Despite growing recognition of the ecosystem services concept as a means for achieving sustainable outcomes, its incorporation within decision-making is not a simple task. The process by which ecosystems provide benefits to humans is highly complex. This process ranges across different value domains, which is why its assessment requires expertise from various disciplines.

To facilitate the assessment of ecosystem services and the interpretation of assessment results by decision-makers, consistency in ecosystem services terminology, operationalisation frameworks, and models is needed. To support this need, a rapidly increasing number of operationalisation and assessment approaches have been developed in recent years. Despite their practicality, operationalisation approaches and their respective terminology (e.g., Haines-Young & Potschin, 2018; TEEB, 2010; MA, 2005; Potschin & Haines-Young, 2016; Maes et al., 2015a; Villamagna et al., 2013; Crossman et al., 2013; Johnson et al., 2010) are subject to diverse interpretations, leading to substantial ambiguity in the way ecosystem services are assessed. Moreover, the range of tools that have been developed to facilitate the assessment process (e.g., InVEST, ARIES, ESTIMAP; Tallis and Polaski, 2011; Villa et al., 2014; Zulian et al., 2014) often model ecosystem services by incorporation of spatial and non-spatial input data available at relatively large scales (e.g., international statistical databases; land use/land cover maps instead of high-resolution vegetation maps). This leads to a substantial underrepresentation of the (ecological, sociocultural, economic) heterogeneity that defines spatiotemporal gradients, and which forms a core aspect ecosystem service delivery. As a result, output produced by implementation of these tools often lacks the spatial and thematic detail necessary to support sustainable decision-making in local and regional settings. This may explain why assessments performed at these scales are often explorative rather than conclusive in nature.

6.7.2 Methodological advances of this thesis and relevance in a decision-support setting

It is becoming increasingly evident that incorporating ecosystem services within local and regional decision-making requires the development of (i) a simplified and reliable system for operationalising the delivery process and of (ii) assessment approaches that produce time- and scale-specific results relevant at the local and regional levels. This thesis harmonised a plurality of novel yet mutually conflicting approaches for operationalising the delivery process within one optimised framework. This development presents a step forward towards consistency in the assessment of ecosystem services and the elements that constitute their delivery. The consistent assessment of ecosystem service delivery components under one framework is important for assessing their relative distribution across spatiotemporal gradients, defined by heterogeneous ecological and socioeconomic characteristics. It is also necessary for evaluating the efficacy of ecosystem-based management strategies by systematically monitoring their effect on ecosystem service supply and use. From a practical perspective, the developed framework was proven useful, as it successfully enabled the operationalisation of commonly assessed ecosystem service indicators across different stages of the delivery process. The operationalisation framework was also successfully implemented to assess ecosystem services across the urban-rural gradient in the Netherlands.

Approaches were additionally developed for assessing ecosystem services at high spatial and thematic detail and across diverse value domains. Detailed guidance for the implementation of these approaches was provided, supporting their subsequent replication and potential adaptation across different settings. This development presents a step forward from standardised approaches that are useful for assessing ecosystem services at relatively large scales, but which oversimplify the (ecological, sociocultural, economic) heterogeneity that characterises the local and regional scales. Developed approaches were effectively implemented to illustrate the distribution of ecosystem service supply and use in the municipalities of Amsterdam (urban) and the Hoeksche Waard (rural) in the Netherlands, contributing to the fulfilment of national and international environmental policy targets (EZ, 2013a, b; IenM, 2016, 2017; IenW, 2019; EC, 2011, 2013, 2019a). By incorporating local knowledge and data, locally-relevant ecosystem services were identified and assessed in a way that reflects the ecological and socioeconomic conditions of each study site. Ecosystem service indicators assessed covered various value domains, speaking to local and regional decision-makers with diverse backgrounds and preferences. Results presented in various formats provide decision-makers with the tools necessary to incorporate ecosystems within local and regional spatial planning in a scientifically-sound manner.

As developed approaches can be implemented to assess locally-relevant ecosystem services in a consistent manner and in high detail, their implementation is instrumental for informing ecosystem-based decision-making at the subnational scale. In doing so, these approaches can contribute to achieving national and international environmental policy targets that aim to maintain and enhance the well-being of current and future generations in the face of a rapidly changing environment. In the Netherlands, the NC-Model is already in use by various well-established knowledge institutes to inform local and regional decision-making. It is being made available for all local decision-makers and currently underpins the Netherlands Atlas of Natural Capital (https://atlasnaturalcapital.nl). In the future, developed approaches could be optimised in alignment with additional frameworks that support the harmonised integration of ecosystem services within decision-making, such as the System of Environmental-Economic Accounting (SEEA; UN, 2014) at the international level, or the Natural Capital Accounts (Hein et al., 2020; CBS, n.d. -b) at the Dutch level.

6.8 Conclusions

- A plurality of international initiatives calls for integration of ecosystem services within decision-making at the national level (e.g., CBD; 2010; Pascual et al., 2017; UN, 2017; EC, 2011). However, incorporation of ecosystem services within local and regional decision-making remains limited. In this thesis, approaches were developed to tackle three core challenges that currently limit the integration of ecosystem services within decision-making at the national and subnational levels.
- Harmonisation in the operationalisation of ecosystem services is key for enhancing the comparability of model output across space and time. In this thesis, a framework was developed, optimising approaches for operationalising the delivery process within one coherent system. It was effectively implemented to operationalise ecosystem service supply and use across the mapping literature and within two case studies in the Netherlands.
- Incorporation of local knowledge and data within assessments is instrumental for identifying locally-relevant ecosystem services and for producing output that speaks to decision-makers with diverse backgrounds and interests. In this thesis, output includes detailed information on the distribution of ecosystem services, capturing location-specific ecological, sociocultural, and economic characteristics. This enabled identification of factors that influence ecosystem service delivery. Key factors included the configuration and typology of natural elements, as well as the distribution of inhabitants.
- At the start of the project, it was hypothesised that the assessment of ecosystem services in a consistent manner, considering location-specific spatial and thematic detail, and covering diverse value domains, is instrumental for informing ecosystem-based decision-making. Approaches developed in this thesis were designed to facilitate the assessment of ecosystem services and the interpretation of assessment results by decision-makers. Developed approaches were effectively implemented to inform decision-makers in the municipalities of Amsterdam and the Hoeksche Waard, considering local environmental objectives, as well as the needs and preferences of local stakeholders.
- Reifying and measuring the benefits that natural capital generates for humans is necessary for building business cases for investments in ecosystem-based management, emphasising not just the costs but also the sociocultural and monetary benefits that the maintenance and enhancement of natural capital can generate for society.

6.9 Remaining challenges and the way forward

In this thesis, three general limitations that negatively influence the incorporation of ecosystem services within local and regional decision-making were addressed and, to a large extent, solved. These included (i) the ambiguous operationalisation of the delivery process, (ii) limited consideration of distinct value domains, and (iii) limited consideration of spatial and thematic detail within ecosystem service assessments. For the successful assimilation of ecosystem services within decision-making, additional challenges need to be addressed. Two of these challenges are described below:

6.9.1 Uncertainty analysis

The transparent communication of uncertainty to decision-makers is necessary to provide them with the level of confidence necessary when devising spatial strategies that will permanently alter the landscape. Uncertainty may arise from difficulties in the communication of ecosystem service endpoints by assessors to end users. Communication uncertainty was partially addressed by developing a unified approach for operationalising the delivery process into intermediate (supply) and final (use) indicators. Methodological uncertainty (i.e., uncertainty related to the choice of methods; Bilcke et al., 2011) was addressed by presenting and demonstrating the implementation of various approaches for the spatial assessment of ecosystem services. These approaches can be implemented by assessors based on case-specific resource endowments (i.e., data, time, knowledge) and their methodological preferences. This thesis did not deal with parameter uncertainty (e.g., measurement errors, sampling errors, variability; Bilcke et al., 2011) due to difficulty in its examination when considering large spatial datasets (Schuwirth et al., 2019; Abily et al., 2016). This includes difficulty in the validation of model parameters due to, for instance, their abstract nature, privacy concerns, or incurred costs. In addition, this thesis did not deal with structural uncertainty (i.e., uncertainty related to the functional form of models; Bojke et al., 2009). Parameter and structural uncertainty can be addressed by performing uncertainty analyses (e.g., Monte Carlo simulation, Bayesian Inference, multi-model ensembles; Schuwirth et al., 2019) with the aid of the growing number of software tools that facilitate this process (e.g., R package spup, JAGS, Winbugs; Sawicka et al., 2018; Lunn et al., 2000; Plummer, 2003). In addition, structural uncertainty may be reduced by periodically conducting regression studies capturing scale-dependent relationships between biophysical and socioeconomic parameters, making use of best-available local knowledge and data as input. Based on identified relationships, models can be continuously developed and calibrated, ensuring their continuous improvement, as well as the production of results that effectively reflect particular ecological and socioeconomic contexts. This task can additionally be facilitated by periodically developing and improving the quality of data used as input for models and regressions, thereby also reducing parameter uncertainty.

6.9.2 Environmental policy targets

While approaches presented in this thesis are useful to inform ecosystem-based decision-making, the actual incorporation of ecosystem services within tangible decision-making requires practical approaches providing guidance for translating assessment results into environmental policy targets. These may include environmental quality standards based on (i) ecological thresholds determined by scientists and technicians, or based on (ii) normative standards reached by mutual agreement between decision-makers and stakeholders. Setting policy targets is necessary for monitoring the efficacy of formulated management strategies, which is necessary for optimising ecosystem service delivery in the long run. Today, a number of approaches aim to facilitate decision-making based on information obtained from ecosystem service assessments (e.g., Ranganathan et al., 2008; Olander et al., 2015; EC, 2019b). For instance, the Corporate Ecosystem Services Review (Hanson et al., 2012) provides guidance for assessing the dependency and impact of firms on ecosystem services and for identifying the risks and opportunities that result from these relationships. While these approaches present a step towards tangible ecosystem-based management, further developments are required to support the actual translation of assessment output into environmental management strategies and policy targets. Given the growing use of cost-benefit analyses (CBA) to inform decision-making (Greenhalgh et al., 2017), ecosystem services can be included within management strategies through their incorporation within social cost-benefit analyses (SCBA). SCBA assess both internal and external impacts that could result from implementation of management strategies in order to reduce uncertainty and risk associated with such an endeavour (Medina-Mijangos et al., 2020). During integration of ecosystem services within SCBA, changes in ecosystem services reflect external costs and benefits that would result from implementation of envisioned management strategies, which would otherwise be excluded from conventional CBA. Based on information from conducted SCBA, decision-makers can design management strategies aiming to maintain or enhance ecosystem service delivery where desirable and set policy targets to monitor progress towards formulated objectives. Moreover, frameworks such as the ‘planetary boundaries’ (Rockström et al., 2009) and the ‘doughnut’ (Raworth, 2012) could provide guidance for formulating ecosystem service normative targets or for establishing thresholds under which human well-being in current and future generations is threatened.

References, Summary, Samenvatting, Acknowledgements, Curriculum Vitae

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SUMMARY

During the past 50 years, Earth's ecosystems have been altered at rates unprecedented in human history. Halting their collapse to safeguard human well-being requires a paradigm shift in the way we embrace our entangled relationship with natural systems. The ecosystem services concept provides an opportunity for reifying nature's contributions to people. This enables consideration of these contributions within conventional management schemes and their communication to decision-makers at different levels. Recognising its instrumental value, a number of initiatives calling for integration of the ecosystem services concept within decision-making have emerged, such as the Convention on Biological Diversity (CBD), UN Sustainable Development Goals (SDGs), and the EU 2020 Biodiversity Strategy. Despite its instrumental value, the consideration of ecosystem services within local and regional decision-making remains limited. To address this challenge, this thesis aims to develop approaches that facilitate the spatial assessment of ecosystem services at local and regional levels. Developed approaches are expected to tackle issues that limit the consideration of assessment output within sustainable decision-making. Limiting factors addressed in this thesis concern ambiguity in the operationalisation of the ecosystem service delivery process, as well as limited consideration of different value domains (i.e., values commonly regarded under distinct disciplines, such as biophysical, economic, and sociocultural values) and spatial and thematic detail within assessments. Tackling these issues could lead to a better uptake of the ecosystem services concept within decision-making at local and regional levels by adding consistency, credibility, and legitimacy to assessments. It is hypothesised that assessing ecosystem services in a consistent manner, considering location-specific spatial and thematic detail, and covering diverse value domains, is instrumental for informing decision-makers who wish to optimise the distribution of natural capital to support human well-being. The aim of this thesis is addressed by evaluating four research questions (RQ):

RQ 1) How can existing approaches for operationalising the delivery process be harmonised to endorse clarity and consistency within ecosystem service assessments?

RQ 2) How can distinct value domains (i.e., biophysical, sociocultural, economic) be integrated within ecosystem service assessments?

RQ 3) How can ecosystem services be quantified at high spatial and thematic detail and across distinct value domains?

RQ 4) How can approaches for spatially quantifying ecosystem services be implemented to inform decision-making at the subnational (i.e., local, regional) level?

RQ 1 is addressed in Chapter 2, which reviews two well-established approaches for operationalising the delivery process (i.e., cascade framework; supply and demand). Due to identified ambiguity in the way the delivery process is commonly operationalised, an operationalisation framework has been developed, synthesising cascade and supply and demand components within one harmonised system. The developed framework is flexible in character, as delivery components can be spatially quantified by use of various modelling approaches based on case-specific resource endowments (i.e., knowledge, data, time). Despite the framework's flexible character, delivery components are conceptualised in enough detail as to avoid ambiguity in the interpretation and subsequent use of each concept in practice. To ensure consistency, case examples of the implementation of the framework to operationalise the delivery process are presented. Chapter 2 additionally addresses RQ 2. To answer this RQ, recent multifunctional ecosystem service mapping studies have been systematically reviewed to assess the consideration of diverse value domains across the mapping literature. A particular focus was placed on the extent to which delivery components (i.e., ecological and socioeconomic building blocks that constitute the ecosystem service delivery process) and sections (i.e., provisioning, regulation and maintenance, and cultural

services) are generally considered in multifunctional mapping studies. Indicators mapped in the literature were operationalised into sections based on the Common International Classification of Ecosystem Services (CICES) and were operationalised into delivery components based on the operationalisation framework previously developed. It is found that a vast majority of ecosystem service indicators mapped are constricted to the supply side of delivery (i.e., ecological), expressed in biophysical terms, and capture regulation and maintenance services. This calls for a better representation of the demand side (i.e., sociocultural, economic) within mapping studies, acknowledging the inextricable relationship that links ecological and socioeconomic aspects of delivery.

RQ 3 is addressed in Chapters 3 and 5. In Chapter 3, the Natural Capital Model (NC-Model) is presented, namely a set of models for spatially quantifying ecosystem services in the Netherlands across the urban-rural gradient and at high spatial and thematic detail. Models capture spatial detail by considering best-available spatial data at a high resolution (10 x 10 m). Thematic detail is captured by integrating relationships between biophysical and socioeconomic parameters respective to the Netherlands. Models are continuously updated, reflecting changes in knowledge and environmental factors that influence final ecosystem service supply and use. In Chapter 5, an approach has been developed for integrating local knowledge and data within models to spatially quantify locally-relevant ecosystem services across value domains. Using local spatial data as input for models is useful for capturing the distribution of supply and use in high spatial detail. Using local knowledge and non-spatial data as input is useful for capturing the thematic detail that defines spatiotemporal gradients. Best-available local knowledge and data is incorporated within models by integrating various spatial quantification methods. These methods vary in suitability based on case-specific resource endowments. Locally-relevant ecosystem services were not selected a priori by but were rather selected as a result of semi-structured interviews with local stakeholders and by reviewing local literature sources.

RQ 4 is addressed in Chapter 4, where the NC-Model is implemented to assess the effect of changes in green infrastructure (GI) on ecosystem services in the Municipality of Amsterdam. The assessment has been performed to inform decision-makers from the Municipality involved in the development of the Green Quality Impulse (*KwaliteitsImpuls Groen*), a spatial plan for the expansion and improvement of Amsterdam's GI by the year 2025. RQ 4 is additionally addressed in Chapter 5, where the approach developed for integrating local knowledge and data within assessments is successfully implemented to assess locally-relevant ecosystem services in the Municipality of the Hoeksche Waard. In both case studies, quantification of ecosystem services in high spatial and thematic detail proves instrumental for visualising the spatial heterogeneity that defines ecosystem service supply and use, and for identifying potential factors that influence their distribution. Key factors identified include the configuration and typology of natural elements, as well as the distribution of inhabitants. The distribution of inhabitants acts as a proxy for the distribution of ecosystem service beneficiaries and is often linked to ecological pressures that require mitigation (e.g., pollution in cities). Results obtained by implementation of developed approaches are useful for raising awareness among decision-makers regarding the complex mechanism that underpins ecosystem service delivery. In better understanding how strategies that affect the spatial distribution of natural capital also can affect human well-being, spatial planners can develop, rethink, and prioritise management strategies in alignment with local objectives, based on scientifically-sound information. Equipped with the right tools, they can additionally build a case for investments in natural capital, emphasising its maintenance and enhancement as a potential investment opportunity rather than merely as a cost.

Chapter 6 synthesises the key findings of this thesis and discusses how RQ posed were addressed in preceding chapters. Based on this information, the methodological advances of this thesis and its relevance in a decision making context are synthesised. In general, this thesis fulfils its aim by effectively developing approaches facilitating the spatial assessment of ecosystem services in high spatial and thematic detail, across diverse value domains, and in a consistent manner. The hypothesis of the thesis is proven correct, elucidating how developed approaches can contribute to informing sustainable decision-making, with the aid of theoretical and practical examples. In particular, the developed operationalisation framework was effectively implemented to operationalise ecosystem service supply and use across the mapping literature and within two case studies in the Netherlands. Developed spatial assessment approaches, incorporating best-available local knowledge and data, enabled identification of locally-relevant ecosystem services and the production of assessment output that speaks to decision-makers with diverse backgrounds and interests. Reifying and measuring nature's contributions to people is key for building business cases for investments in natural capital, emphasising not just the costs but also the sociocultural and monetary benefits that its maintenance and enhancement can generate for society. This thesis concludes by briefly discussing remaining challenges that require attention to further endorse the integration of ecosystem services within decision-making (Chapter 6).

SAMENVATTING

In de laatste 50 jaar veranderen ecosystemen wereldwijd met een snelheid die ongekend is in de geschiedenis van de mensheid. Voor het welzijn van de mens dient deze afbraak te stoppen. Dit vereist echter een verschuiving binnen ons denken over hoe ons bestaan is verweven met natuurlijke systemen, opdat we een duurzame samenleving kunnen bouwen. Een benadering waarbij de voordelen die de natuur aan de mens levert – de ecosysteemdiensten – in beeld gebracht worden, biedt daarvoor kansen. Ecosysteemdiensten zijn uit te leggen aan besluitvormers op verschillende beleidsniveaus en zijn inpasbaar in traditionele vormen van omgevingsbeheer. Diverse initiatieven roepen daarom op tot integratie van het concept binnen gremia waar de plan- en besluitvorming plaatsvindt, zoals de VN Conventie over Biologische Diversiteit (CBD), de VN Duurzame Ontwikkelingsdoelen (SDGs) en de EU-biodiversiteitsstrategie voor 2020. Ondanks de instrumentele waarde van een benadering met ecosysteemdiensten blijft de toepassing daarvan binnen lokale en regionale besluitvorming tot op heden beperkt.

Dit proefschrift draagt bij aan het ontwikkelen en verbeteren van instrumenten om ecosysteemdiensten te kwantificeren en op lokale en regionale schaal in kaart te brengen. Dit proefschrift richt zich op de opheffing van ambiguïteit in het operationaliseren van het leveringsproces, d.w.z. het proces dat tot levering van ecosysteemdiensten leidt. Een tweede punt richt zich op een betere inpassing van de elkaar aanvullende domeinen waar waarde wordt toegevoegd, namelijk ecologische, socioculturele en economische waarde-domeinen. Het laatste onderwerp betreft het wegnemen van beperkingen om ruimtelijk, temporeel en thematisch detail binnen waardering van ecosysteemdiensten aan te brengen. Door deze beperkingen weg te nemen, wordt consistentie, geloofwaardigheid en legitimiteit van inschattingen bevorderd, wat tot een verhoogde acceptatie van informatie over ecosysteemdiensten binnen besluitvorming op lokaal en regionaal niveau kan leiden. De bijbehorende hypothese stelt dat besluitvormers die de waarde en de verdeling van natuurlijk kapitaal willen optimaliseren kunnen profiteren van het kwantificeren van ecosysteemdiensten: (i) op consistente manier, (ii) met beschouwing van alle van belang zijnde locatie-specifieke, ruimtelijke en thematische details, en (iii) door alle waarde-domeinen erbij te betrekken. Het doel van dit proefschrift wordt omschreven in vier onderzoeksvragen (OV):

OV 1) Hoe kunnen bestaande benaderingen voor het operationaliseren van het leveringsproces geharmoniseerd worden om helderheid en consistentie bij de kwantificering van ecosysteemdiensten te bevorderen?

OV 2) Hoe kunnen de diverse waarde-domeinen (d.w.z. ecologisch, sociocultureel, economisch) binnen inschattingen geïntegreerd worden?

OV 3) Hoe kunnen ecosysteemdiensten met voldoende ruimtelijk en thematisch detail worden gekwantificeerd, waarbij ook rekening gehouden wordt met de uitdrukking in verschillende waarde-domeinen?

OV 4) Hoe dienen complete sets met ruimtelijk gekwantificeerde ecosysteemdiensten aangeboden te worden om besluitvormers op lokaal en regionaal niveau op een praktische en heldere manier te informeren?

OV 1 wordt in Hoofdstuk 2 behandeld. Twee welbekende benaderingen voor het operationaliseren van het proces om levering van ecosysteemdiensten te beschrijven (het 'cascade' kader en het vraag-en-aanbod kader) zijn gereviewd. De ambiguïteit in de manier waarop het leveringsproces vaak wordt geoperationaliseerd vormde de aanleiding om een kader te ontwikkelen dat cascade componenten, zoals vraag-en-aanbod componenten, eenduidig formuleert en harmoniseert. Dit kader is flexibel aangezien

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componenten die samen het leveringsproces omvatten, gekwantificeerd kunnen worden door gebruik te maken van verschillende modellen, afhankelijk van casus-gerelateerde beschikbare informatie, kennis en data. Ondanks deze flexibiliteit worden leveringscomponenten op een zodanige manier geformuleerd dat ambiguïteit in de interpretatie en het gebruik van concepten in de praktijk wordt vermeden. Om ervoor te zorgen dat het resultaat consistent is, wordt de toepassing van het kader voor het operationaliseren van het leveringsproces toegelicht door middel van praktische en theoretische voorbeelden.

In Hoofdstuk 2 wordt ook OV 2 behandeld. Recent onderzoek over het in kaart brengen van ecosysteemdiensten werd op een systematische manier gereviewd, om inzicht te krijgen in de mate en wijze waarop diverse waarde-domeinen tijdens het in kaart brengen van ecosysteemdiensten worden beschouwd. Dit leverde inzicht in de mate waarin verschillende (ecologische en socio-economische) leveringscomponenten worden meegenomen en welke secties (d.w.z. productie, regulerende, en culturele ecosysteemdiensten) vooral aandacht krijgen bij het in kaart brengen van ecosysteemdiensten. Indicatoren voor het in kaart brengen waren (i) in secties ingedeeld op basis van de standaard van de Gemeenschappelijke Internationale Classificatie van Ecosysteemdiensten (CICES), en waren (ii) in leveringscomponenten ingedeeld op basis van het ontwikkelde operationalisatie-kader. Uit de studie bleek dat de meeste kaarten met ecosysteemdiensten beperkt zijn tot de aanbodkant van het leveringsproces, en vaak regulerende ecosysteemdiensten betreffen die in fysische eenheden uitgedrukt worden. Dit vraagt om een betere kwantificering van ecosysteemdiensten in socioculturele en economische waarden, zodat de relatie tussen ecosystemen, maatschappij, en economie meer recht wordt gedaan.

OV 3 wordt behandeld in Hoofdstukken 3 en 5. In Hoofdstuk 3 wordt het Natuurlijk Kapitaal Model (NC-Model) geïntroduceerd. Door toepassing van het NC-Model kunnen ecosysteemdiensten met een hoge ruimtelijke en thematische resolutie worden gekwantificeerd. Ruimtelijke representatie wordt verkregen door gebruik te maken van hoge resolutie (10 x 10 m) ruimtelijke data als input. Er zijn modellen voor het stedelijk gebied en voor de rurale omgeving. Thematische representatie wordt verkregen door het integreren van relaties tussen fysische en socio-economische variabelen binnen ecosysteemdienstmodellen. In Hoofdstuk 5 wordt een aanpak gepresenteerd voor het integreren van lokale kennis en data in modellen, met als doel de ruimtelijke kwantificering van lokale ecosysteemdiensten voor verschillende waarde-domeinen te ondersteunen. Het gebruik van lokale ruimtelijke data als input voor modellen bleek nuttig te zijn voor het in beeld brengen van vraag en aanbod met veel ruimtelijk detail. Het gebruik van lokale kennis en niet-ruimtelijke data als input bleek nuttig te zijn om thematische details in ruimte en tijd in kaart te brengen. Lokaal beschikbare kennis en data werden door gebruik van verschillende ruimtelijke kwantificeringsmethoden in modellen geïntegreerd. De geschiktheid van deze methoden verschilt als gevolg van casus-gerelateerde beschikbare middelen. De selectie van de voor het gebied relevante ecosysteemdiensten werd gebaseerd op de uitkomsten van semigestructureerde interviews met lokale stakeholders en door raadpleging van lokale informatiebronnen.

OV 4 wordt behandeld in Hoofdstuk 4. In dit Hoofdstuk wordt het NC-Model toegepast om het effect van veranderingen in de groene infrastructuur (GI) van de Gemeente Amsterdam op ecosysteemdiensten in te schatten. De analyse werd uitgevoerd om besluitvormers die betrokken zijn bij de *KwaliteitsImpuls Groen*, een ruimtelijk plan voor de uitbreiding en verbetering van de GI van Amsterdam tot en met 2025, te informeren. OV 4 wordt verder behandeld voor een ruraal gebied in Hoofdstuk 5. Lokale data en kennis van de gemeente Hoeksche Waard, een agrarisch gebied ten zuiden van Rotterdam, werden geïntegreerd om ecosysteemdiensten te kwantificeren. Zowel voor Amsterdam als de Hoeksche Waard bleek de kwantificering van ecosysteemdiensten op hoog ruimtelijk en thematisch detail mogelijk te zijn zodat zowel de ruimtelijke heterogeniteit van vraag en aanbod in beeld gebracht werden, als de factoren die de levering van ecosysteemdiensten beïnvloeden. Deze factoren omvatten een samenstel van informatie over de configuratie en typologie van natuurlijke elementen, alsmede de dichtheid en ruimtelijke verdeling van de bewonerscentra in Hoeksche Waard (Hoofdstuk 5) en Amsterdam (Hoofdstuk 4). De verdeling van inwoners is vaak een proxy voor de verdeling van begunstigden van ecosysteemdiensten en wordt vaak aan ecologische impact gekoppeld waardoor effecten van maatregelen expliciet kunnen worden gemaakt (zoals vervuiling in steden verminderen en de planvorming rond groene infrastructuur). De resultaten van dit onderzoek kunnen een belangrijke rol spelen bij de bewustmaking van besluitvormers van de complexe mechanismen die het leveringsproces onderbouwen. Door ruimtelijke planners bewust te maken van hoe ecosysteemdiensten en natuurlijk kapitaal het welzijn van de mens beïnvloeden, kunnen ze beheerstrategieën formuleren, prioriteren en ontwikkelen om lokale doelen te bereiken.

In Hoofdstuk 6 worden de belangrijkste bevindingen van dit proefschrift samengevoegd aan de hand van de vier OVs in eerdere hoofdstukken. Op basis hiervan worden de methodologische bijdragen van het onderzoek aan de wetenschap en de relevantie van het onderzoek voor het omgevingsbeleid en -beheer bediscussieerd. De benaderingen voor de ruimtelijke inschatting van ecosysteemdiensten met een hoog ruimtelijk en thematisch detail, en geankerd in de diverse waarde-domeinen zijn consistent en praktisch gezien werkend gemaakt. De theoretische en praktische voorbeelden toonden aan dat de verschillende benaderingen besluitvorming kunnen ondersteunen. Het kader waarin het proces van het leveren van ecosysteemdiensten geoperationaliseerd werd, kon effectief worden toegepast om vraag en aanbod van ecosysteemdiensten uit de literatuur en de twee in dit onderzoek uitgevoerde Nederlandse casestudies te kwantificeren. Door lokale ruimtelijke data en kennis te integreren werden ecosysteemdiensten ruimtelijk zichtbaar gemaakt en werd bruikbare informatie voor het beheer en beleid samengesteld zodat deze bij de besluitvorming toegepast kan worden, ongeacht achtergronden en belangen.

Het kwantificeren en in beeld brengen van de diensten die de natuur aan de mens levert kan een belangrijke rol (gaan) spelen bij de onderbouwing van ingrepen en investeringen in het natuurlijk kapitaal. Daarbij dienen niet alleen de kosten van investeringen en impact meegenomen te worden, maar ook de brede verscheidenheid van alle socioculturele en monetaire baten nu en in de toekomst. Op die wijze kan de maatschappij als geheel werken aan het onderhouden en eventueel herstel van het natuurlijk kapitaal, zodat aan het streven naar duurzaamheid invulling wordt gegeven. Dit proefschrift sluit af met een korte discussie over uitdagingen die in dit proefschrift niet onderzocht werden maar wel aandacht dienen te krijgen om de integratie van ecosysteemdiensten in besluitvorming verder te bevorderen.

GLOSSARY

Actual supply (of ecosystem services): Delivery component that captures ecological resources and processes that effectively benefit humans, also referred to as ecosystem functions. Expressed in biophysical units.

Benefits and values (of ecosystem services): Natural systems' contributions to human well-being, irrespective of whether these are perceived by their very receptors.

Cascade framework: Framework for operationalising the ecosystem service delivery process into delivery components (de Groot et al., 2002; Potschin & Haines-Young, 2016).

Cultural services: The non-material cultural benefits that ecosystems provide for humans (MA, 2005).

Data-based approach: Spatial quantification approach that integrates spatial and non-spatial data (e.g., from statistics, publications, field samples) to capture the relationship between biophysical and socioeconomic variables.

Delivery components: The ecological and socioeconomic building blocks that constitute the ecosystem service delivery process (Villamagna et al., 2013).

Delivery process (of ecosystem services): The process that underpins ecosystem service production and use, spanning across biophysical, sociocultural, and economic value domains (Villamagna et al., 2013).

Demand (for ecosystem services): Delivery component that captures human desires for ecosystem functions, as well as the actual benefits and values that they enjoy due to the current availability of ecosystem functions. Can be operationalised into two sub-components: ecosystem service (i) use and (ii) societal demand.

Ecosystem functions: Delivery component that captures ecological structures and processes that are beneficial for human well-being (de Groot et al., 2002).

Ecosystem services: The direct and indirect benefits that ecosystems provide to humans (Costanza et al., 2017; MA, 2005).

Expert elicitation approach: Mobilisation of experts to assign values to spatial and non-spatial variable categories (e.g., through ranking, rating, assigning, weights, photo elicitation) based on their knowledge (Martinez-Harms & Balvanera, 2012).

Natural capital: Earth's ecosystems and underpinning geophysical systems (Haines-Young & Potschin, 2018).

Operationalisation: The process by which concepts are made usable by practitioners (e.g., assessors, decision-makers; Potschin et al., 2014).

Potential supply (of ecosystem services): Delivery component that captures the range of ecological resources and processes that may potentially benefit humans, irrespective of whether they are actually used or valued by them (Tallis et al., 2012).

Provisioning services: The material contributions that natural capital endows for humans (MA, 2005).

Regulation and maintenance services: Ecological processes that directly or indirectly contribute to human well-being (MA, 2005).

Sections: Groups of ecosystem services that share similar characteristics, including provisioning, regulation and maintenance, and cultural services (https://cices.eu).

Social-ecological assessment models: Model the relationship among measurable biophysical and socioeconomic variables to spatially quantify ecosystem service proxy indicators, based on our mechanistic understanding of coupled human-natural systems (Martinez-Harms & Balvanera, 2012).

Societal demand (for ecosystem services): Delivery component that captures human desires for ecosystem services, irrespective of whether these are met by their actual supply and use (Wolff et al., 2015).

Stakeholders: Groups of people that influence (manage) or benefit from the provision of ecosystem services.

Supply: Delivery component that captures the full potential of ecological resources and processes to generate benefits that contribute to human well-being. Can be operationalised into two sub-components: ecosystem service (i) potential supply and (ii) actual supply.

Use (of ecosystem services): Delivery component that captures the socioeconomic benefits, expressed as sociocultural or monetary values, that humans receive from their use of ecosystem functions.

Value domains: Values commonly regarded under distinct disciplines, such as biophysical, economic, and sociocultural values.

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Five years ago, I embarked on an intricate journey to obtain a doctoral degree. During my first day working at RIVM, my promotor, Ton Breure, suggested we meet every Monday morning on a weekly basis and carry out all of our communication in Dutch. Starting my PhD studies while entirely communicating in a language so foreign to me was a strenuous task. As the months went by, our Monday meetings were joined by an increasing number of members. Soon, the Monday team was joined by my daily supervisors, Michiel Rutgers and Ton de Nijs, as well as my promotor, Jan Hendriks. Gradually, these virtually uninterrupted weekly meetings became not an obligation but a pleasure to me, where I was able to brainstorm in the company of four brilliant minds.

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CURRICULUM VITAE

Martina Paulin was born in Guayaquil, Ecuador, on April 3, 1990. She attended 'Colegio Menor San Francisco de Quito' in Quito, Ecuador, where she received her high school diploma in 2008. She moved to Stockholm, Sweden, in 2008, where she reconnected with her Swedish roots and studied the Swedish language. In 2009, she started her Bachelor of Science study in Economics at Jönköping University in Jönköping, Sweden. As part of her bachelor's studies, she did an exchange program at the School of Economics, Management, Accounting, and Actuarial Sciences (FEA) from the University of São Paulo (USP) in São Paulo, Brazil. During her time in São Paulo, she acquired the basic knowledge necessary to communicate in the Portuguese language. After graduation, in 2014, she started the master Environment and Resource Management at Vrije Universiteit (VU) in Amsterdam, the Netherlands. As part of her master's thesis, she worked for environmental consultancy firm Wolfs Company. In collaboration with the World Wildlife Fund for Nature (WWF), she and her peers conducted an ecosystem service assessment for one of Chile's largest salmon farming companies, formerly known as Los Fiordos. As part of her research, she travelled to various locations in Chile, where she interviewed dozens of stakeholders relevant to the company under assessment. To further expand her knowledge in the areas of sustainability and ecosystem services, she conducted internships at environmental consultancy firm CREM and at the Global Reporting Initiative (GRI), located in Amsterdam, the Netherlands. In March 2016, she started her PhD at the Dutch National Institute for Public Health and the Environment (RIVM) and Radboud University in the Netherlands. As part of her doctoral degree, she supervised various master's students during their thesis projects and presented in international conferences on ecosystem services and related fields. During her time working at RIVM, she performed support roles within the project Atlas of Natural Capital (ANK). Her contributions included the development of existing and new models from the Natural Capital Model (NC-Model), as well as attending and presenting in conferences and local government meetings. Throughout her years living in the Netherlands, she additionally learned to communicate in the Dutch language.

Publications

Peer-reviewed journal articles

Paulin, M.J., Remme, R.P., de Nijs, T., Rutgers, M., Koopman, K.R., de Knegt, B., Van der Hoek, D.C.J., & Breure, A.M. (2020). Application of the Natural Capital Model to assess changes in ecosystem services from changes in green. *Ecosystem Services*, 43, p. 101114. https://doi.org/10.1016/j.ecoser.2020.101114

Paulin, M.J., Remme, R.P., Van der Hoek, D.C.J., de Knegt, B., Koopman, K.R., Breure, A.M., Rutgers, M., & de Nijs, T. (2020). Towards nationally harmonized mapping and quantification of ecosystem services. *Science of the Total Environment*, 703, 134973. https://doi.org/10.1016/j.scitotenv.2019.134973

Paulin, M.J., Rutgers, M., de Nijs, T., Hendriks, A.J., Koopman, K.R., van Buul, T., Frambach, M., Sardano, G., & Breure, A.M., (2020). Integration of local knowledge and data for spatially quantifying ecosystem services in the Hoeksche Waard, the Netherlands. *Ecological Modelling*, 438, 109331. https://doi.org/10.1016/j.ecolmodel.2020.109331.

Paulin, M.J., Hendriks, A.J., Rutgers, M., de Nijs, T., Breure, A.M. Review: Ecosystem service operationalisation, conceptualisation, and mapping across value domains. *Ecosystem Services* (submitted).

Peer-reviewed reports

Paulin, M.J., Remme, R.P, & De Nijs, T. (2019). Amsterdam's Green Infrastructure: Valuing Nature's Contributions to People. RIVM Letter report 2019-0021. Bilthoven: National Institute for Public Health and the Environment (RIVM).

Remme, R.P., De Nijs, T., & Paulin, M.J. (2018). The Natural Capital Model: Technical documentation of the quantification, mapping and monetary valuation of urban ecosystem services. RIVM Report 2017-0040. Bilthoven: National Institute for Public Health and the Environment (RIVM).

Non-peer-reviewed reports

Luján Gallegos, V., Paulin, M.J., Steber, T., Wolfs, E., & van Beukering, P. (2016). *Business dependence on ecosystem services: How to identify risks and opportunities? An Ecosystem Services Review on Salmon Aquaculture in Chile.* Amsterdam: Wolfs Company.

C

Appendix

A

Appendix 1 - Supplementary material Chapter 2

Appendix 1 – 1

Table A1 - 1: Systematic literature selection process

Key word	Results
ABS("ecosystem services" OR "nature's contributions to people" OR "ecosystem goods and services" OR "ecological services")	26,579
AND ABS("map" OR "mapping" OR "mapped" OR "spatial distribution" OR "spatially-explicit" OR "spatially explicit")	3,158
AND (LIMIT-TO(PUBYEAR, 2019) OR LIMIT-TO(PUBYEAR, 2018) OR LIMIT-TO(PUBYEAR, 2017))	1,379
AND (LIMIT-TO(DOCTYPE, "ar")) AND (LIMIT-TO(PUBSTAGE, "final"))	1,173
Relevant articles	**111**

Appendix 1 – 2

A1 – 2.1 Systematically analysed sources

1 Adhikari, S., Baral, H., & Nitschke, C.R. (2018). Identification, prioritization and mapping of ecosystem services in the Panchase Mountain Ecological Region of Western Nepal. *Forests, 9*(9), 554.

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Appendix 1 – 3

A1 – 3.1 Input maps

This Section provides an overview of the input maps required for developing delivery component maps for the ecosystem service Air Quality Regulation. Most datasets are publicly available at standard international data repositories, such as the INSPIRE (https://inspire-geoportal.ec.europa.eu/) and ESRI geoportals (http://esrinl-content.maps.arcgis.com/), and governmental data registries (https://www.pdok.nl/; https://www.nationaalgeoregister.nl/; https://data.overheid.nl/). INSPIRE geoportal is the central European access point to the data provided by EU Member States and several EFTA countries under the INSPIRE Directive.

Table A1 - 2: Input datasets necessary to model delivery components for the ecosystem service Air Quality Regulation. CBS: Statistics Netherlands (CBS), RIVM: Rijksintituut Voor Volksgezondheid en Milieu, RWS: Rijkswaterstaat.

Original dataset name	Description	Resolution	Year	Source	Responsible organisation	Email
Actueel Hoogtebestand Nederland (AHN2)	Elevation map of the Netherlands	5 x 5 m	2007-2012	RWS	Stuurgroep AHN	servicedesk-data@rws.nl
Basisregistratie Adressen en Gebouwen (BAG)	Basic registry of addresses and buildings	2.5 x 2.5 m	2016	Kadaster (2019a)	Kadaster	PPB-GVA@kadaster.nl
Basisregistratie Gewaspercelen (BRP)	Agricultural areas of the Netherlands	25 x 25 m	2018	RWS	RWS	servicedesk-data@rws.nl
Ecosystem Unit Map (EUM)	Land use map of the Netherlands	10 x 10 m	2013	Van Leeuwen et al. (2017)	CBS	infoservice@cbs.nl
Fijnstof 2017 (pm10)	Concentration of particulate matter up to 10 micrograms (PM_{10})	25 x 25 m	2017	Velders et al. (2017)	RIVM	geodata@rivm.nl
Luchtfoto	High resolution aerial photograph of the Netherlands	0.25 x 0.25 m	2017	Beeldmateriaal Nederland	Beeldmateriaal Nederland	info@beeldmateriaal.nl
Population	Distribution of inhabitants in the Netherlands	10 x 10 m	2017	Remme et al. (2018)	RIVM	geodata@rivm.nl

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Actueel Hoogtebestand Nederland (AHN2)

The *Elevation ma of the Netherlands (AHN2)* is the second version of the digital elevation model of the Netherlands. It contains detailed and precise data containing elevation information for the Netherlands (relative to Dutch Ordinance Level - NAP). Elevation is measured through laser altimetry, a technique in which the earth's surface is scanned from an aircraft or helicopter with a laser beam. The measurement of the transit time of the laser reflection and of the position of the aircraft together give a very accurate result. Two types of elevation information were used, each from a separate TIF-file (raster files at a 0.5-metre resolution): ground-level elevation and the top-level elevation of all objects, relative to NAP.

Basisregistratie Adressen en Gebouwen (BAG)

The *Basic registry of addresses and buildings (BAG)* map contains information from the government system of basic registries. Data on addresses and buildings is collected by municipalities, which are also responsible for the quality of the data. Organisations with a public task, such as ministries, water boards, police forces and security regions, are obliged to use the authentic data from the registrations.

Basisregistratie Gewaspercelen (BRP)

The *Agricultural crop parcels (BRP)* map contains information on the distribution of agricultural plots along with the type of crops grown, selected by the Netherlands Enterprise Agency (*Rijksdienst voor Ondernemend Nederland*). The contours of agricultural plots come from the Agricultural Area Netherlands (AAN) dataset. Users of a parcel must annually indicate which type of crop is grown within the parcel. Datasets are generated yearly using May 15 as the reference date.

Ecosystem Unit Map (EUM)

Commissioned by the Ministry of Economic Affairs, the *Land use map of the Netherlands (EUM)* delineates ecosystem units in the Netherlands, categorised into six main themes; agriculture, dunes and beaches, forests and other (semi) natural and unpaved terrain, marshes and floodplains, water and paved and built-up land.

Fijnstof 2017 (pm10)

The *Concentration of particulate matter up to 10 micrograms, Fijnstof 2017 (pm10),* map contains information on the atmospheric concentrations of nitrogen dioxide and particulate matter in the Netherlands. Maps were developed for the National Air Quality Cooperation Program (NSL), a program to improve air quality in the Netherlands. The data on emissions and future scenarios also serve as a basis for monitoring the Nitrogen Approach Program (PAS). These programs test, among other things, the effects of spatial plans on the concentrations of pollutants in the air.

Luchtfoto

The *High resolution aerial photograph of the Netherlands, Luchtfoto,* map contains detailed aerial photos commissioned by the national government, provinces, and water boards at a 25 cm resolution.

Population

The *Population* map provides information on the distribution of inhabitants in the Netherlands, used as input for models in the NC-Model. Its reproduction can be achieved by following the methodology in Paulin et al. (2020b).

Appendix 1 – 4

A1 – 4.1: 'Air Quality Regulation' ecosystem service modelling specifications

This Section describes the process required for modelling ecosystem service supply and use indicators for the ecosystem service Air Quality Regulation presented in this study (Table A1 - 3). Four indicators were modelled by use of the Natural Capital Model (NC-Model). One indicator, PM_{10} concentration, was readily-available as a spatial dataset, developed by the Dutch National Institute for Public Health and the Environment (RIVM).

Table A1 - 3: Descriptions of delivery component indicators considered within example

Delivery component	Indicator	Description	Unit	Model/dataset	Source
Potential supply	Average deposition velocity	Average deposition velocity of PM_{10} adjusted based on the resuspension fraction	s/m^3	NC-Model	Paulin et al. (2020b)
Actual supply	PM_{10} retention	Reduction in atmospheric PM_{10} concentrations by vegetation and water	kg/yr	NC-Model	Paulin et al. (2020b)
Use	Reduced health costs	Reduction in health costs from avoided PM_{10} related mortalities	€/yr	NC-Model	Paulin et al. (2020b)
Societal demand	PM_{10} concentration	Atmospheric PM_{10} concentrations map	$\mu g/m^3/yr$	Fijnstof 2017 (pm10)	Velders et al. (2017)
Societal demand	Population density	Population map	Inhabitants/ha	NC-Model	Paulin et al. (2020b)

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Appendix 1 – 5

Table A1 - 4: Respective quantiles for legends of supply and demand maps in Figure 2.3

ES	Unit	Quantile 1	Quantile 2	Quantile 3	Quantile 4	Quantile 5
Average deposition velocity (adjusted)	s/m^3	0.00002 - 0.00053	0.00054 - 0.00097	0.00098 - 0.00099	0.00100 - 0.00134	0.00135 - 0.00496
PM_{10} retention	kg/100 m^2/year	0.001 - 0.033	0.034 - 0.053	0.054 - 0.059	0.06 - 0.071	0.072 - 0.340
Reduced health costs	€/100 m^2/year	6 - 146	147 - 233	234 - 256	257 - 309	310 - 1,485
PM_{10} concentration	µg/m^3/year	16.1 - 17.9	18.0 - 18.7	18.8 - 19.5	19.6 - 20.5	20.6 - 30.1
Population density	inhabitants/ha	0.0 - 1.2	1.3 - 1.8	1.9 - 2.7	2.8 - 3.9	4.0 - 75.9

Appendix 2 - Supplementary material Chapter 3

Appendix 2 – 1

A2 – 1.1: Input maps

This Section provides an overview of the input maps required for implementing models from the NC-Model described in in this paper. Most datasets are publicly available at standard international data repositories, such as the INSPIRE (https://inspire-geoportal.ec.europa.eu/) and ESRI geoportals (http://esrinl-content.maps.arcgis.com/), and all datasets can be found at governmental data registries or upon request. INSPIRE geoportal is the central European access point to the data provided by EU Member States and several EFTA countries under the INSPIRE Directive.

Table A2 - 1 : Input datasets necessary to model six urban ecosystem services. CBS: Statistics Netherlands, KNMI: Koninklijk Nederlands Meteorologisch Instituut, RIVM: Rijksinstituut voor Volksgezondheid en Milieu, RvO: Rijksdienst voor Ondernemend Nederland, RWS: Rijkswaterstaat.

Dataset name	Source	Responsible organisation	Email
Actueel Hoogtebestand Nederland (AHN2)	Stuurgroep AHN	RWS	servicedesk-data@rws.nl
Basisregistratie Adressen en Gebouwen (BAG)	Kadaster (2019a)	Kadaster	PPB-GVA@kadaster.nl
Basisregistratie Gewaspercelen (BRP)	RWS	RWS	servicedesk-data@rws.nl
Bevolkingskernen	CBS	CBS	infoservice@cbs.nl
Ecosystem Unit Map (EUM)	Van Leeuwen et al. (2017)	CBS	infoservice@cbs.nl
Fijnstof 2017 (pm10)	Velders et al. (2017)	RIVM	geodata@rivm.nl
Luchtfoto	Beeldmateriaal Nederland	Beeldmateriaal Nederland	info@beeldmateriaal.nl
Top10NL	CBS	Kadaster	PPB-GVA@kadaster.nl
Wijk- en buurtkaart	Bresters (2019)	CBS	infoservice@cbs.nl
Windsnelheden op 100m hoogte	KNMI	RvO	geodatabeheer.giscc@rvo.nl
WOZ-waarde	CBS	CBS	infoservice@cbs.nl

Actueel Hoogtebestand Nederland (AHN2)

The *Elevation map of the Netherlands (AHN2)* is the second version of the digital elevation model of the Netherlands. It contains detailed and precise data containing elevation information for the Netherlands (relative to Dutch Ordinance Level - NAP). Elevation is measured through laser altimetry, a technique in which the earth's surface is scanned from an aircraft or helicopter with a laser beam. The measurement of the transit time of the laser reflection and of the position of the aircraft together give a very accurate result. Two types of elevation information were used, each from a separate TIF-file (raster files at a 0.5-metre resolution): ground-level elevation and the top-level elevation of all objects, relative to NAP.

Basisregistratie Adressen en Gebouwen (BAG)

The *Basic registry of addresses and buildings (BAG)* map contains information from the government system of basic registries. Data on addresses and buildings is collected by municipalities, which are also responsible for the quality of the data. Organisations with a public task, such as ministries, water boards, police forces and security regions, are obliged to use the authentic data from the registrations.

Basisregistratie Gewaspercelen (BRP)

The *Agricultural crop parcels (BRP)* map contains information on the distribution of agricultural plots along with the type of crops grown, selected by the Netherlands Enterprise Agency (*Rijksdienst voor Ondernemend Nederland*). The contours of agricultural plots come from the Agricultural Area Netherlands (AAN) dataset. Users of a parcel must annually indicate which type of crop is grown within the parcel. Datasets are generated yearly using May 15 as the reference date.

Bevolkingskernen

The map of the *Contour of populated areas, Bevolkingskernen*, in the Netherlands contains the digital geometry of the contours of the population centres with key figures for 2011 in ESRI™ format.

Ecosystem Unit Map (EUM)

Commissioned by the Ministry of Economic Affairs, the *Land use map of the Netherlands (EUM)* delineates ecosystem units in the Netherlands, categorised into six main themes; agriculture, dunes and beaches, forests and other (semi) natural and unpaved terrain, marshes and floodplains, water and paved and built-up land.

Fijnstof 2017 (pm10)

The *Concentration of particulate matter up to 10 micrograms, Fijnstof 2017 (pm10),* map contains information on the atmospheric concentrations of nitrogen dioxide and particulate matter in the Netherlands. Maps were developed for the National Air Quality Cooperation Program (NSL), a program to improve air quality in the Netherlands. The data on emissions and future scenarios also serve as a basis for monitoring the Nitrogen Approach Program (PAS). These programs test, among other things, the effects of spatial plans on the concentrations of pollutants in the air.

Luchtfoto

The *High resolution aerial photograph of the Netherlands, Luchtfoto,* map contains detailed aerial photos commissioned by the national government, provinces, and water boards at a 25 cm resolution.

Top10NL

The *Topographic land use map of the Netherlands, TOP10NL*, is the basic digital topographic file of the Netherlands for the national registry. It is the most detailed product within the Basic Registration Topography (BRT) and can be used at scale levels between 1:5,000 and 1:25,000.

Wijk- en buurtkaart

The *Key neighbourhood statistics, Wijk- en buurtkaart,* map contains the digital geometry of the boundaries of neighbourhoods, districts, and municipalities. It includes the key figures of neighbourhoods and the aggregated key figures of the districts and municipalities. The *Key neighbourhood statistics* map comprises three sources: the municipal boundaries come from the Land Registry Basic Register (BRK); the neighbourhood boundaries are specified by municipalities, and the boundaries of the country including larger waters is based on Statistics Netherland's (CBS) Soil Use File.

Windsnelheden op 100m hoogte

The *Average windspeed at an elevation of 100 m, Gemiddelde windsnelheid 100m,* map shows the average wind speed in the Netherlands at 100 m elevation for the years 2004-2013. It is the operational working model for the Royal Dutch Meteorological Institute (KNMI), HARMONIE, which reconstructs hourly wind patterns. This is done at a horizontal resolution of 2.5 km. The average wind speed in HARMONIE has been compared with different measuring approaches throughout the Netherlands, a correction that is applied for the entire Netherlands. The inaccuracy of the average wind within 2.5 x 2.5 km grid cell is estimated at ± 0.3m/s.

WOZ-waarde

The *Real estate value, WOZ-Waarde*, map contains data on the value of all properties (WOZ objects), based on the Real Estate Valuation Act (WOZ). The figures are broken down according to the value of property and non-residential property objects and the average home value. The figures are hierarchically classified by country, province, COROP (Dutch regional division) area and municipality. Due to various social developments, the philosophy and method underlying the definition are no longer up to date. In addition, it appears that other authorities, depending on the area of application, use a different classification of metropolitan agglomerations and urban regions, so that it is no longer possible to speak of one standard.

Appendix 2 – 2

A2 – 2.1: ‘Air quality regulation’ model specifications

This Section describes the process required for modelling the indicators presented in this study for the ecosystem service ‘air quality regulation’, conform the NC-Model. The model is based on Remme et al. (2018), which is itself largely based on De Nocker et al. (2017). A schematic overview of the steps described hereunder is presented in Figure A2 - 1.

A2 – 2.1.1 Spatial data requirements

- Agricultural crop parcels (BRP)
- Basic registry of addresses and buildings (BAG)
- Concentration of particulate matter up to 10 micrograms (Fijnstof 2017, pm10)
- Elevation map of the Netherlands (AHN2)
- High resolution aerial photograph of the Netherlands (Luchtfoto)
- Land use map of the Netherlands (EUM)
- Key neighbourhood statistics (Wijk- en buurtkaart)

A2 – 2.1.2 Model procedure

A2 – 2.1.2.1 Creating vegetation layers (based on Remme et al., 2018)

1. In order to perform calculations in PCRaster, all input spatial data must be converted to raster format, the same resolution, and the same extent. To create vegetation layers, maps displaying the *High resolution aerial photograph of the Netherlands, Luchtfoto,* and the *Elevation map of the Netherlands, AHN2,* will remain at their original 0.5 m resolution. The *Basic registry of addresses and buildings, BAG,* map will be converted to a 0.5 meter resolution.

2. From the *AHN2* map, extract the raster layers showing the *Ground-level elevation of all objects (m), GLE,* and the *Top-level elevation of all objects (m), TLE.*

3. Create a layer displaying *Adjusted ground-level elevation of all objects (m),* GLE_{adj}, where cells containing no data within the *GLE* layer, meaning that buildings or other objects are found, are filled in as ground-level.

4. The ground level elevation is subtracted from the top elevation of all objects layer to obtain the *Total elevation of all objects, TE,* implementing Formula A2 - 1.

$$TE = TLE - GLE_{adj}$$

Formula A2 - 1

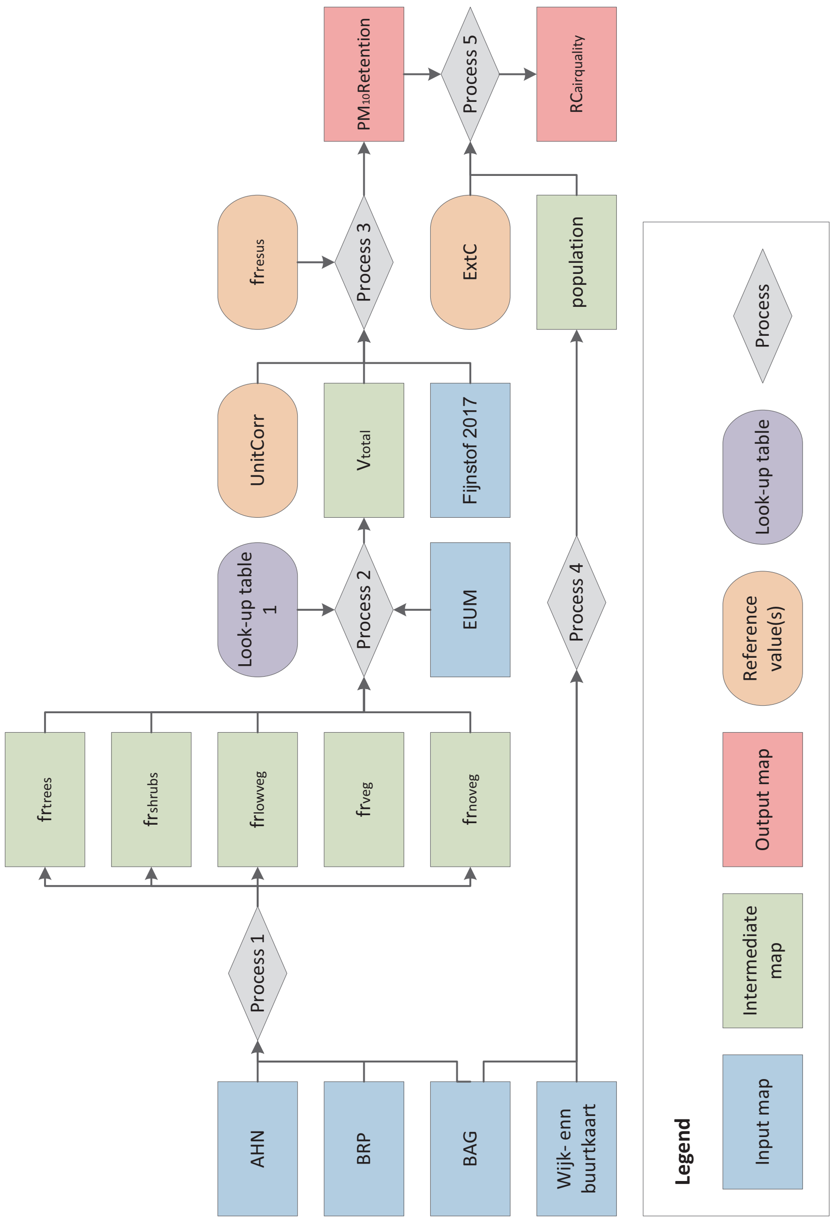

Figure A2 - 1: Urban model for the ecosystem service 'air quality' (adapted from Remme et al., 2018)

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5. Create a layer *BAG_{adj}* where all areas where buildings are found in *BAG* are assigned a value of 0 and all other cells are assigned a value of 1.

6. Remove buildings from the *TE* layer to create a layer containing the *Elevation of all objects excluding buildings, TE_{excl}*, by implementing Formula A2 - 2.

$$TE_{excl} = BAG_{adj} \times TE$$

Formula A2 - 2

7. Create a *Normalised Difference Vegetation Index, NDVI,* layer displaying the density of green vegetation, by distinguishing between vegetation and other objects. To do this, the *Infrared band, IRB,* and the *Red band*, *RB,* from the *Luchtfoto* map are combined into a 1-band *NDVI* layer, implementing Formula A2 - 3, based on Remme et al. (2018).

$$NDVI = (IRB - RB)/(IRB + RB)$$

Formula A2 - 3

8. Within the *NDVI* layer, all vegetated cells are assigned a value of 1. All other cells are assigned a value of 0.

9. By multiplying the layer displaying the distribution of all green vegetation with the layer displaying the height of all objects excluding buildings, a layer displaying the *Elevation of all green vegetation, Veg*, is created, following Formula A2 - 4.

$$Veg = NDVI \times TE_{excl}$$

Formula A2 - 4

10. Create a layer displaying the *Distribution of trees, Trees*, by assigning all vegetation taller than 2.5 meters within the *Veg* layer a value of 1 and all other cells a value of 0.

11. Create a layer displaying the *Distribution of bushes/shrubs, Shrubs*, by assigning all vegetation between 1 and 2.5 meters within the *Veg* layer a value of 1 and all other cells a value of 0.

12. Create a layer displaying the *Distribution of low vegetation, LowVeg*, by assigning all vegetation between 0 and 1 meters within the *Veg* layer a value of 1 and all other cells a value of 0.

13. All cells where agricultural areas are present in the *Agricultural crop parcels, BRP,* map are assigned a value of 0 within the *Trees, Shrubs,* and *LowVeg* layers.

14. Aggregate the *Trees* layer into a 10 m resolution layer showing the *Fraction per grid cell covered by trees (0-1), fr_{tree}.*

15. Aggregate the *Shrubs* layer into a 10 m resolution layer showing the *Fraction per grid cell covered by bushes/shrubs (0-1), fr_{shrub}.*

16. Aggregate the *LowVeg* layer into a 10 m resolution layer showing the *Fraction per grid cell covered by low vegetation (0-1), fr_{lowveg}.*

17. A layer displaying the *Fraction of a cell that is vegetated (0-1)*, fr_{veg}, is created by implementing Formula A2 - 5.

$$fr_{veg} = fr_{tree} + fr_{shrub} + fr_{lowveg}$$

Formula A2 - 5

18. A layer displaying the *Fraction of a cell that is not vegetated (0-1)*, fr_{noveg}, is created by implementing Formula A2 - 6.

$$fr_{noveg} = 1 - fr_{veg}$$

Formula A2 - 6

A2 – 2.1.2.2 Calculating the deposition velocity of PM_{10} for vegetation and water

19. Convert the *Land use map of the Netherlands, EUM*, to raster format and at a 10 m resolution.

20. Create a layer displaying the *Deposition velocities for trees (cm/s), DV_{tree}*. To do this, all cells where land cover types *coniferous forest* and *mixed forest* are present within the *EUM* map are assigned values of 0.7 and 0.6, respectively, conform Table A2 - 2 (De Nocker et al., 2017). Based on expert elicitation (scientists), all remaining cells where trees are present from the fr_{tree} layer, are assigned a value of 0.5.

Table A2 - 2: Deposition velocities for land cover types (based on De Nocker et al., 2017)

Vegetation type	**Deposition velocity (cm/s)**
Coniferous forest	0.7
Mixed forest	0.6
Deciduous forest	0.5
Shrubs	0.3
Meadows/grassland	0.2
Arable land	0.2
Low-stem orchard	0.2
Other low vegetation	0.2
Water	0.1
No vegetation	0 - 0.2

21. Create a layer displaying the *Total deposition velocity of vegetation in an area, DV_{total}*, implementing Formula A2 - 7,

$$DV_{total} = fr_{tree}DV_{tree} + fr_{shrub}DV_{shrub} + fr_{lowveg}DV_{lowveg} + fr_{noveg}DV_{noveg}$$

Formula A2 - 7

where the *Deposition velocity of shrubs/bushes,* DV_{shrub}, and the *Deposition velocity of low vegetation*, DV_{lowveg}, are 0.3m/s and 0.2m/s respectively, based on Table A2 - 2 (De Nocker et al., 2017). The *Deposition velocity in cells where no vegetation is present,* DV_{noveg}, is assigned a value of 0, since these cells do not contain any vegetation so PM_{10} retention by vegetation does not take place.

A2 – 2.1.2.3 Calculating PM_{10} retention by vegetation and water

22. Convert the *Concentration of particulate matter up to 10 micrograms (µg/m³/year)*, *Fijnstof 2017 (pm10),* map to a 10 m resolution.

23. Create a layer displaying *PM_{10} retention by vegetation and water (µg/year)*, *RET_{PM10}*, implementing Formula A2 - 8, based on De Nocker et al. (2017).

$$RET_{PM10} = DV_{total} \times C_{PM10} \times [1 - fr_{resus}] \times UnitCorr$$

Formula A2 - 8

where C_{PM10} is displays the *Concentration of particulate matter up to 10 micrograms, fr_{resus}* is the *Resuspension fraction from suspended particles (0-1)*, and *UnitCorr* is the *Unit correction value for translating units to kg/ha/yr (cm/s x µg/m³). C_{PM10}* is established based on the *Fijnstof 2017 (pm10)* dataset. Following De Nocker et al. (2017), fr_{resus} is equal to 0.5 for all land cover types and 0.0 for water, and *UnitCorr* is equal to 3.1536 (or 0.031536 per 100 m^2 cell).

A2 – 2.1.2.4 Creating the population layer (based on Remme et al., 2018)

24. A feature class that contains *All residential buildings in the Netherlands* along with an attribute that expresses the *Number of domestic housing units, HU,* is established based on the *BAG* dataset**.** A query is performed to select the domestic housing units that are currently in use.

25. A feature class that contains *Dutch neighbourhood statistics (including population statistics), NS,* is established based on the *Key neighbourhood statistics, Wijk- en buurtkaart,* dataset.

26. A spatial join is performed between *NS* and *HU* to create a feature class that contains the *Extended neighbourhood statistics,* NS_{ext}*.* In the new feature class, statistics displayed for each neighbourhood now include the number of domestic housing units.

27. A spatial join is performed between NS_{ext} and *HU* to create a feature class that contains the *Population per residential object (number of inhabitants/residential object),* POP_{RO}*.* To calculate POP_{RO} per residential object, Formula A2 - 9 is applied.

$$POP_{RO} = \left[\frac{HU_{RO}}{HU_{NEI}} \times POP_{NEI}\right]$$

Formula A2 - 9

*w*here HU_{RO} is the *Number of domestic housing units in each residential object* from the *HU* feature class, HU_{NEI} is the *Number of domestic housing units in each neighbourhood* from the NS_{ext} feature class, and POP_{NEI} is the *Population of each neighbourhood* from the NS_{ext} feature class.

28. Prior to converting POP_{RO} into a 10 m raster in order to continue with this model's calculations using PCRaster, the population value for each residential feature in POP_{RO} is divided by a *Factor,* which adjusts the population in each residential feature based on the number of 100 m^2 cells covered by it. This results in a feature class displaying an *Adjusted population,* POP_{adj}*,* layer following Formulas A2 - 10 and A2 - 11.

$$POP_{adj} = \frac{POP_{RO}}{Factor}$$

Formula A2 - 10

$$Factor = \frac{ARF}{100}$$

Formula A2 - 11

where *ARF* is the *Area of a specific residential feature (m*^{2}*).*

29. Convert POP_{adj} into a 10 m resolution raster layer displaying the *Dutch population,* POP_{RAS}.

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30. The rasterization of the feature class might lead to an over- or underestimation of the total population. To correct for this, apply Formula A2 - 12 and Formula A2 - 13 to create an *Adjusted final population, Population.* layer*.*

$$Population = POP_{RAS} \times Factor$$

Formula A2 - 12

$$Factor = \left(\frac{ACTPOP}{SUMPOP_{RAS}}\right)$$

Formula A2 - 13

where *Factor* is a *Correcting factor to adjust the total population in* POP_{RAS} to the actual Dutch population value, $SUMPOP_{RAS}$ is the *Sum of inhabitants in* POP_{RAS}, and *ACTPOP* is the actual Dutch population in 2017.

A2 – 2.1.2.5 Calculating the monetary benefits of PM_{10} retention by vegetation and water

31. Following Remme et al. (2018), the *Reduction in health costs from increased* PM_{10}*Retention (€/year)*, $RC_{airquality}$, is calculated, implementing Formula A2 - 14 and Formula A2 - 15.

$$RC_{airquality} = PM_{10}Retention \times ExtC$$

Formula A2 - 14

$$ExtC = 48.34 + 1.32 \times Population$$

Formula A2 - 15

where *ExtC* are the *External costs per inhabitant density associated with* PM_{10} *related diseases*. CE-Delft (2017) defined lower, central, and upper values of 31.80, 44.60, and 69.10 €/kg for the external costs of PM_{10} (based on 2015 € values), based on the ReCiPe methodology (Goedkoop et al., 2013). These external costs should be proportional to the population density (Künzli et al. 2000, CE-Delft, 2014), which is why the NC-Model (Remme et al., 2018) makes a difference in external costs for metropolitan, urban, and rural areas. In metropolitan areas, the external costs are 247.36 €/kg, in urban areas 79.76 €/kg, and in rural areas 48.34 €/kg [converted from 2010 to 2016 € values]. To correct for spatial discontinuities between metropolitan, urban, and rural areas, a linear relationship is assumed between changes in population density and changes in external costs (Remme et al., 2018; see Figure A2 - 2).

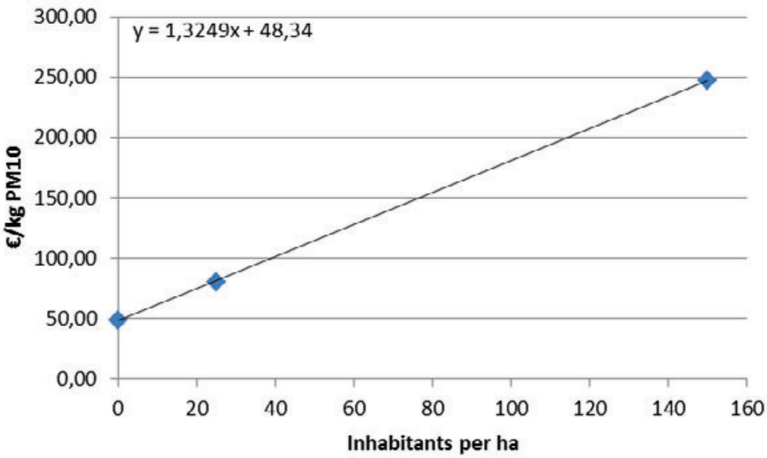

Figure A2 - 2: Linear relation between the inhabitant densities and external cost of PM_{10} (source: Remme et al., 2018). The blue points are estimates for average rural, urban and metropolitan population densities for the Netherlands (per ha).

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A2 – 2.2: ‘Physical activity’ model specifications

This Section describes the process required for modelling the indicators presented in this study for the ecosystem service ‘physical activity’, conform the NC-Model, based on Paulin et al. (2019). A schematic overview of the steps described hereunder is presented in Figure A2 - 3.

A2 – 2.2.1 Spatial data requirements

- Agricultural crop parcels (BRP)
- Basic registry of addresses and buildings (BAG)
- Elevation map of the Netherlands (AHN2)
- High resolution aerial photograph of the Netherlands (Luchtfoto)
- Key neighbourhood statistics (Wijk- en buurtkaart)

A2 – 2.2.2 Model procedure

A2 – 2.2.2.1 Creating vegetation layers

1. Vegetation layers at a 10 m resolution are created, conform Appendix 2 – 2 (Section A2 – 2.1.2.1), using the *AHN2, BAG*, *Luchtfoto,* and *BRP* layers as input. Vegetation layers include *Trees* (fr_{tree}), *Bushes/shrubs* (fr_{shrub}), and *Low vegetation* ($fr_{loweveg}$). Each layer shows the *Fraction per grid cell (0-1) covered by each vegetation type.*

A2 – 2.2.2.2 Creating the population layer

2. A *Population (number of inhabitants)* layer at a 10 m resolution is created conform Appendix 2 – 2 (Section A2 – 2.1.2.4), using the *BAG* and *Wijk- en buurtkaart* maps as input.

A2 – 2.2.2.3 Calculating the contribution to cycling for commuting purposes

A regression study conducted by Maas et al. (2008) conducted in the Netherlands (n = 4,899 people) found a positive relationship between the percentage of green space within a buffer (radius) of 1000 m from households and the number of minutes people cycle per year for commuting purposes. It found that, for every percentage increase in green space, people who cycle for commuting purposes will cycle 0.83 additional minutes on average. This relationship is illustrated in Figure A2 - 4. Maas et al. (2008) also found that, on average, people with 20% green space within a 1000 m radius around their home cycle 120 minutes per week for commuting purposes, whereas people with 80% green space within a 1000 m radius cycle approximately 170 minutes per week for commuting purposes. Based on this information, the intercept is estimated at 103.4 minutes per week per person.

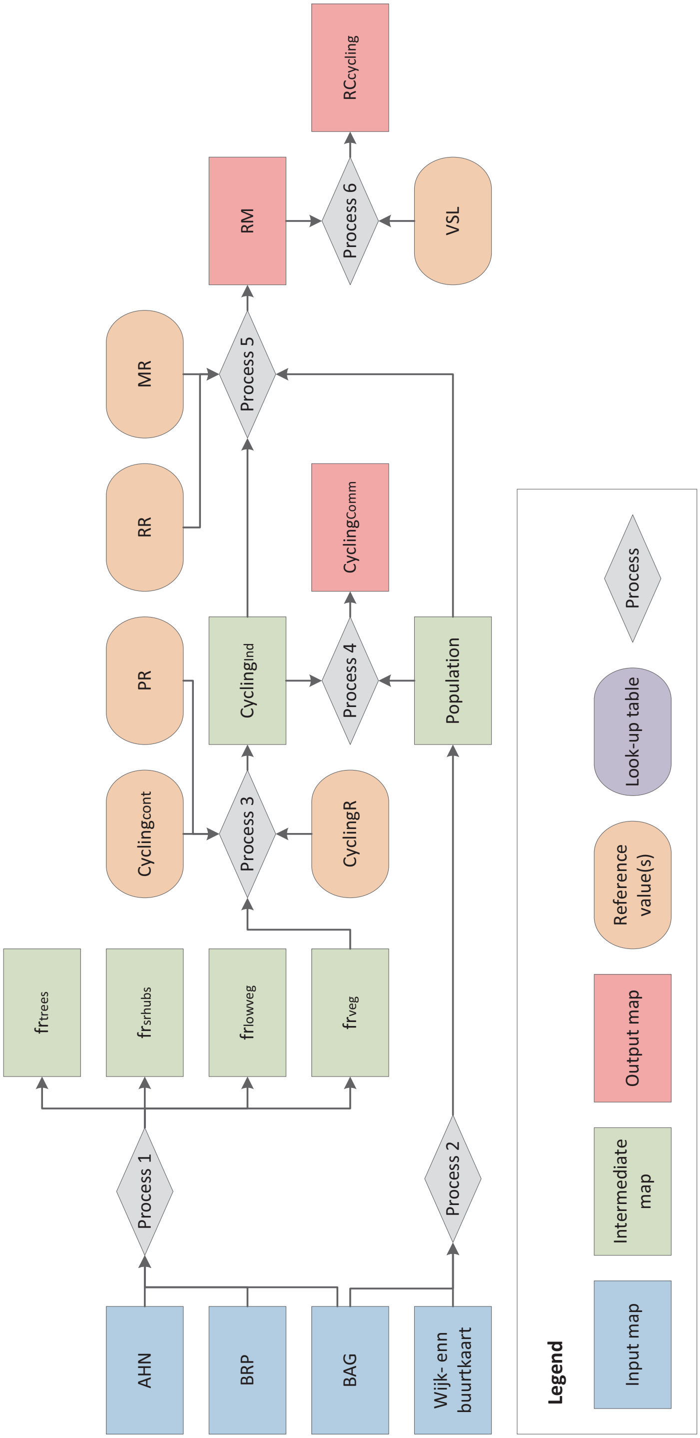

Figure A2 - 3: Urban model for the ecosystem service 'physical activity' (source: Paulin et al., 2019)

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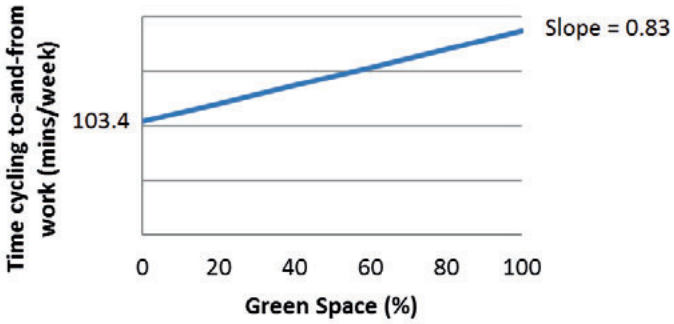

Figure A2 - 4: Relationship between the percentage of green space and the minutes people engage in cycling for commuting purposes (source: Paulin et al., 2019)

3. Using the fr_{veg} layer, calculate the *Percentage of green space within a 1km radius/buffer of a cell (0-100), $Green_{1km}$,* where green space refers to the percentage vegetation coverage within a specified area.

4. Apply Formula A2 - 16, formulated based on Maas et al. (2008), to calculate the *Hypothetical average contribution to cycling for commuting purposes by green space (min/individual/week), $Cycling_H$*.

$$Cycling_H = Cycling_{green} \times Green_{1km} \times PR \times CyclingR$$

Formula A2 - 16

where $Cycling_{green}$ captures the *Time cycled to-from work per week per individual for every percentage of green space within a 1000 m radius of a household*. *PR* represents the *Fraction of the population that participates in the labour market (0-1),* as commuting is specific to people participating in the labour market. Since not all commuting is done cycling, *CyclingR* represents the *Ratio of time spent cycling to-from work against the total time spent commuting by use of all necessary means of transportation within an urban area (0-1). $Cycling_{green}$* is equal to 0.83 minutes cycled per percentage of green space (Maas et al., 2008). In Amsterdam, *PR* was 0.63 and *CyclingR* was 0.23 in 2016 (https://data.amsterdam.nl/). *PR* and *CyclingR* will vary per urban area and across time. For this purpose, a look-up table with predefined *PR* and *CyclingR* for different urban areas can be generated, if desired.

6. Following Formula A2 - 17, create an *Actual contribution to cycling for commuting purposes by green space (minutes/week), $Cycling_A$*, layer by multiplying the population per cell times the hypothetical number of minutes cycled weekly to-from work per individual inhabiting a cell, due to the presence of green space.

$$Cycling_A = Cycling_H \times Population$$

Formula A2 - 17

A2 – 2.2.2.4 Calculating reduced mortality

For this part of the model, the model underlying the WHO's HEAT Tool (Kahlmeier et al., 2017) was adapted to determine the effect of cycling on human lifespans (reduced risk of all-cause mortality). The HEAT Tool uses well-established epidemiological relationships to quantify the relative risk of mortality among people who are exposed to cycling against the risk among people who are less exposed to cycling (Kahlmeier et al., 2017).

7. Following Formula A2 - 18, calculate the *Hypothetical reduced risk of mortality per individual due to the contribution to cycling by green space (ratio, 0-1), (RRM),* layer.

$$RRM = (Cycling_H / 100) \times (1 - \mathrm{RR})$$

Formula A2 - 18

where *RR* is the *Relative risk of mortality of cyclists against non-cyclists (ratio, 0-1)*. In the HEAT Tool, the *RR* is equal to 0.903 for a reference value of 100 minutes cycled per week per individual. This value accounts for the relative risks from air pollution while cycling. It is based on a meta-analysis of reduction in all-cause mortality from cycling by Kelly et al. (2014), which is itself based on six studies conducted in Europe and one in China (n = 187,000 individuals). Based on expert elicitation (scientists), RR is expected to increase or decrease proportionally based on the number of minutes cycled weekly per individual, based on a reference of 100 minutes. Following the HEAT Tool (Kahlmeier et al., 2017), a cap is set at a 45% risk reduction.

8. Calculate the *Actual average reduced number of mortalities from increased cycling to-from work (individuals/year)*, *RM*, layer, following Formula A2 - 19.

$$RM = (MR \times Population) - [MR(1 - RRM) \times Population]$$

$$RM = MR \times RRM \times Population$$

Formula A2 - 19

where *MR* is the *Mortality rate (0-1)*, which in Amsterdam was equal to 0.6% (0.006) in 2016 (https://www.allecijfers.nl). As *MR* varies per urban area, a look-up table can be generated with predefined *MR* values for different urban areas, if desired.

A2 – 2.2.2.5 Calculating reduced costs from reduced mortality

For this part of the model, the model underlying the WHO's HEAT Tool (Kahlmeier et al., 2017) was adapted in order to estimate the economic gains from reduced all-cause mortality that would result from increased cycling due to the presence of green space in an area. To determine the economic value of reduced mortalities, the HEAT Tool multiplies the number of reduced mortalities by the value of a statistical life (VSL). The VSL captures an aggregation of individual values for small changes in risk of death.

9. Calculate the *Reduced costs from reduced mortalities (€/year)*, $RC_{cycling}$, following Formula A2 - 20.

$$RC_{cycling} = RM \times VSL$$

Formula A2 - 20

where *VSL* is the *Value of a statistical life (€/life).* The VSL used by the HEAT Tool is based on a comprehensive review of VSL studies published by the Organisation for Economic Co-operation and Development (OECD) in 2012. Following the HEAT Tool, the *VSL* is equal to €2.132 million (European default values for 2015 for the WHO European Region; Kahlmeier et al., 2017).

A2 – 2.3: 'Property value' model specifications

This Section describes the process required for modelling the indicators presented in this study for the ecosystem service 'property value', conform the NC-Model, based on Remme et al. (2018). A schematic overview of the steps described hereunder is presented in Figure A2 - 5.

A2 – 2.3.1 Spatial data requirements

- Agricultural crop parcels (BRP)
- Basic registry of addresses and buildings (BAG)
- Elevation map of the Netherlands (AHN2)
- High resolution aerial photograph of the Netherlands (Luchtfoto)
- Key neighbourhood statistics (Wijk- en buurtkaart)
- Real estate value (WOZ-Waarde)

A2 – 2.3.2 Model procedure

A2 – 2.3.2.1 Creating vegetation layers

1. Vegetation layers at a 10 m resolution are created, conform Appendix 2 – 2 (Section A2 – 2.1.2.1), using the *AHN2, BAG*, *Luchtfoto,* and *BRP* layers as input. Vegetation layers include *Trees* (fr_{tree}), *Bushes/shrubs* (fr_{shrub}), and *Low vegetation* ($fr_{loweveg}$). Each layer shows the *Fraction per grid cell (0-1) covered by each vegetation type.*

A2 – 2.3.2.2 Creating the population layer

2. A *Population (number of inhabitants)* layer at a 10 m resolution is created conform Appendix 2 – 2 (Section A2 – 2.1.2.4), using the *BAG* and *Wijk- en buurtkaart* maps as input.

A2 – 2.3.2.3 Creating the property value layer

3. A feature class that displays the distribution of *All residential buildings in the Netherlands, RB,* along with an attribute that expresses the *Number of domestic housing units per residential feature, HU,* is established based on the *Basic registry of addresses and buildings, BAG,* dataset.

4. A feature class that contains the *Average property value per neighbourhood (€),* PV_{NEI}, is established based on the *Real estate value, WOZ-Waarde,* map.

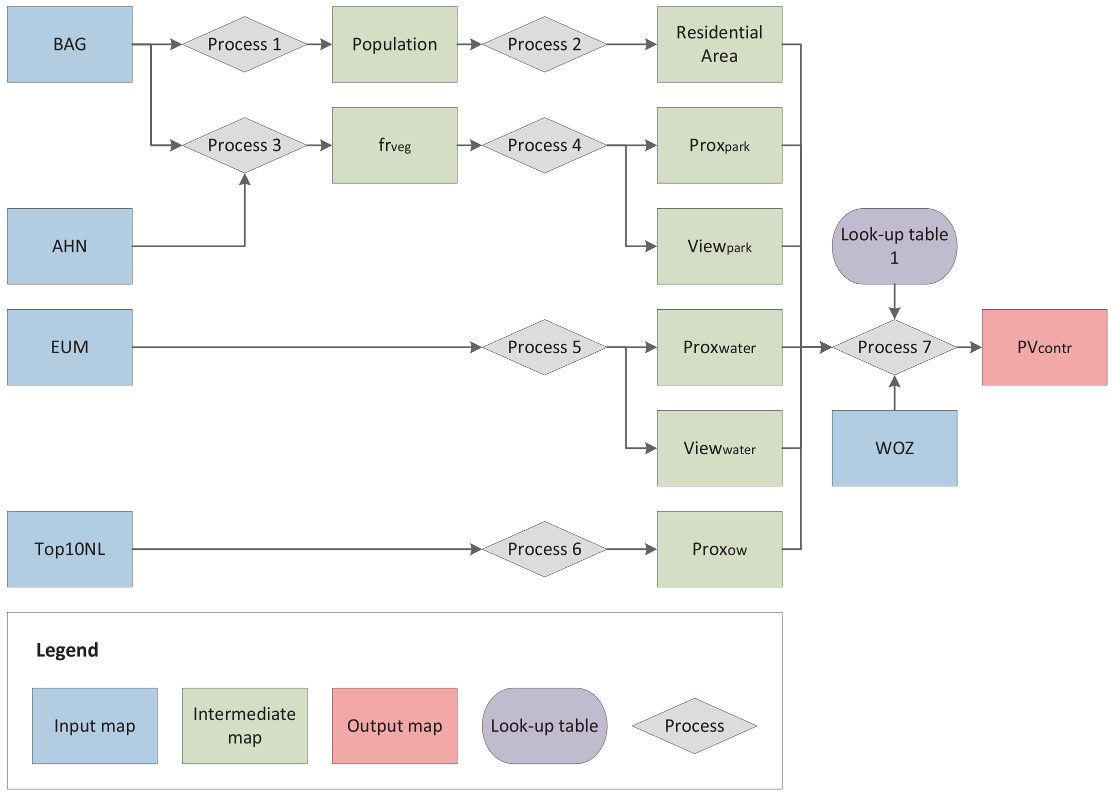

Figure A2 - 5: Urban model for the ecosystem service 'Property value' (adapted from Remme et al., 2018)

5. Following Formula A2 – 21, create an *Aggregate value of all properties per residential feature (€), PV,* layer. This is done by multiplying the number of housing units in a residential feature by the average property value in the neighbourhood where it is located.

$$PV = HU \times PV_{NEI}$$

Formula A2 - 21

6. Prior to converting *PV* into a 10 m raster in order to continue with this model's calculations using PCRaster, the property value for each residential feature in PV is divided by a *Factor.* This factor adjusts the property value of each residential feature based on the number of 100 m^2 cells covered by it. This results in a feature class displaying an *Adjusted value of all properties in a residential feature (€),* PV_{adj}, layer following Formulas A2 - 22 and A2 - 23.

$$PV_{adj} = \frac{PV}{Factor}$$

Formula A2 - 22

$$Factor = \frac{ARF}{100}$$

Formula A2 - 23

where *ARF* is the *Area of a specific residential feature* (m^2).

7. Convert PV_{adj} into a 10 m resolution raster layer displaying the *Property values of all residential objects (raster),* PV_{RAS}.

8. The rasterization of the feature class might lead to an over- or underestimation of total property values in a neighbourhood. To correct for this, apply Formula A2 - 24 and Formula A2 - 25 to create a *Total value of all properties (€),* PV_{TOTAL}, layer.

A

$$PV_{TOTAL} = PV_{RAS} \times Factor$$

Formula A2 - 24

$$Factor = \left(\frac{ACTPV}{SUMPV_{RAS}}\right)$$

Formula A2 - 25

where *Factor* is a *Correcting factor to adjust the property values in* PV_{RAS} to the aggregate value of properties in a neighbourhood, $SUMPV_{RAS}$ is the *Sum of the values of all properties in* PV_{RAS}, and *SUMPV* is the *Sum of the values of all properties in PV*, prior rasterization of the layer into PV_{RAS}.

A2 – 2.3.2.4 Calculating the proximity to and view of parks

9. Create a *Parks* layer displaying the *Distribution of parks*, by considering areas from the fr_{veg} map that are larger than one ha in size and that contain more than 60% vegetation. Cells falling under areas considered as parks are assigned a value of 1, while all other cells are assigned a value of 0.

10. Create a *Proximity to parks,* $Prox_{parks}$, layer, where cells containing residential areas (value >1 in the PV_{Total} layer) that contain at least one cell within a 400 m distance from the *Parks* layer, are assigned a value of 1. All other cells are assigned a value of 0.

11. Create a *View of parks,* $View_{park}$, layer, where cells containing residential areas (value >1 in the PV_{Total} layer) that contain at least one cell within a 30 m distance from the *Parks* layer, are assigned a value of 1. All other cells are assigned a value of 0.

A2 – 2.3.2.5 Calculating the proximity to and view of water

12. A *Water* layer, displaying the *Distribution of water,* is created by assigning a value of 1 to areas as small as 100 m^2 (one cell) in the *EUM* map that contain water. All other cells are assigned a value of 0.

13. A *Proximity to water,* $Prox_{water}$, layer is created, where cells containing residential areas (value >1 in the PV_{Total} layer) that contain at least one cell within a 400 m distance from the *Water* layer, are assigned a value of 1. All other cells are assigned a value of 0.

14. A *View of water,* $View_{water}$, layer is created, where cells containing residential areas (value >1 in the PV_{Total} layer) that contain at least one cell within a 30 m distance from the *Water* layer, are assigned a value of 1. All other cells are assigned a value of 0.

A2 – 2.3.2.6 Calculating the proximity to open water

15. An *Open water*, *OW*, layer is created, considering areas from the *Topographic land use map of the Netherlands, Top10NL,* map that are larger than one ha in size from the *open water* category. Cells falling under areas considered as *open water* are assigned a value of 1, while all other cells are assigned a value of 0.

16. A *Proximity to open water,* $Prox_{OW}$, layer is created, where cells containing residential areas (value >1 in the PV_{Total} layer) that contain at least one cell within a 50 m distance from the *OW* layer, are assigned a value of 1. All other cells are assigned a value of 0.

A2 – 2.3.2.7 Calculating the contribution to property value by green and blue

A regression study by Luttik & Zijlstra (1997) evaluated the influence of different natural element typologies (e.g., park, water, tree line) on property prices. Based on Luttik & Zijlstra (1997), Table A2 - 3 displays the contribution of different natural element typologies and configurations to property value.

Table A2 - 3: Contribution of vegetation and water to property value (0-1) (sources: Luttik & Zijlstra, 1997; Ruijgrok & De Groot, 2006)

Natural element typology/configuration	Fraction contribution to property value
View of a park or water (*fr_{view}*)	0.08
Proximity to a park or water (*fr_{prox}*)	0.06
Proximity to open water (*fr_{OW}*)	0.12

17. A *View of a park or water, $View_{PW}$,* layer is created. This is done by assigning a value of 1 to all cells where the layers $View_{park}$ or $View_{water}$ are equal to 1. All other cells are assigned a value of 0.

18. Calculate the *Contribution to property value based on the view of a park or water (€), PV_{view},* implementing Formula A2 - 26.

$$PV_{View} = PV_{total} \times fr_{view} \times View_{PW}$$

Formula A2 - 26

where *fr_{view}* is the *Fraction of property value determined by whether a residence has a view of parks or water (0-1).* Based on Table A2 - 3, *fr_{view}* is assigned a value of 0.08.

19. A *Proximity to a park or water, $Prox_{PW}$,* layer is created. This is done by assigning a value of 1 to all cells where the layers $Prox_{park}$ or $Prox_{water}$ are equal to 1. All other cells are assigned a value of 0.

20. Calculate the *Contribution to property value based on the proximity to a park or water (€), PV_{PP},* implementing Formula A2 - 27.

$$PV_{prox} = PV_{total} \times fr_{prox} \times Prox_{PW}$$

Formula A2 - 27

where *fr_{prox}* is the *Fraction of property value determined by whether a residence is close to parks or water (0-1).* Based on Table A2 - 3, *fr_{prox}* is assigned a value of 0.06.

21. Calculate the *Contribution to property value based on the proximity to open water (€), PV_{OW},* implementing Formula A2 - 28.

$$PV_{OW} = PV_{total} \times fr_{OW} \times Prox_{OW}$$

Formula A2 - 28

where fr_{OW} is the *Fraction of property value determined by whether a residence is close to open water (0-1).* Based on Table A2 - 3, fr_{OW} is assigned a value of 0.12.

22. The presence of multiple natural element typologies and configurations on property values is not accounted for, which is why, for each cell, the highest fraction increase is considered. Hence, a layer displaying the *Contribution to property value by vegetation and water (€), PV_{contr},* is created by performing an overlay containing
    - all positive values from the PV_{OW} layer
    - where PV_{OW} is 0, all positive values from the PV_{view} layer
    - where PV_{OW} and PV_{view} are 0, all positive values from the PV_{prox} layer

A2 – 2.4: 'Urban cooling' model specifications

This Section describes the process required for modelling the indicators presented in this study for the ecosystem service 'urban cooling', conform the NC-Model, based on Remme et al. (2018). A schematic overview of the steps described hereunder is presented in Figure A2 - 6.

A2 – 2.4.1 Spatial data requirements

- Agricultural crop parcels (BRP)
- Average windspeed at an elevation of 100 m (Windsnelheden op 100m hoogte)
- Basic registry of addresses and buildings (BAG)
- Elevation map of the Netherlands (AHN2)
- High resolution aerial photograph of the Netherlands (Luchtfoto)
- Key neighbourhood statistics (Wijk- en buurtkaart)
- Land use map of the Netherlands (EUM)

A2 – 2.4.2 Model procedure

A2 – 2.4.2.1 Creating vegetation layers

1. Vegetation layers at a 10 m resolution are created, conform Appendix 2 – 2 (Section A2 – 2.1.2.1), using the *AHN2, BAG*, *Luchtfoto,* and *BRP* layers as input. Vegetation layers include *Trees* (fr_{tree}), *Bushes/shrubs* (fr_{shrub}), and *Low vegetation* ($fr_{loweveg}$). Each layer shows the *Fraction per grid cell (0-1) covered by each vegetation type.*

A2 – 2.4.2.2 Creating the population layer

2. A *Population (number of inhabitants)* layer at a 10 m resolution is created conform Appendix 3 – 2 (Section A2 – 2.1.2.4), using the *BAG* and *Wijk- en buurtkaart* maps as input.

A2 – 2.4.2.3 Calculating the maximum UHI effect

3. Create a layer displaying the *Roughness length for momentum* value*, z0m,* for each land cover type within the *Land use map of the Netherlands, EUM,* raster and for different green vegetation types (fr_{tree}, fr_{shrub}, fr_{lowveg}). The value for *z0m* represents the height at which the wind speed theoretically becomes zero for the given land cover type. The *z0m* for each land cover type is assigned based on Table A2 - 4, which is based on Wieringa (1986).

A

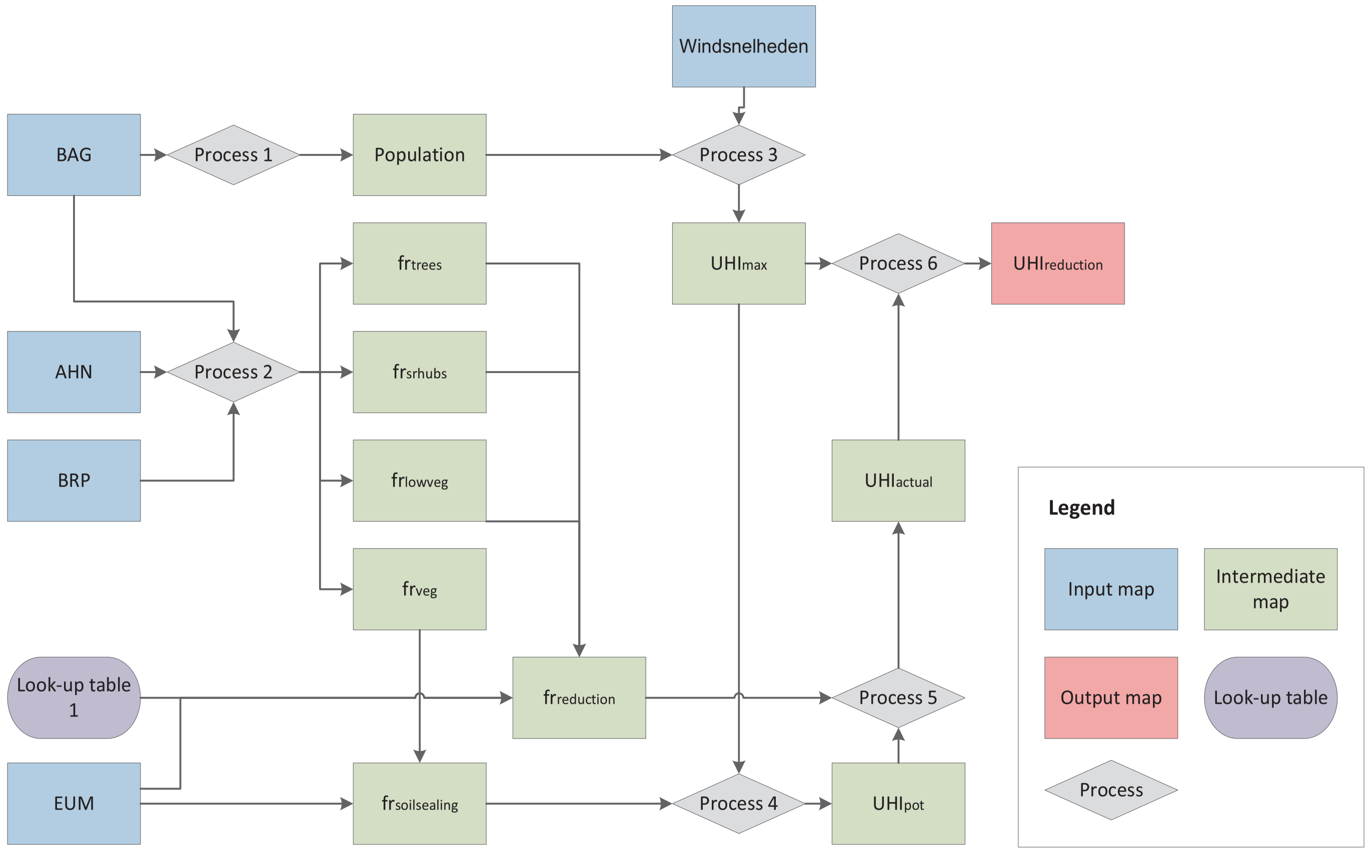

Figure A2 - 6: Urban model for the ecosystem service 'Urban Cooling' (adapted from Remme et al., 2018)

Table A2 - 4: Roughness length for momentum, z0m, for different types of land cover (Wieringa, 1986)

Land cover category	z0m
Open water (at least 5 km^2)	0.0002
No vegetation, no obstacles	0.005
Low vegetation	0.03
Low crops	0.10
High crops, scattered obstacles	0.25
Bushes/shrubs, numerous obstacles	0.5
Trees, forests, large obstacles	1

4. The *Average wind speed at an elevation of 100 m*, WS_{100}, map shows the *Average wind speed at 100 m above ground level (m/s^{-1})*. To downscale it to a map displaying the *Average wind speed at 10 m above ground level (m/s^{-1})*, $WS_{10,}$ Formula A2 - 29 (Wieringa, 1986), is applied.

$$WS_{10m} = WS_{100m} \times \ln(10/z0m)/\ln(100/z0m)$$

Formula A2 - 29

5. WS_{10m} is smoothed out by calculating the value of the average wind speed in a 50 m radius around a given cell and applying this value to the cell.

6. Create a layer displaying the *Population within a 10 km radius of a cell (number of inhabitants), Population*$_{10km,}$ using the *Population* map as input.

7. Create a layer displaying the *Maximum UHI effect that can occur in an area (degrees C), UHI*$_{max}$, by implementing Formula A2 - 30. The equation is based on the UrbClim model that was validated and used during the EU FP7 project RAMSES for 100 European cities (De Ridder et al., 2015; Lauwaet et al., 2015; Lauwaet et al., 2016). Results from the RAMSES project reveal that the maximum UHI effect in an area can be estimated based on (i) the annual average wind speed at 10 m above the ground and (ii) the population size within a 10 km radius.

$$UHI_{max} = -1.605 + 1.062 \log(Population_{10km}) - 0.356(WS_{10m})$$

Formula A2 - 30

A

A2 – 2.4.2.4 Calculating the potential UHI effect

The UHI effect only takes place in built-up areas. Hence, it is necessary to determine the amount of soil-sealing in an area to determine the potential UHI effect that can occur.

9. A new layer displaying the *Fraction of soil sealing within a 1 km radius (0-1), fr_{sealed}*, is created. This is done by assigning a value of 1 to cells containing built-up areas within *EUM* and subtracting the vegetated fraction of the cell (0-1) based on the fr_{veg} map. All other cells are assigned a value of 0. Subsequently, the value of each cell is recalculated, capturing the percentage (fraction) of soil sealing within a 1 km radius of the cell (0-1).
10. Create a layer displaying the *Potential UHI effect (degrees C), UHI_{pot}*, which reflects the UHI effect that can be expected in a cell based on the fraction of soil sealing in a 1 km radius of it (0-1). This is done by implementing Formula A2 - 31, which is based on Remme et al. (2018). Only areas with at least 20% (0.2) sealing within a 1 km radius are considered, which is why, prior to implementing Formula A2 - 31, all cells in fr_{sealed} with a value lower than 0.2 are assigned a value of 0.

$$UHI_{pot} = UHI_{max} \times fr_{sealed}$$

Formula A2 - 31

A2 – 2.4.2.5 Calculating the actual UHI effect

Based on expert elicitation (Remme et al., 2018), Table A2 - 5 displays the fraction-reduction of the UHI effect by vegetation and water land cover types from the *EUM* map. Also based on expert elicitation, Table A2 - 6 displays the fraction-reduction of the UHI effect by *Trees (fr_{tree}), Shrubs/bushes (fr_{shrub}),* and *Low vegetation (fr_{lowveg}).* Since vegetation and water can have a cooling effect throughout their surroundings, vegetation and water are assumed to have a cooling effect within a distance of 30 m from a cell, based on Remme et al. (2018), which is in turn based on expert judgment.

Table A2 - 5: Reduction of UHI effect by EMU land cover classes based on expert judgement (Remme et al., 2018)

Land cover or vegetation type	Reduction fraction (0-1)
Sea	1.0
(Semi)natural vegetation	0.2
Agricultural land	0.15 - 0.30
Bare soil	0
Built-up area	0
Inland water	0.30

Table A2 - 6: Reduction of UHI effect by vegetation classes based on expert judgement (Remme et al., 2018)

Land cover or vegetation type	Reduction fraction (0-1)
Trees	0.5
Bushes/shrubs	0.3
Low vegetation	0.2

11. Create a layer displaying *Semi-natural elements, EMU_{semi},* where all cells in the *EMU* map where semi-natural elements are found are assigned a value of 1. All other cells are assigned a value 0.

12. Create a layer displaying the *Fraction reduction of the UHI effect by semi-natural vegetation (0-1), fr_{semi}.* This is done by calculating, for each cell, the *Fraction coverage of EMU_{semi} within a 30 m radius of a cell (0-1)* and multiplying that value by the UHI-effect fraction-reduction value assigned to semi-natural vegetation (0.2), based on Table A2 - 5. All other cells are assigned a value 0.
13. Create a layer displaying *Agricultural areas, EMU_{agri},* where all cells in the *EMU* map where agricultural land is found are assigned a value of 1. All other cells are assigned a value 0.
14. Create a layer displaying the *Fraction reduction of the UHI effect by agricultural areas (0-1), fr_{agri}.* This is done by calculating the *Fraction coverage of EMU_{agri} within a 30 m radius of a cell (0-1)* and multiplying that value by the UHI-effect fraction-reduction value assigned to agricultural areas (0.15), based on Table A2 - 5. All other cells are assigned a value 0.
15. Create a layer displaying *Inland water, EMU_{water},* where all cells in the *EMU* map where inland water is found are assigned a value of 1. All other cells are assigned a value 0.
16. Create a layer displaying the *Fraction reduction of the UHI effect by inland water (0-1), fr_{water}.* This is done by calculating the *Fraction coverage of EMU_{water} within a 30 m radius of a cell (0-1)* and multiplying that value by the UHI-effect fraction-reduction value assigned to inland water (0.3), based on Table A2 - 5. All other cells are assigned a value 0.
17. Create a layer displaying *Sea land cover, EMU_{sea},* where all cells in the *EMU* map where sea is present are assigned a value of 1. All other cells are assigned a value 0.
18. Create a layer displaying the *Fraction reduction of the UHI effect by sea (0-1), fr_{sea}.* This is done by calculating the *Fraction coverage of EMU_{sea} within a 30 m radius of a cell (0-1)* and multiplying that value by the UHI-effect fraction-reduction value assigned to sea (1), based on Table A2 - 5. All other cells are assigned a value 0.
19. To avoid double-counting of vegetation types within individual cells in the following calculations, all cells in the fr_{tree}, fr_{shrub}, and fr_{lowveg} maps where the value of EMU_{semi}, EMU_{agri}, EMU_{water}, or EMU_{sea} is equal to 1, are assigned a value of 0.
20. Create a layer displaying the *Fraction reduction of the UHI effect by trees (0-1), fr_{RT}.* This is done by calculating the *Fraction coverage of fr_{tree} within a 30 m radius of a cell (0-1)* and multiplying that value by the UHI-effect fraction-reduction value assigned to trees (0.5), based on Table A2 - 6. All other cells are assigned a value 0.
21. Create a layer displaying the *Fraction reduction of the UHI effect by shrubs/bushes (0-1), fr_{RBS}.* This is done by calculating the *Fraction coverage of fr_{shrub} within a 30 m radius of a cell (0-1)* and multiplying that value by the UHI-effect fraction-reduction value assigned to shrubs/bushes (0.3), based on Table A2 - 6. All other cells are assigned a value 0.

22. Create a layer displaying the *Fraction reduction of the UHI effect by low vegetation (0-1), fr_{RLV}.* This is done by calculating the *Fraction coverage of fr_{lowveg} within a 30 m radius of a cell (0-1)* and multiplying that value by the UHI-effect fraction-reduction value assigned to low vegetation (0.2), based on Table A2 - 6. All other cells are assigned a value 0.
23. Create a layer displaying the *Actual UHI effect within a cell (degrees C), UHI_{actual},* by implementing Formula A2 - 32, based on Remme et al. (2018), which is itself based on expert judgment.

A

$$UHI_{actual} = UHI_{pot} \times (1 - fr_{semi} - fr_{agri} - fr_{water} - fr_{sea} - fr_{RT} - fr_{RBS} - fr_{RLS})$$

Formula A2 - 32

A2 – 2.4.2.6 Calculating the reduction of the UHI effect by green and water

24. Calculate the *Reduction of the UHI effect by green and water (degrees C), $UHI_{reduction}$*, by implementing Formula A2 - 33, based on Remme et al. (2018), which is itself based on expert judgment.

$$UHI_{reduction} = UHI_{max} - UHI_{actual}$$

Formula A2 - 33

A2 – 2.5: 'Urban health' model specifications

This Section describes the process required for modelling the indicators presented in this study for the ecosystem service 'urban health', conform the NC-Model, based on Remme et al. (2018). The approach builds on the TEEB-Stad methodology, using the same input values as the TEEB-Stad tool (www.teebstad.nl). A schematic overview of the steps described hereunder is presented in Figure A2 - 7.

A2 – 2.5.1 Spatial data requirements

- Agricultural crop parcels (BRP)
- Basic registry of addresses and buildings (BAG)
- Elevation map of the Netherlands (AHN2)
- High resolution aerial photograph of the Netherlands (Luchtfoto)
- Key neighbourhood statistics (Wijk- en buurtkaart)

A2 – 2.5.2 Model procedure

A2 – 2.5.2.1 Creating vegetation layers

1. Vegetation layers at a 10 m resolution are created, conform Appendix 2 – 2 (Section A2 – 2.1.2.1), using the *AHN2, BAG*, *Luchtfoto,* and *BRP* layers as input. Vegetation layers include *Trees* (fr_{tree}), *Bushes/shrubs* (fr_{shrub}), and *Low vegetation* ($fr_{loweveg}$). Each layer shows the *Fraction per grid cell (0-1) covered by each vegetation type.*

A2 – 2.5.2.2 Creating the population layer

2. A *Population (number of inhabitants)* layer at a 10 m resolution is created conform Appendix 2 – 2 (Section A2 – 2.1.2.4), using the *BAG* and *Wijk- en buurtkaart* maps as input.

A2 – 2.5.2.3 Calculating the reduced number of visits to the doctor due to urban green

3. Using the fr_{veg} layer, calculate the *Percentage of green space within a 1km buffer (radius) of a cell (0-1),* $Green_{1km}$.

4. Create a layer displaying the Population within a 1 km radius of a cell

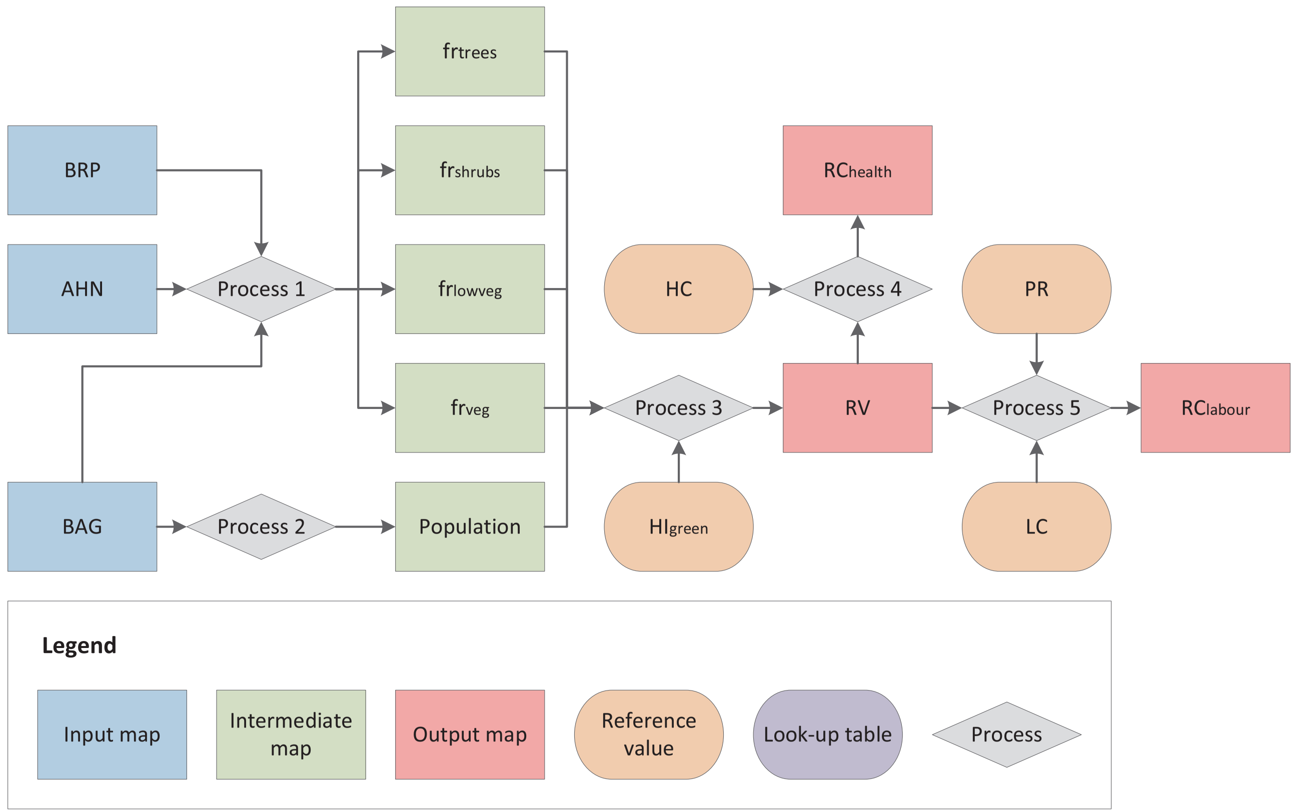

Figure A2 - 7: Urban model for the ecosystem service 'Urban Health' (adapted from Remme et al., 2018)

5. Create a layer displaying the *Reduced number of visits to the doctor per year due to green space (visits/cell/year), RV,* by implementing Formula A2 - 34 Formula A2 - 34.

$$RV = HI_{green} \times Green_{1km} \times Population$$

Formula A2 - 34

where HI_{green} is the *Hypothetical fraction of reduced visits to general practitioners (visits/cell/yr) as a result of the amount of urban green surrounding their household (1 km radius).* HI_{green} is based on a study by Maas (2009) and is equal to 0.000835 visits per percentage of urban green.

A2 – 2.5.2.4 Calculating the reduced health costs due to urban green

6. Calculate the *Reduced health costs due to urban green,* RC_{health}, implementing Formula A2 - 35.

$$RC_{health} = RV \times HC$$

Formula A2 - 35

where *HC* are the *Average reduction in health costs per patient for nine diseases that had a relation to urban green (€/patient/year).* HC values are based on KPMG (2012) and the tool 'Cijfertool Kosten van Ziekten' (RIVM, 2003), and were valued at €868 per patient per year.

A2 – 2.5.2.5 Calculating the reduced labour costs from improved health

7. Calculate the *Reduced labour costs from improved health,* RC_{labour}, implementing Formula A2 - 36.

$$RC_{labour} = RV \times PR \times LC$$

Formula A2 - 36

where *PR* is the *Fraction of people that participate in the labour market (0-1), and LC* are the *Annual avoided health-related labour costs from absenteeism, reduced labour productivity, and job losses (€/person/year), based on* KPMG (2012) and Steenbeek et al. (2010).

A2 – 2.6: 'Water storage' model specifications

This Section describes the process required for modelling the indicators presented in this study for the ecosystem service 'water storage', conform the NC-Model, based on Paulin et al. (2019). A schematic overview of the steps described hereunder is presented in Figure A2 - 8.

A2 – 2.6.1 Spatial data requirements

- Agricultural crop parcels (BRP)
- Basic registry of addresses and buildings (BAG)
- Elevation map of the Netherlands (AHN2)
- High resolution aerial photograph of the Netherlands (Luchtfoto)
- Contour of populated areas (Bevolkingskernen)

A2 – 2.6.2 Model procedure

A2 – 2.6.2.1 Creating vegetation layers

1. Vegetation layers at a 10 m resolution are created, conform Appendix 2 – 2 (Section A2 – 2.1.2.1), using the *AHN2, BAG*, *Luchtfoto,* and *BRP* layers as input. Vegetation layers include *Trees* (fr_{tree}), *Bushes/shrubs* (fr_{shrub}), and *Low vegetation* ($fr_{loweveg}$). Each layer shows the *Fraction per grid cell (0-1) covered by each vegetation type.*

A2 – 2.6.2.2 Calculating water storage

2. Following Paulin et al. (2019), calculate *Water storage by green* (m^3), *WS,* by implementing *Formula A2 - 37.*

$$WS = fr_{rain} \times P \times fr_{veg}$$

Formula A2 - 37

where fr_{rain} is the *Fraction of rainfall water stored by vegetated areas (0-1)*, *P* is the *Value of annual precipitation per cell* (m^3), and fr_{veg} is the *Fraction of a cell that is vegetated (0-1).* Following SBRCURnet (www.sbrcurnet.nl), based on expert elicitation, fr_{rain} is equal to 0.55. In the Netherlands there is an average precipitation of 844 mm per year (https://www.clo.nl/indicatoren/nl0508-jaarlijkse-hoeveelheid-neerslag-in-nederland), which is equivalent to 844 L per m^2, or 0.844 m^3 per m^2, of water on the surface. Since every cell has an area of 100 m^2, the value of annual precipitation per 100 m^2cell, *P*, is equal to 84.4 m^3.

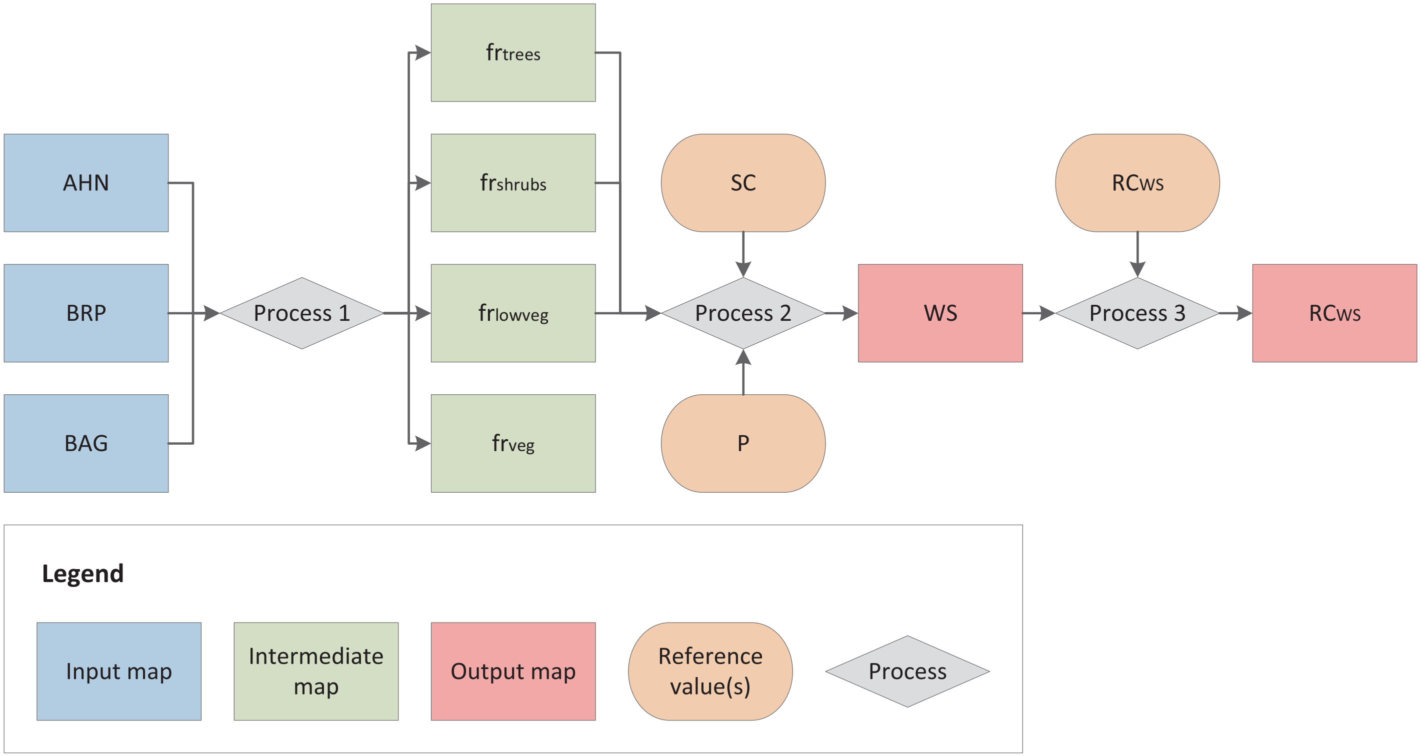

Figure A2 - 8: Urban model for the ecosystem service 'Water Storage' (adapted from Paulin et al., 2019)

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3. Water storage was only estimated for inhabited areas, since these are the areas where the contribution of vegetation to the reduction of rainwater in sewers is most prominent. To do this, an *Adjusted Water storage by green (m^3), WS_{ADJ}*, layer is created considering *WS* values only falling within *Contour of populated areas* from the *Bevolkingskernen*, raster.

A2 – 2.6.2.2 Calculating the economic benefit water storage

Rainfall stored by vegetation would otherwise end up in sewers, incurring higher treatment costs. Based on SBRCURnet , the associated sewage treatment costs are estimated at €0.78/m3/year (www.sbrcurnet.nl).

4. Following Paulin et al. (2019), calculate the *Reduced treatment costs due to the water storage by vegetation (€), RC_{WS},* implementing Formula A2 - 38.

$$RC_{WS} = WS_{ADJ} \times TC$$

Formula A2 - 38

where *TC* is the *Total sewage treatment costs (€/m^3) and is equal to* €0.78/m^3.

Appendix 2 – 3

Table A2 - 7: Respective quantiles for legends of input maps in Figure 3.3 (cell=10 x 10 m)

Ecosystem service	Unit	Quantile 1	Quantile 2	Quantile 3	Quantile 4
Trees	% cover/cell	0 - 5	5 - 15	15 - 38	38 - 100
Bushes/shrubs	% cover/cell	0 - 3	3 - 7	7 - 12	12 - 100
Low vegetation	% cover/cell	0 - 18	18 - 44	44 - 88	88 - 100
Population	inhabitants/cell	0 - 1	1 - 3	3 - 5	5 - 189

Table A2 - 8: Respective quantiles for ecosystem service maps presented in Figure 3.4 (cell = 10 x 10 m)

Ecosystem service	Unit	Quantile 1	Quantile 2	Quantile 3	Quantile 4	Quantile 5	Quantile 6
Air quality regulation	kg/cell/yr	0 - 0.029	0.029 - 0.052	0.052 - 0.056	0.056 - 0.061	0.061 - 0.077	0.077 - 0.340
Physical activity	min/cell/yr	0 - 0.0006	0.0006 - 4	4 - 8	8 - 12	12 - 24	24 - 1,046
Property value	€/cell	0 - 12,000	12,000 - 15,000	15,000 - 18,000	18,000 - 21,000	21,000 - 28,000	28,000 - 129,000
Urban cooling	°C/cell/yr	0 - 1.15	1.15 - 1.50	1.50 - 1.78	1.78 - 2.09	2.09 - 2.35	2.35 - 3.21
Urban health	€/cell/yr	0 - 0.003	0.003 - 25	25 - 50	50 - 74	74 - 124	124 - 6300
Water storage	m^3/cell/yr	0 - 4	4 - 12	12 - 22	22 - 35	35 - 47	47 - 48

Appendix 3 - Supplementary material Chapter 4

Appendix 3 – 1

A3 – 1.1: Input maps

This Section provides an overview of the input maps that were used for implementing models from the NC-Model. Most datasets are publicly available at standard international data repositories, such as the INSPIRE (https://inspire-geoportal.ec.europa.eu/) and ESRI geoportals (http://esrinl-content.maps.arcgis.com/), and all datasets can be found at governmental data registries or upon request. INSPIRE geoportal is the central European access point to the data provided by EU Member States and several EFTA countries under the INSPIRE Directive.

Table A3 - 1: Input datasets used to model six urban ecosystem services

Dataset name	Description	Resolution	Year	Source	Responsible organisation	Email
Actueel Hoogtebestand Nederland (AHN2)	Elevation data	0.5 x 0.5 m	2007-2012	Rijkswaterstaat (RWS)	Stuurgroep AHN	info@ahn.nl
Basisregistratie Adressen en Gebouwen (BAG)	Basic registry of addresses and buildings	2.5 x 2.5 m	2016	Kadaster (2019a)	Kadaster	PPB-GVA@kadaster.nl
Basisregistratie Gewaspercelen (BRP)	Agricultural areas of the Netherlands	25 x 25 m	2017	RWS	RWS	servicedesk-data@rws.nl
Bevolkingskernen	Contour of populated areas	10 x 10 m	2011	Statistics Netherlands (CBS)	CBS	infoservice@cbs.nl
Ecosystem Unit Map (EUM)	Land use map	10 x 10 m	2017	Van Leeuwen et al. (2017)	CBS	infoservice@cbs.nl

Fijnstof 2017 (pm10)	Concentration of particulate matter up to 10 micrograms	25 x 25 m	2017	Velders et al. (2017)	Rijksinstituut voor Volksgezondheid en Milieu (RIVM)	geodata@rivm.nl
Luchtfoto	High resolution aerial photograph of the Netherlands	0.25 x 0.25 m	2017	Beeldmateriaal Nederland	Beeldmateriaal Nederland	info@beeldmateriaal.nl
Top10NL	Topographic land use map of the Netherlands	5 x 5 m	2017	Kadaster (2019b)	Kadaster	PPB-GVA@kadaster.nl
Wijk- en buurtkaart	District and neighbourhood data	160 x 160 m	2017	Bresters (2019)	CBS	infoservice@cbs.nl
Windsnelheden op 100m hoogte	Average wind speed at an elevation of 100 m	2.5 x 2.5 km	2015	Koninklijk Nederlands Meteorologisch Instituut (KNMI)	Rijksdienst voor Ondernemend Nederland (RvO)	geodatabeheer.gisc@rvo.nl
Woningbouwplannenkaart	Housing plans map	10 x 10 m	2018	Gemeente Amsterdam	Gemeente Amsterdam	monitorwoningbouwplannen@amsterdam.nl
WOZ-waarde	Real estate value	10 x 10 m	2016	CBS	CBS	infoservice@cbs.nl

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Actueel Hoogtebestand Nederland (AHN2)

The *Elevation map of the Netherlands (AHN2)* is the second version of the digital elevation model of the Netherlands. It contains detailed and precise data containing elevation information for the Netherlands (relative to Dutch Ordinance Level - NAP). Elevation is measured through laser altimetry, a technique in which the earth's surface is scanned from an aircraft or helicopter with a laser beam. The measurement of the transit time of the laser reflection and of the position of the aircraft together give a very accurate result. Two types of elevation information were used, each from a separate TIF-file (raster files at a 0.5-metre resolution): ground-level elevation and the top-level elevation of all objects, relative to NAP.

Basisregistratie Adressen en Gebouwen (BAG)

The *Basic registry of addresses and buildings (BAG)* map contains information from the government system of basic registries. Data on addresses and buildings is collected by municipalities, which are also responsible for the quality of the data. Organisations with a public task, such as ministries, water boards, police forces and security regions, are obliged to use the authentic data from the registrations.

Basisregistratie Gewaspercelen (BRP)

The *Agricultural crop parcels (BRP)* map contains information on the distribution of agricultural plots along with the type of crops grown. It is a selection of information selected by the Netherlands Enterprise Agency (*Rijksdienst voor Ondernemend Nederland*). The contours of agricultural plots come from the Agricultural Area Netherlands (AAN) dataset. Users of a parcel must annually indicate which type of crop is grown within the parcel. Datasets are generated yearly using May 15 as the reference date.

Bevolkingskernen

The map of the *Contour of populated areas, Bevolkingskernen*, in the Netherlands contains the digital geometry of the contours of the population centres with key figures for 2011 in ESRI™ format.

Ecosystem Unit Map (EUM)

Commissioned by the Ministry of Economic Affairs, the *Land use map of the Netherlands (EUM)* delineates ecosystem units in the Netherlands, categorised into six main themes; agriculture, dunes and beaches, forests and other (semi) natural and unpaved terrain, marshes and floodplains, water and paved and built-up land.

Fijnstof 2017 (pm10)

The *Concentration of particulate matter up to 10 micrograms, Fijnstof 2017 (pm10),* map contains information on the atmospheric concentrations of nitrogen dioxide and particulate matter in the Netherlands. Maps were developed for the National Air Quality Cooperation Program (NSL), a program to improve air quality in the Netherlands. The data on emissions and future scenarios also serve as a basis for monitoring the Nitrogen Approach Program (PAS). These programs test, among other things, the effects of spatial plans on the concentrations of pollutants in the air.

Luchtfoto

The *High resolution aerial photograph of the Netherlands, Luchtfoto,* map contains detailed aerial photos commissioned by the national government, provinces, and water boards at a 25 cm resolution.

Top10NL

The *Topographic land use map of the Netherlands, TOP10NL*, is the basic digital topographic file of the Netherlands for the national registry. It is the most detailed product within the Basic Registration Topography (BRT) and can be used at scale levels between 1:5,000 and 1:25,000.

Wijk- en buurtkaart

The *Key neighbourhood statistics, Wijk- en buurtkaart,* map contains the digital geometry of the boundaries of neighbourhoods, districts, and municipalities. It includes the key figures of neighbourhoods and the aggregated key figures of the districts and municipalities. The *Key neighbourhood statistics* map comprises three sources: the municipal boundaries come from the Land Registry Basic Register (BRK); the neighbourhood boundaries are specified by municipalities, and the boundaries of the country including larger waters is based on Statistics Netherland's (CBS) Soil Use File.

Windsnelheden op 100m hoogte

The *Average windspeed at an elevation of 100 m, Gemiddelde windsnelheid 100m,* map shows the average wind speed in the Netherlands at 100 m elevation for the years 2004-2013. It is the operational working model for the Royal Dutch Meteorological Institute (KNMI), HARMONIE, which reconstructs hourly wind patterns. This is done at a horizontal resolution of 2.5 km. The average wind speed in HARMONIE has been compared with different measuring approaches throughout the Netherlands, a correction that is applied for the entire Netherlands. The inaccuracy of the average wind within 2.5 x 2.5 km grid cell is estimated at ± 0.3m/s.

Woningbouwplannenkaart

The *Housing plans map, Woningbouwplannenkaart,* dataset from the Municipality of Amsterdam, showing areas where new residential plans have been made for the upcoming years, including the number of housing units that are expected to be built.

WOZ-waarde

The *Real estate value, WOZ-Waarde*, map contains data on the value of all properties (WOZ objects), based on the Real Estate Valuation Act (WOZ). The figures are broken down according to the value of property and non-residential property objects and the average home value. The figures are hierarchically classified by country, province, COROP (Dutch regional division) area and municipality. Due to various social developments, the philosophy and method underlying the definition are no longer up to date. In addition, it appears that other authorities, depending on the area of application, use a different classification of metropolitan agglomerations and urban regions, so that it is no longer possible to speak of one standard.

Appendix 3 – 2

Table A3 - 2: Classification of ecosystem service supply and use proxy indicators, based on the CICES (version 5.1; https://cices.eu). Indicators visible under the columns "Ecosystem service indicator" and "Good/benefit indicator" (NA = Not applicable).

CICES V5.1						NC-Model			
Section	**Division**	**Group**	**Class**	**CICES code**	**Class type**	**Ecosystem service indicator**	**Description**	**Good/ benefit indicator**	**Description**
Regulation & Mainte-nance (Biotic)	Transfor-mation of bi-ochemical or physical inputs to ecosystems	Mediation of wastes or toxic substances of anthropogenic origin by living processes	Filtration/ sequestration/ storage/ accumulation by micro-organisms, algae, plants, and animals	2.1.1.2	By type of living system, or by water or substance type	PM_{10} retention by urban vegetation	Reduction in atmospheric PM_{10} concentrations by vegetation and water	Reduced health costs	Reduction in health costs from avoided PM_{10} related mortalities
Regulation & Mainte-nance (Biotic)	Regulation of physical, chemical, biological conditions	Regulation of baseline flows and extreme events	Control of erosion rates	2.2.1.1	By reduction in risk, area protected	Runoff reduction - Reduced rainwater in sewers	Avoided rainwater in the drainage system due to water storage by vegetation	Reduced water treatment costs	Reduction in water treatment costs from avoided rainwater in the drainage system
Regulation & Mainte-nance (Biotic)	Regulation of physical, chemical, biological conditions	Atmospheric composition and conditions	Regulation of temperature and humidity, including ventilation and transpiration	2.2.6.2	By contribution of type of living system to amount, concentration or climatic parameter	Reduction in UHI effect	Contribution by vegetation and water to mitigation of the UHI effect	NA	NA

Regulation & Mainte-nance (Biotic)	Other	Other	Other	2.3.X.X	Other	Vegetation and water	Amount of vegetation in a 1km radius of a person's residence	Reduced health costs	Reduction in health costs linked to the contribution by green space to mitigating the incidence of seven disease categories (i.e., cardiovascular, musculoskeletal, mental, respiratory, neurological, and digestive diseases, and a miscellaneous category)
								Reduced visits to GP	Avoided visits to GP linked to the contribution of green space to improved health conditions
								Reduced labour costs	Reduction in costs of absen-teeism, reduced labour productivity, and job losses, linked to the contribution of green space to

									improved health conditions
Cultural (Biotic)	Direct, in-situ and outdoor interactions with living systems that depend on presence in the environ-mental setting	Physical and experiential interactions with natural environment	Characteristics of living systems that that enable activities promoting health, recuperation or enjoyment through active or immersive interactions	3.1.1.1	By type of living system or environ-mental setting	Vegetation abundance	Amount of vegetation in a 1km radius of a person's residence	Cycling for commuting purposes	Contribution to time cycled by individuals for commuting purposes that can be attributed to the availability of green space in their surroundings
									Avoided all-cause mortalities from enhanced health benefits due to the contribution to cycling (commuting)
									Economic gains from reduced all-cause mortalities, based on the value of a statistical life
Cultural (Biotic)	Direct, in-situ and outdoor interactions with living systems that depend on presence in the environ-	Physical and experiential interactions with natural environment	Characteristics of living systems that enable activi-ties promoting health, recuperation or enjoyment	3.1.1.2	By type of living system or environ-mental setting	Vegetation and water	Different vegetation and water typologies and configurations	Contribu-tion to property value	Contribution by vegetation and open water to property prices due to the value assigned to these elements by urban dwellers

	mental setting		through passive or observational interactions						

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Appendix 3 – 3

Table A3 - 3: Respective quantiles for legends of vegetation cover and population maps in Figure 4.3 (cell=10 x 10 m)

ES	Unit	Q1	Q2	Q3	Q4
Trees	% cover/cell	0 - 5	5 - 16	15 - 38	38 - 100
Bushes/shrubs	% cover/cell	0 - 3	3 - 7	7 - 12	12 - 100
Low vegetation	% cover/cell	0 - 18	18 - 43	43 - 87	88 - 100
Population density	inhabitants/cell	0 - 1	1 - 2	2 - 4	4 - 185

Table A3 - 4: Respective quantiles for ecosystem service change maps presented in Figure 4.5 (cell = 10 x 10 m)

Scenario	Ecosystem service	Indicator	Unit	Q1	Q2	Q3	Q4	Q5
Green Neighbourhoods	Physical Activity	Contribution to cycling (commuting)	mins/cell/yr	0 - 1.8	1.8 - 3.5	3.5 - 5.3	5.3 - 7.1	7.1 - 228
Green Neighbourhoods.	Water Storage	Reduced water treatment costs	EUR/cell/yr	0 - 0.3	0.3 - 1.5	1.5 - 8.5	8.5 - 24	24 - 38
Green Network	Air Quality Regulation	PM_{10} retention	kg/cell/yr	0.00 - 0.03	0.03 - 0.04	0.04 - 0.05	0.05 - 0.07	0.007 - 0.20
Urban Parks	Urban Cooling	Mitigation of the UHI effect	°C/cell/yr	0 - 0.02	0.02 - 0.04	0.04 - 0.06	0.06 - 0.10	0.10 - 0.90

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Table A3 - 5: Total relative change (percentage increase) in ecosystem service values. GNei = Green Neighbourhoods; GNet = Green Network; UP = Urban Parks. Section 'a' presents the percentage increase per indicator relative to the Business-As-Usual per GI scenario (lowest = yellow, highest = dark green). Section 'b' presents the mean and standard deviation (SD) of percentages changes for each indicator (lowest = yellow, highest = dark green). Section 'c' presents the mean and standard deviation of percentage changes in each scenario (lowest = yellow, highest = dark green).

a) Percentage increase in ecosystem service supply and use per scenario / **b) Mean and std. dev. per indicator**

ES	Indicator	GNei	GNet	UP	Mean	SD
Air Quality Regulation	PM_{10} retention	3.0%	8.2%	3.4%	**5%**	**3%**
Air Quality Regulation	Reduced health costs	3.1%	8.3%	3.4%	**5%**	**3%**
Physical Activity	Contribution to cycling (commuting)	14.0%	9.4%	3.7%	**9%**	**5%**
Physical Activity	Reduced mortality	15.8%	10.5%	5.3%	**11%**	**5%**
Physical Activity	Reduced costs from reduced mortality	14.0%	9.4%	3.7%	**9%**	**5%**
Property Value	Contribution to property value	1.6%	1.2%	0.4%	**1%**	**1%**
Urban Cooling	Reduction of UHI effect	0.0%	0.6%	2.3%	**1%**	**1%**
Urban Health	Reduced visits to GP	14.0%	9.4%	3.7%	**9%**	**5%**
Urban Health	Reduced health costs	14.0%	9.4%	3.7%	**9%**	**5%**
Urban Health	Reduced labour costs	14.0%	9.4%	3.7%	**9%**	**5%**
Water Storage	Reduced rainwater in sewers	6.8%	7.7%	4.4%	**6%**	**2%**
Water Storage	Reduced water treatment costs	6.8%	7.7%	4.4%	**6%**	**2%**

c) Mean and std. dev. per indicator per scenario

	GNei	GNet	UP
Mean	**9%**	**8%**	**3%**
SD	**6%**	**3%**	**1%**
Heterogeneity	**67%**	**43%**	**34%**

Appendix 3 – 4

Figure A3 - 1: Changes in performance of 'Air Quality - PM_{10} retention', for different scenarios in reference to the Business-As-Usual scenario (cell size = 10 x 10 m). For each map, legends show quantile values. Unshaded areas comprise areas where no value has been assigned.

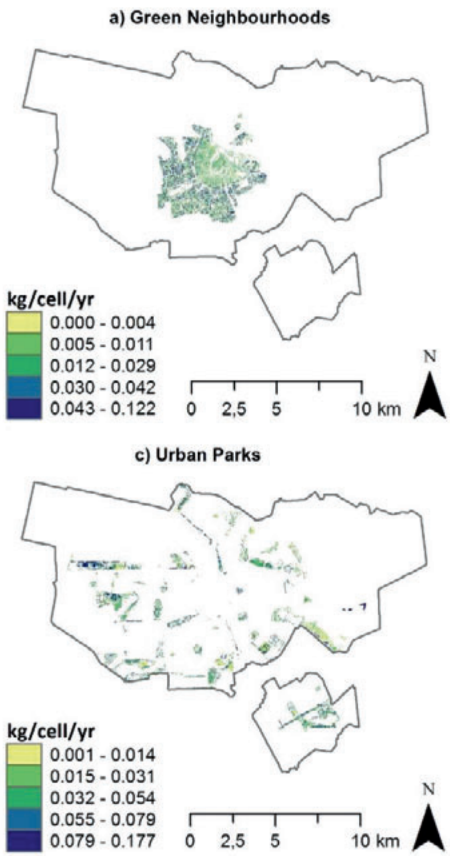

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Figure A3 - 2: Changes in performance of 'Physical Activity - Contribution to cycling (commuting)', for different scenarios in reference to the Business-As-Usual scenario (cell size = 10 x 10 m). For each map, legends show quantile values. Unshaded areas comprise areas where no value has been assigned.

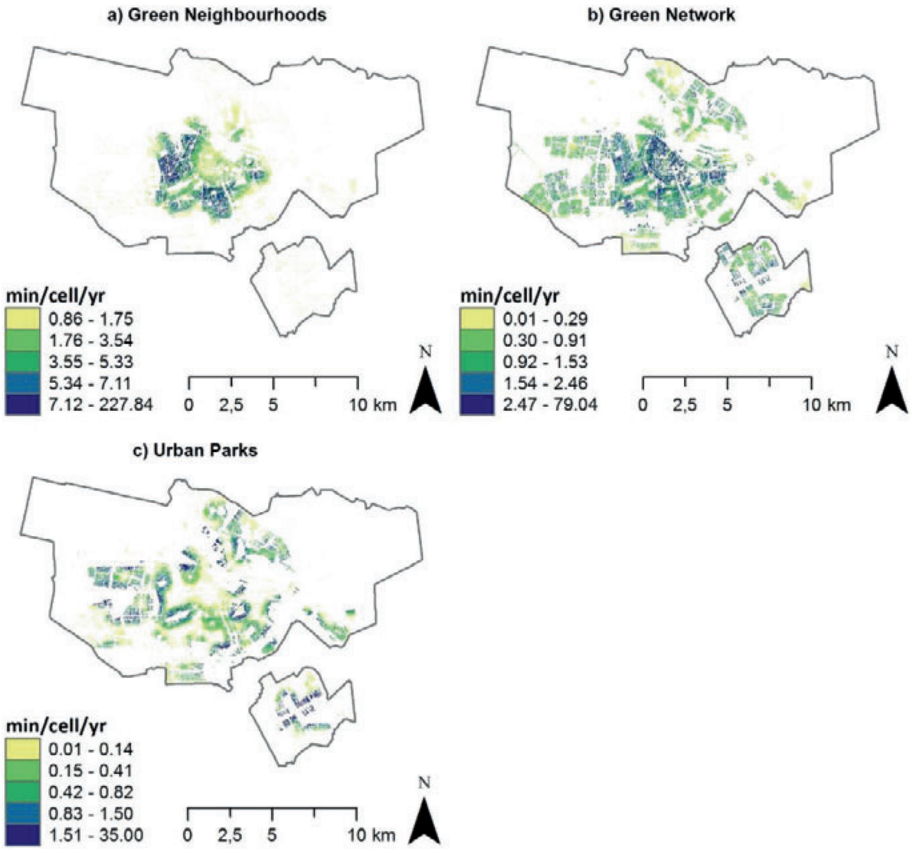

Figure A3 - 3: Changes in performance of 'Urban Cooling - Reduction in UHI effect', for different scenarios in reference to the Business-As-Usual scenario (cell size = 10 x 10 m). For each map, legends show quantile values. Unshaded areas comprise areas where no value has been assigned.

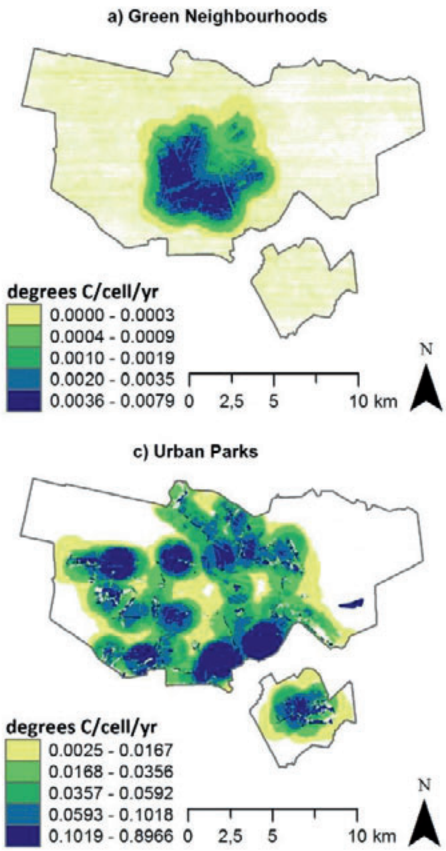

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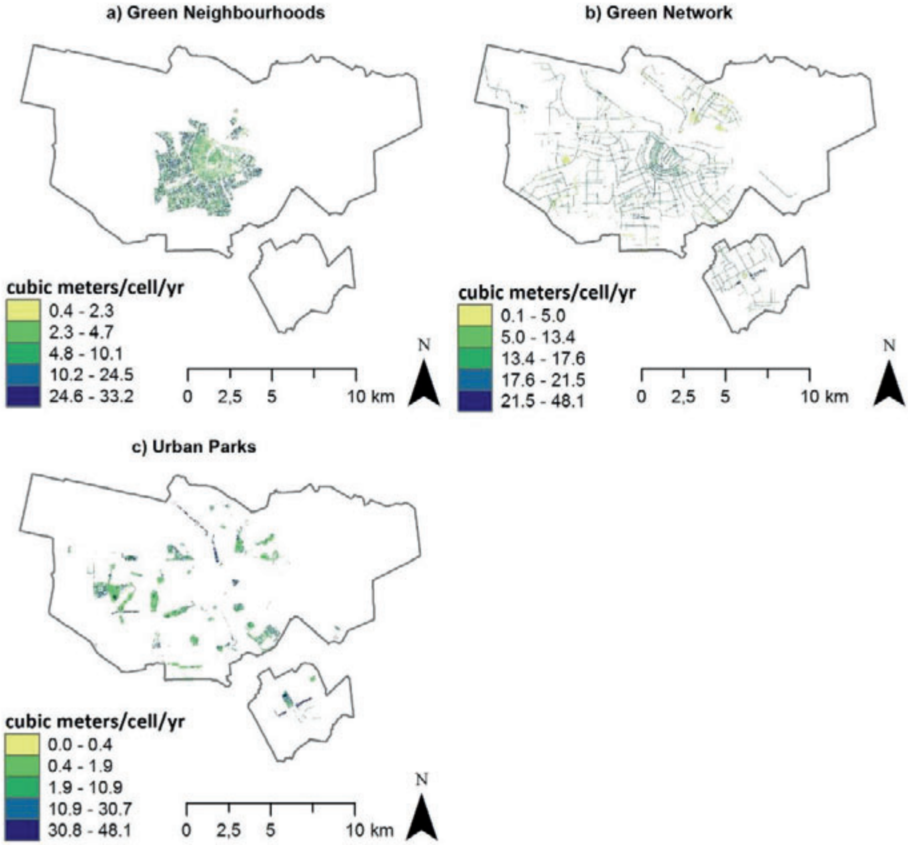
Figure A3 - 4: Changes in performance of 'Water Storage – Reduced rainwater in sewers', for different scenarios in reference to the Business-As-Usual scenario (cell size = 10 x 10 m). For each map, legends show quantile values. Unshaded areas comprise areas where no value has been assigned.



Appendix 4 - Supplementary material Chapter 5

Appendix 4 – 1

Table A4 - 1: Stakeholder groups considered within the assessment, their identified key objectives (based on the literature and semi-structured interviews), and ecosystem services relevant based on these objectives.

Stakeholder group	Key objectives	Relevant ecosystem services	Actor group	Website	Interviewed
Farmers	-Innovative, multifunctional agriculture -Long-term production -Aesthetic landscape	-Crop Production -Pest Control -Soil Biodiversity -Landscape Services	Farmers	-	Yes
			Land en Tuinbouw Organisatie Nederland (LTO Nederland)	https://www.ltonoord.nl/	Yes
Environmental organisations	-Bird species protection (e.g., meadow birds) -R&D: Innovative agriculture -Meadow bird management and protection targets -Field margin management and protection targets	-Crop Production -Pest Control -Soil Biodiversity -Landscape Services	Cooperatie Collectief Hoeksche Waard (CCHW)		Yes
			H-WodKa	https://cchw.eu/	No
			Stichting Rietgors		Yes
Local governments	-Innovative, multifunctional agriculture -Biodiversity protection and management -Natura2000 objectives -Meadow bird management and protection targets -Field margin management and protection targets -Recreational opportunities	-Soil Biodiversity -Landscape Services	Hoeksche Waard Municipality	http://hwodka.nl/	Yes
			South Holland Province		Yes
			European Union	www.rietgorsinfo.nl/home/	No

Nature organisations	-Protected areas objectives -Nature and biodiversity (culturally and ecologically relevant species) protection and management -Aesthetic landscape -Innovative, multifunctional agriculture -Research, education and communication	-Air Quality Regulation -Pest Control -Soil Biodiversity -Landscape Services	Staatsbosbeheer		Yes
			Natuurmonumenten	https://www.gemeenteh w.nl/	Yes
			Hoeksche Waards Landschap (HWL)		No
Water board	-Waste water treatment -Providing clean water -Safe water ways	-Air Quality Regulation -Pest Control -Soil Biodiversity	Waterschap Hollandse Delta	https://www.zuid-holland.nl/	Yes
Local inhabitants	-Good quality of life -Recreational opportunities -Aesthetic landscape -Nature protection	-Air Quality Regulation -Human Health -Landscape Services -Property Value	Local inhabitants		No

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Appendix 4 – 2

Table A4 - 2: Classification of ecosystem service supply and use proxy indicators, based on CICES (version 5.1; https://cices.eu). Modelled indicators are visible under the columns "Supply indicator" and "Use indicator". ES = Ecosystem service; NA = not applicable.

CICES V5.1			Indicators modelled				
Section	**Class**	**CICES code**	**ES simple descriptor (model name)**	**Supply indicator(s)**	**Description**	**Use indicator(s)**	**Description**
Provisioning (Biotic)	Cultivated terrestrial plants (including fungi, algae) grown for nutritional purposes	1.1.1.1	Crop Production	Volume of harvested crop	The ecological contribution to the growth of cultivated, land-based crops	Net output of harvested crops	Total market value of harvested crops before subtracting expenses
						Value added of harvested crops	Total market value of harvested crops once the costs of purchased goods and services required for producing and harvesting the crop have been deducted. This value does not exclude government incentives (e.g., subsidies) and factor costs.
Regulation & Maintenance (Biotic)	Filtration/sequestration/ storage/accumulation by micro-organisms, algae, plants, and animals	2.1.1.2	Air Quality Regulation	PM_{10} retention by vegetation and water	Reduction in atmospheric PM_{10} concentrations by vegetation and water	Reduced health costs	Reduction in health costs from avoided PM_{10} related mortalities
Regulation & Maintenance (Biotic)	Other	2.3.X.X	Human Health	Vegetation	Abundance of vegetation surrounding a person's residence	Reduced health costs	Reduction in health costs linked to the contribution by green space to mitigating the incidence of seven disease categories (i.e., cardiovascular diseases, musculoskeletal diseases, mental diseases, respiratory diseases, neurological diseases, digestive

							diseases, and a miscellaneous category)
						Reduced visits to general practitioners	Avoided visits to general practitioners linked to the contribution of green space to improved health conditions
						Reduced labour costs	Reduction in costs of absenteeism, reduced labour productivity, and job losses, linked to the contribution of green space to improved health conditions
Regulation & Maintenance (Biotic)	Pest control (including invasive species)	2.2.3.1	Pest Control	Yearly effective pest control	Reduction in abundance of pests that cause damage to cultivated crops due to the presence of natural enemies inhabiting field margins	NA	NA
Regulation & Maintenance (Biotic)	Maintaining nursery populations and habitats (Including gene pool protection)	2.2.2.3	Soil Biodiversity	Performance of soil biodiversity	State of populations of species that support soil quality, based on biotic and abiotic characteristics of the landscape	NA	NA
Cultural (Biotic)	Characteristics of living systems that are resonant in terms of culture or heritage	3.1.2.3	Landscape Services	Semi-natural landscape elements	Agricultural fields, large water bodies, grasslands, shrubs, polder structures, ditches, creeks, trees, hedges, wood walls	Cultural identity and heritage	The perceived contribution of landscape semi-natural elements to the cultural identity and heritage of the area
Cultural (Biotic)	Characteristics of living systems that enable scientific investigation	3.1.2.1	Landscape Services	Semi-natural landscape elements	Agricultural fields, large water bodies, grasslands, shrubs,	Educational and	The perceived contribution of landscape semi-natural elements to educational and scientific knowledge

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	or the creation of traditional ecological knowledge				polder structures, ditches, creeks, trees, hedges, wood walls	scientific knowledge	
Cultural (Biotic)	Characteristics of living systems that enable activities promoting health, recuperation or enjoyment through passive or observational interactions	3.1.1.2	Landscape Services	Semi-natural landscape elements	Agricultural fields, large water bodies, grasslands, shrubs, polder structures, ditches, creeks, trees, hedges, wood walls	Habitability	The perceived contribution of landscape semi-natural elements to the habitability of the area
Cultural (Biotic)	Elements of living systems that have symbolic meaning	3.2.1.1	Landscape Services	Semi-natural landscape elements	Agricultural fields, large water bodies, grasslands, shrubs, polder structures, ditches, creeks, trees, hedges, wood walls	Intrinsic value	The perceived contribution of landscape semi-natural elements to intrinsic values
Cultural (Biotic)	Characteristics of living systems that enable aesthetic experiences	3.1.2.4	Landscape Services	Semi-natural landscape elements	Agricultural fields, large water bodies, grasslands, shrubs, polder structures, ditches, creeks, trees, hedges, wood walls	Landscape aesthetics	The perceived contribution of landscape semi-natural elements to landscape aesthetics
Cultural (Biotic)	Characteristics of living systems that that enable activities promoting health, recuperation or enjoyment through active or immersive interactions	3.1.1.1	Landscape Services	Semi-natural landscape elements	Agricultural fields, large water bodies, grasslands, shrubs, polder structures, ditches, creeks, trees, hedges, wood walls	Recreational potential	The perceived contribution of landscape semi-natural elements to an area's recreational potential

Cultural (Biotic)	Characteristics of living systems that enable activities promoting health, recuperation or enjoyment through passive or observational interactions	3.1.1.2	Property Value	Vegetation and water	Different vegetation typologies and configurations	Contribution to property value	Contribution by vegetation to property prices due to the value assigned to these elements

Appendix 4 – 3

A4 – 3.1: Input maps

This Section provides an overview of the input maps required for implementing models described in in this paper. Most datasets are publicly available at standard international data repositories, such as the INSPIRE (https://inspire-geoportal.ec.europa.eu/) and ESRI geoportals (http://esrinl-content.maps.arcgis.com/), and governmental data registries (https://www.pdok.nl/; https://www.nationaalgeoregister.nl/; https://data.overheid.nl/). INSPIRE geoportal is the central European access point to the data provided by EU Member States and several EFTA countries under the INSPIRE Directive. The field margins map was obtained from engineering consultancy IB-Lerink (www.ib-lerink.nl/.

Table A4 - 3: Input datasets necessary to model supply and use of seven local ecosystem services

Original dataset name	Description	Source	Responsible organisation	Email
Akkerranden Hoeksche Waard 2017	Field margins of the Hoeksche Waard	Waterschap Hollandse Delta (WSHD)	WSHD	info@ahn.nl
Actueel Hoogtebestand Nederland (AHN2)	Elevation map of the Netherlands	National water board (RWS)	Stuurgroep AHN	servicedesk-data@rws.nl
Basisregistratie Adressen en Gebouwen (BAG)	Basic registry of addresses and buildings	Kadaster (2019a)	Kadaster	PPB-GVA@kadaster.nl
Basisregistratie Gewaspercelen (BRP)	Agricultural areas of the Netherlands	RWS	RWS	servicedesk-data@rws.nl
Bodem Biologische indicator (BoBi)	Soil biological, physical, and chemical characteristics	Rutgers et al. (2009), Van Wijnen et al. (2012)	National Institute for Public Health and the Environment (RIVM)	geodata@rivm.nl
Bodemfysische Eenhedenkaart (BOFEK2012)	Soil biophysical units	Wageningen University & Research (WUR, 2012)	WUR	Not publicly available
Ecosystem Unit Map (EUM)	Land use map of the Netherlands	Van Leeuwen et al. (2017)	Statistics Netherlands (CBS)	infoservice@cbs.nl

Fijnstof 2017 (pm10)	Concentration of particulate matter up to 10 micrograms (PM_{10})	Velders et al. (2017)	RIVM	geodata@rivm.nl
Luchtfoto	High resolution aerial photograph of the Netherlands	Beeldmateriaal Nederland	Beeldmateriaal Nederland	info@beeldmateriaal.nl
Population	Distribution of inhabitants in the Netherlands	Remme et al. (2018)	RIVM	geodata@rivm.nl
Top10NL	Topographic land use map of the Netherlands	Kadaster (2019b)	Land Registry (Kadaster)	PPB-GVA@kadaster.nl
Vegetation	Distribution of green vegetation	Remme et al. (2018)	RIVM	geodata@rivm.nl
Wijk- en buurtkaart	Key neighbourhood statistics	Bresters (2019)	CBS	infoservice@cbs.nl
WOZ-Waarde	Real estate value	CBS	CBS	infoservice@cbs.nl

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Actueel Hoogtebestand Nederland (AHN2)

The *Elevation map of the Netherlands (AHN2)* is the second version of the digital elevation model of the Netherlands. It contains detailed and precise data containing elevation information for the Netherlands (relative to Dutch Ordinance Level - NAP). Elevation is measured through laser altimetry, a technique in which the earth's surface is scanned from an aircraft or helicopter with a laser beam. The measurement of the transit time of the laser reflection and of the position of the aircraft together give a very accurate result. Two types of elevation information were used, each from a separate TIF-file (raster files at a 0.5-metre resolution): ground-level elevation and the top-level elevation of all objects, relative to NAP.

Akkerranden Hoeksche Waard 2017

The *Field margins of the Hoeksche Waard, Akkerranden Hoeksche Waard 2017*, map provides information on the distribution of field margins in the Hoeksche Waard. Developed by the water board from the South Holland Province, *Waterschap Hollandse Delta* (*WSHD*). Includes 11 different types of field margins, including combinations of grassy and flower-rich field margins, as well as margins directed as habitat for different bird species.

Basisregistratie Adressen en Gebouwen (BAG)

The *Basic registry of addresses and buildings (BAG)* map contains information from the government system of basic registries. Data on addresses and buildings is collected by municipalities, which are also responsible for the quality of the data. Organisations with a public task, such as ministries, water boards, police forces and security regions, are obliged to use the authentic data from the registrations.

Basisregistratie Gewaspercelen (BRP)

The *Agricultural crop parcels (BRP)* map contains information on the distribution of agricultural plots along with the type of crops grown. It is a selection of information from the Basisregistratie Gewaspercelen (BRP) from the Netherlands Enterprise Agency (*Rijksdienst voor Ondernemend Nederland*). The contours of agricultural plots come from the Agricultural Area Netherlands (AAN) dataset. Users of a parcel must annually indicate which type of crop is grown within the parcel. Datasets are generated yearly using May 15 as the reference date.

Bodem Biologische Indicator (BoBi)

The *Soil biological, physical, and chemical characteristics, Bodem Biologische Indicator (BoBi),* dataset provides information on soil biological, chemical, and physical attributes (i.e., measurable characteristics) in the Netherlands. Based on Van Wijnen et al. (2012), which developed maps based on data obtained from the National Soil Quality Monitoring Network (Rutgers et al., 2009). As part of the Soil Monitoring Network, 10 different combinations of soil type and land use have been sampled in 200 locations, mostly agricultural businesses but also nature reserves, since 1999 and in 5-year cycles. All groups of organisms are sampled simultaneously at the same locations.

Bodemfysische Eenhedenkaart (BOFEK2012)

The *Soil biophysical units, Bodemfysische Eenhedenkaart (BOFEK2012),* spatial dataset displays information on the distribution of soil biophysical units (72 classes) in the Netherlands.

Ecosystem Unit Map (EUM)

Commissioned by the Ministry of Economic Affairs, the *Land use map of the Netherlands (EUM)* delineates ecosystem units in the Netherlands, categorised into six main themes; agriculture, dunes and beaches, forests and other (semi) natural and unpaved terrain, marshes and floodplains, water and paved and built-up land.

Fijnstof 2017 (pm10)

The *Concentration of particulate matter up to 10 micrograms, Fijnstof 2017 (pm10),* map contains information on the atmospheric concentrations of nitrogen dioxide and particulate matter in the Netherlands. Maps were developed for the National Air Quality Cooperation Program (NSL), a program to improve air quality in the Netherlands. The data on emissions and future scenarios also serve as a basis for monitoring the Nitrogen Approach Program (PAS). These programs test, among other things, the effects of spatial plans on the concentrations of pollutants in the air.

Population

Spatial dataset providing information on the distribution of inhabitants in the Netherlands, used as input for models in the NC-Model. Its reproduction can be achieved by following the methodology in Paulin et al. (2020b).

Top10NL

The *Topographic land use map of the Netherlands, TOP10NL*, is the basic digital topographic file of the Netherlands for the national registry. It is the most detailed product within the Basic Registration Topography (BRT) and can be used at scale levels between 1:5,000 and 1:25,000.

Vegetation

The *Distribution of green vegetation, Vegetation,* spatial dataset provides information on the distribution of three different types of vegetation (i.e., low vegetation, shrubs/bushes, trees) in the Netherlands, used as input for models in the NC-Model. Its reproduction can be achieved by following the methodology in Paulin et al. (2020b).

Wijk- en buurtkaart

The *Key neighbourhood statistics, Wijk- en buurtkaart,* map contains the digital geometry of the boundaries of neighbourhoods, districts, and municipalities. It includes the key figures of neighbourhoods and the aggregated key figures of the districts and municipalities. The *Key neighbourhood statistics* map comprises three sources: the municipal boundaries come from the Land Registry Basic Register (BRK); the neighbourhood boundaries are specified by municipalities, and the boundaries of the country including larger waters is based on Statistics Netherland's (CBS) Soil Use File.

Appendix 4 – 4

A4 – 4.1: 'Crop Production' model specifications

This Section describes the process required for modelling ecosystem service supply and use indicators presented in this study by use of the 'Crop Production' model. Below, a stepwise procedure is provided guiding implementation of the model. Only crops for which information was available on their total harvested volume (kg/ha) and gross profit per volume (€/kg), were included (88% of the spatial extent of crops).

A4 – 4.1.1 Spatial data requirements

- Agricultural crop parcels (BRP)

A4 – 4.1.2 Non-spatial data requirements

- Reported statistics on production per crop type in kg/ha (CBS, n.d. -a)
- Reported statistics on gross profit per crop type in €/kg (CBS, n.d. -a; WUR, n.d.)
- Reported statistics on added value per crop type in €/kg (CBS, n.d. -a)

A4 – 4.1.3 Model procedure

1. The *Agricultural crop parcels, BRP,* layer is converted to raster format and to a 10 m resolution.
2. Create a layer where crop type typologies from *BRP* into *Crop production volumes (kg), PV,* by use of Table A4 - 4, for crops considered (source: CBS, n.d. - a). Average values obtained are representative for the Province of South Holland, where the Hoeksche Waard is located. Where unavailable, average values for the Netherlands were used.
3. Reclassify crop type typologies from *BRP* into *Net output values (€), NET,* by use of Table A4 - 5, for crops considered (source: CBS, n.d. - a). Average values obtained are representative for the Province of South Holland. Where unavailable, average values for the Netherlands were used.
4. Reclassify crop type typologies from *BRP* into *Value added values (€), VA,* by use of Table A4 - 6 (source: CBS, n.d. - a). Average values obtained are representative for the Province of South Holland. Where unavailable, average values for the Netherlands were used. Where value added values were missing, net output values were multiplied by a refence value (0.63), which captures the averaged ratio of value added against net output, for crops considered for which data was available.

Table A4 - 4: Volume of harvested crops per cell for the year 2018 (data source: CBS, n.d. -a).

BRP code	Crop type typology	Production (kg/100 m^2)
233	Tarwe, winter-	92
234	Tarwe, zomer-	74
235	Gerst, winter-	87
236	Gerst, zomer-	68
242	Bonen, bruine-	18
247	Blauwmaanzaad	18
256	Bieten, suiker-	771
257	Bieten, voeder	771
259	Mais, snij-	429
262	Uien, zaai-	328
316	Mais, korrel-	109
1096	Appelen, aangeplant voorafgaande aan lopende seizoen	408
1098	Peren, aangeplant voorafgaande aan lopende seizoen	403
1923	Koolzaad, zomer (incl. boterzaad)	24
1931	Uien, poot- en plant- (incl. sjalotten)	328
2014	Aardappelen, consumptie	443
2015	Aardappelen, poot NAK	278
2016	Aardappelen, poot TBM	278
2777	Spruitkool/spruitjes, productie	193
2795	Bloemkool, winter, productie	176
2797	Bloemkool, zomer, productie	176

Table A4 - 5: Net output of harvested crops per cell for the year 2018 (data sources: CBS, n.d. -a; WUR, n.d.).

BRP code	Crop type typology	Net output (€/100 m^2)
233	Tarwe, winter-	18
234	Tarwe, zomer-	14
235	Gerst, winter-	17
236	Gerst, zomer-	13
242	Bonen, bruine-	17
247	Blauwmaanzaad	50
256	Bieten, suiker-	27
257	Bieten, voeder	27
259	Mais, snij-	65
262	Uien, zaai-	100
316	Mais, korrel-	10
1096	Appelen, aangeplant voorafgaande aan lopende seizoen	209
1098	Peren, aangeplant voorafgaande aan lopende seizoen	366
1923	Koolzaad, zomer (incl. boterzaad)	9
1931	Uien, poot- en plant- (incl. sjalotten)	100
2014	Aardappelen, consumptie	96
2015	Aardappelen, poot NAK	102
2016	Aardappelen, poot TBM	102
2777	Spruitkool/spruitjes, productie	623
2795	Bloemkool, winter, productie	116
2797	Bloemkool, zomer, productie	116

Table A4 - 6: Value added of harvested crops per cell for the year 2018 (data sources: CBS, n.d. -a).

BRP code	Crop type typology	Value added (€/100 m^2)
233	Tarwe, winter-	11
234	Tarwe, zomer-	7
235	Gerst, winter-	11
236	Gerst, zomer-	8
242	Bonen, bruine-	11
247	Blauwmaanzaad	32
256	Bieten, suiker-	15
257	Bieten, voeder	15
259	Mais, snij-	41
262	Uien, zaai-	74
316	Mais, korrel-	6
1096	Appelen, aangeplant voorafgaande aan lopende seizoen	132
1098	Peren, aangeplant voorafgaande aan lopende seizoen	231
1923	Koolzaad, zomer (incl. boterzaad)	6
1931	Uien, poot- en plant- (incl. sjalotten)	74
2014	Aardappelen, consumptie	69
2015	Aardappelen, poot NAK	64
2016	Aardappelen, poot TBM	64
2777	Spruitkool/spruitjes, productie	393
2795	Bloemkool, winter, productie	73
2797	Bloemkool, zomer, productie	73

A4 – 4.2: 'Air Quality Regulation' model specifications

Ecosystem service supply and use indicators presented in this study were modelled by use of the 'Air Quality Regulation' model, conform the NC-Model (Remme et al., 2018; Paulin et al., 2020b). A stepwise procedure guiding the model's application is available in Paulin et al. (2020b).

A4 – 4.2.1 Spatial data requirements

- Agricultural crop parcels (BRP)
- Basic registry of addresses and buildings (BAG)
- Concentration of particulate matter up to 10 micrograms (Fijnstof 2017, pm10)
- Elevation map of the Netherlands (AHN2)
- High resolution aerial photograph of the Netherlands (Luchtfoto)
- Key neighbourhood statistics (Wijk- en buurtkaart)
- Land use map of the Netherlands (EUM)

A4 – 4.2.2 Non-spatial data requirements

- Various reference values found in Appendix 3 – 2.1.

A4 – 4.3: ‘Human Health’ model specifications

This Section describes the process required for modelling ecosystem service supply and use indicators presented in this study for the ecosystem service ‘Human Health’, conform the NC-Model (Remme et al., 2018; Paulin et al., 2020b). In the NC-Model, the model implemented is introduced as the ‘Urban Health’ model. However, given that reference values in the model were obtained from regression studies that have been performed at relatively large scales, including non-urban areas, the model was deemed adequate for this case study. A stepwise procedure guiding the model’s application is available in Paulin et al. (2020b).

A4 – 4.3.1 Spatial data requirements

- Agricultural crop parcels (BRP)
- Basic registry of addresses and buildings (BAG)
- Elevation map of the Netherlands (AHN2)
- High resolution aerial photograph of the Netherlands (Luchtfoto)
- Key neighbourhood statistics (Wijk- en buurtkaart)

A4 – 4.3.2 Non-spatial data requirements

- Various reference values found in Appendix 3 – 2.5.

A4 – 4.4: 'Pest Control' model specifications

This Section describes the process required for modelling ecosystem service supply and use indicators presented in this study by use of the 'Pest Control' model, conform the NC-Model (Paulin et al., 2020b). This paper presents the first published version of the 'Pest Control' model. Below, a stepwise procedure is provided guiding its implementation.

A4 – 4.4.1 Spatial data requirements

- Agricultural crop parcels (BRP)
- Basic registry of addresses and buildings (BAG)
- Elevation map of the Netherlands (AHN2)
- Field margins (Akkerranden Hoeksche Waard 2017)
- High resolution aerial photograph of the Netherlands (Luchtfoto)

A4 – 4.4.2 Non-spatial data requirements

- Look-up tables for designing habitats map for particular pest and natural enemy species
- Expert-based (i.e., scientists) look-up tables for determining effective pest control scores based on the occurrence of combinations of insects (pests and natural enemies) in a particular habitat cell.

A4 – 4.4.3 Model procedure

A4 – 4.4.3.1 Creating habitat for agricultural crop parcels

Table A4 - 7 is based on expert elicitation (scientists) and it displays the potential of agricultural crop typologies in the *Agricultural crop parcels, BRP,* map to act as habitat for aphids, hoverflies, coccinellids, and carabids. A value of 1 suggests that a specified crop typology is suitable as habitat for a specified insect population, while a value of 0 suggests it is unsuitable for this purpose.

1. The *BRP* layer is converted to raster format and to a 10 m resolution.
2. Following Table A4 - 7, reclassify crop typologies from *BRP* into an *Aphids habitat (BRP),* BRP_{aphid}, layer displaying the presence of aphids within each cell (0 = not present, 1 = present).
3. Following Table A4 - 7, reclassify crop typologies from *BRP* into a *Hoverflies habitat (BRP),* BRP_{hover}, layer displaying the presence of hoverflies within each cell (0 = not present, 1 = present).

Table A4 - 7: Look-up table for reclassifying crop type classes (BRP) into potential habitats for pest populations (aphids) and natural enemy populations (hoverflies, coccinellids, carabids)

BRP code	Hoverflies (BRP)	Coccinellids (BRP)	Carabids (BRP)	Aphids (BRP)
174	0	1	0	1
233	0	0	0	1
234	0	0	0	1
235	0	0	0	1

236	0	0	0	1
237	0	1	0	1
238	0	0	0	1
241	1	1	0	1
242	0	1	0	1
244	0	1	0	1
247	0	0	0	1
256	1	1	1	1
257	0	1	0	1
258	0	1	0	1
259	1	1	1	1
262	0	0	0	1
265	0	1	1	1
266	0	1	0	1
308	0	1	0	1
311	0	1	0	1
331	1	1	1	1
332	0	1	0	1
334	0	1	0	1
335	0	0	0	1
336	1	1	0	1
345	1	1	1	1
372	0	0	0	1
382	0	0	0	1
383	0	1	0	1
427	0	0	0	1
428	0	1	0	1
511	0	1	0	1
515	0	0	0	1
653	0	0	0	1
665	1	1	0	1
666	0	0	0	1
670	0	1	0	1
799	0	1	0	1
803	0	0	0	1
814	0	1	0	1
854	0	1	0	1
863	0	1	0	1
991	0	1	0	1
992	0	1	0	1
1004	1	1	1	1
1006	0	1	0	1
1022	0	0	0	1
1023	0	1	0	1
1036	0	1	0	1
1039	0	0	0	1
1044	0	1	0	1
1068	0	1	0	1
1075	0	1	0	1
1094	0	1	0	1
1095	0	1	0	1
1096	0	1	0	1

1097	0	1	0	1
1098	0	1	0	1
1870	0	1	0	1
1925	0	1	0	1
1926	0	0	0	1
1931	0	0	0	1
1936	0	0	0	1
2014	0	1	0	1
2015	0	1	0	1
2016	0	1	0	1
2033	0	1	0	1
2328	0	1	0	1
2645	0	1	0	1
2652	0	1	0	1
2706	0	0	0	1
2708	0	1	0	1
2715	0	1	0	1
2723	0	1	0	1
2725	0	0	0	1
2735	0	0	0	1
2741	0	1	0	1
2743	0	1	0	1
2755	0	1	0	1
2756	1	1	0	1
2758	0	0	0	1
2759	0	1	0	1
2761	0	0	0	1
2765	0	0	0	1
2767	0	0	0	1
2771	0	0	0	1
2773	0	0	0	1
2774	0	1	0	1
2775	0	0	0	1
2777	0	1	0	1
2779	0	1	0	1
2780	0	1	0	1
2783	0	0	0	1
2785	1	1	0	1
2789	0	0	0	1
2792	1	1	0	1
2793	0	1	0	1
2797	0	0	0	1
2799	0	1	0	1
3504	0	1	0	1
3506	0	1	0	1
3510	0	1	0	1
3512	0	1	0	1
3516	0	1	0	1
3519	0	1	0	1

4. Following Table A4 - 7, reclassify crop typologies from *BRP* into a *Coccinellids habitat (BRP), BRP_{cocci}*, layer displaying the presence of coccinellids within each cell (0 = not present, 1 = present).
5. Following Table A4 - 7, reclassify crop typologies from *BRP* into a *Carabids habitat (BRP), BRP_{cara}*, layer displaying the presence of carabids within each cell (0 = not present, 1 = present).

A4 – 4.4.3.2 Creating habitat layers for field margins

Table A4 - 8 is based on expert elicitation (scientists) and it displays the potential of different field-margin typologies to act as habitat for aphids, hoverflies, coccinellids, and carabids. A value of 1 suggests that the specified typology is suitable as habitat for a specified insect population, while a value of 0 suggests it is unsuitable for this purpose.

6. Following Table A4 - 8, reclassify the field margin classes from *FM* into a *Hoverflies habitat (FM), FM_{hover}*, layer displaying the presence of hoverflies within each cell (0 = not present, 1 = present).
7. Following Table A4 - 8, reclassify the field margin classes from *FM* into a *Coccinellids habitat (FM), FM_{cocci}*, layer displaying the presence of coccinellids within each cell (0 = not present, 1 = present).
8. Following Table A4 - 8, reclassify the field margin classes from *FM* into a *Carabids habitat (FM), FM_{cara}*, layer displaying the presence of carabids within each cell (0 = not present, 1 = present).

Table A4 - 8: Look-up table for reclassifying areas where field margins are present into potential habitats for pest populations (aphids) and natural enemy populations (hoverflies, coccinellids, carabids)

Layer source	Code	Land cover class	Aphids (Akker)	Hover-flies (Akker)	Cocci-nellids (Akker)	Cara-bids (Akker)
Akkerranden	1	New grass margin	1	1	1	1
Akkerranden	2	Grass margin, 2+ years	1	1	1	1
Akkerranden	3	Grass margin	1	0	1	0
Akkerranden	4	Flower margin	1	0	1	1
Akkerranden	5	Combined: Grass (2+ years), flower, winter bird margin	1	1	1	1
Akkerranden	6	Winter bird field margin	1	0	1	1
Akkerranden		Combined: New grass, flower, winter bird margins	1	1	1	1
Akkerranden		Combined: grass, flower, winter bird margins	1	0	1	1

A4 – 4.4.3.3 Creating habitat layers for woody vegetation

Table A4 - 9 is based on expert elicitation (scientists) and it displays the potential of woody vegetation to act as habitat for aphids, hoverflies, coccinellids, and carabids. A value of 1 suggests that woody vegetation is suitable as habitat for a specified insect population, while a value of 0 suggests it is unsuitable for this purpose.

Table A4 - 9: Look-up table for reclassifying areas where woody vegetation is present into potential habitats for pest populations (aphids) and natural enemy populations (hoverflies, coccinellids, carabids)

Layer source	Land cover class	Aphids (woody)	Hoverflies (woody)	Coccinellids (woody)	Carabids (woody)
Woody vegetation	Woody vegetation	1	1	1	1

9. Vegetation layers at a 10 m resolution are created, conform Appendix 3 – 2 (Section A2 – 2.1.2.1), using the *AHN2, BAG*, *Luchtfoto,* and *BRP* layers as input. Vegetation layers display the *Fraction per grid cell (0-1) covered by trees (fr_{tree})*, *bushes/shrubs (fr_{shrub})*, and *low vegetation ($fr_{loweveg}$)*.
10. Create a layer displaying the *Fraction of a cell covered by woody vegetation (0-1), fr_{woody},* by implementing Formula A4 - 1. *fr_{tree}* and *fr_{shrub}* consider the fraction of a cell covered by a particular vegetation, which is why their summed value should not exceed a value of 1.

$$fr_{woody} = fr_{tree} + fr_{shrub}$$

Formula A4 - 1

11. Only cells with a woody percentage cover higher than 10% (0.1) are considered as woody habitat. Hence, $fr_{woody\ i}$s reclassified so that any cell containing a woody cover higher than 10% are assigned a value of 1.
12. Since, based on Table A4 - 9, woody vegetation is suitable as habitat for aphids, *fr_{woody}* is reclassified into a layer displaying the *Distribution of aphids, $woody_{aphid}$,* containing identical values.
13. Since, based on Table A4 - 9, woody vegetation is suitable as habitat for hoverflies, *fr_{woody}* is reclassified into a layer displaying the *Distribution of hoverflies, $woody_{hover}$,* containing identical values.
14. Since, based on Table A4 - 9, woody vegetation is suitable as habitat for coccinellids, *fr_{woody}* is reclassified into a layer displaying the *Distribution of coccinellids, $woody_{cocci}$,* containing identical values.
15. Since, based on Table A4 - 9, woody vegetation is suitable as habitat for carabids, *fr_{woody}* is reclassified into a layer displaying the *Distribution of carabids, $woody_{cara}$,* containing identical values.

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A4 – 4.4.3.4 Creating habitat layers considering insect mobility

Table A4 - 10 is based on expert elicitation (scientists) and it displays the mobility of aphids, hoverflies, coccinellids, and carabids within a radius of vegetation typologies that act as habitat for them.

Table A4 - 10: Mobility factor that determines the mobility of each insect species, measured as the number of cells (10 x 10 m) that it can move within a radius of its habitat

Insect	Insect type	Mobility radius (number of cells)
Aphids	Pest	0
Hoverflies	Natural enemy	10
Coccinellids	Natural enemy	15
Carabids	Natural enemy	2

16. Create a *Combined aphid habitat, HA_{aphid},* layer by adding the FM_{aphid}, $Woody_{aphid}$, and BRP_{aphid} layers, and making sure that the maximum value for all cells is equal to 1.
17. Create a *Combined hoverflies habitat, CO_{hover},* layer by adding the FM_{hover}, $Woody_{hover}$, and BRP_{hover} layers, and making sure that the maximum value for all cells is equal to 1.
18. Based on Table A4 - 10, an A*djusted hoverflies habitat, HA_{hover},* layer is created, which extends the habitat of hoverflies based on a buffer (radius) from a vegetation typology in which an insect can be present, based on its mobility factor (10 cells). All values within that buffer will receive a value of 1 (= present).
19. Create a *Combined coccinellids habitat, CO_{cocci},* layer by adding the FM_{cocci}, $Woody_{cocci}$, and BRP_{cocci} layers, and making sure that the maximum value for all cells is equal to 1.
20. Based on Table A4 - 10, an A*djusted coccinellids habitat, HA_{cocci},* layer is created, which extends the habitat of coccinellids based on a buffer (radius) from a vegetation typology in which an insect can be present, based on its mobility factor (15 cells). All values within that buffer will receive a value of 1 (= present).
21. Create a *Combined carabids habitat, CO_{cara},* layer by adding the FM_{cara}, $Woody_{cara}$, and BRP_{cara} layers, and making sure that the maximum value for all cells is equal to 1.
22. Based on Table A4 - 10, an A*djusted carabids habitat, HA_{cara},* layer is created, which extends the habitat of carabids based on a buffer (radius) from a vegetation typology in which an insect can be present, based on its mobility factor (2 cells). All values within that buffer will receive a value of 1 (= present).

A4 – 4.4.3.5 Calculating the effective pest (aphid) control by natural enemies

Table A4 - 11 is based on expert elicitation (scientists) and it displays the effectiveness of pest control given the presence of aphids and its natural enemies (carabids, hoverflies, coccinellids). The effectiveness of pest control will vary (0-1.5) depending on intra-guild interactions that may occur based on different insect combinations interacting in a cell. Areas assigned a score of 0.75 are habitat to aphids, hoverflies and coccinellids, as the presence of hoverflies and coccinellids may lead to intraguild predation and hence reduce their effectiveness in aphid suppression (Alhmedi et al., 2010). Areas assigned a score of 1.5 are habitat to aphids, coccinellids, and carabids, as coccinellids and carabids may interact synergistically to enhance aphid predation (Hindayana et al., 2001).

23. Create a layer displaying the *Effective pest control scores (0 - 1.5)*, by reclassifying the layers HA_{aphid}, HA_{hover}, HA_{cocci}, and HA_{cara}, based on Table A4 - 11.

Table A4 - 11: Effective pest (aphid) control score based on the presence of natural enemies within a particular cell, where 0 = no effectiveness and 1.5 = is the highest attainable effectiveness level. For different insect populations, 0 = not present, 1 = present.

Pest control effectiveness	Aphids	Hoverflies	Coccinellids	Carabids
0.00	0	0	0	0
0.00	0	1	0	0
0.00	0	1	1	0
0.00	0	1	1	1
0.00	0	0	1	0
0.00	0	0	1	1
0.00	0	0	0	1
0.75	1	1	1	0
1.00	1	1	0	0
1.00	1	0	1	0
1.00	1	0	0	1
1.25	1	1	0	1
1.50	1	0	1	1
1.00	1	1	1	1

A4 – 4.5: 'Soil Biodiversity' model specifications

This Section describes the process required for modelling ecosystem service indicators presented in this study by use of the 'Soil Biodiversity' model. Provided below is a detailed explanation of the way the model was adapted, and the steps required for its implementation.

A4 – 4.5.1 Spatial data requirements

- Agricultural crop parcels (BRP)
- Land use map of the Netherlands (EUM)
- Soil biophysical units (BOFEK2012)
- Soil (biological, physical, and chemical) characteristics (BoBi)
    - Earthworm richness
    - Earthworm abundance
    - Nematode richness
    - Nematode abundance
    - Enchytraeid richness
    - Enchytraeid abundance
    - Microarthropod richness
    - Microarthropod abundance
    - Bulk density
    - pH
    - Total C
    - Total N
    - Soil organic matter

A4 – 4.5.2 Non-spatial data requirements

- Expert-based thresholds values for assigning scores to measured attributes
- Expert-based look-up tables for integrating rules for integrating measurable soil biotic and abiotic parameters (Soil Navigator, 2020)

A4 – 4.5.3 Model mechanism

This model is an adaptation of the Soil Navigator (http://www.soilnavigator.eu/; Van Leeuwen et al., 2019), an expert-based decision-tree approach for quantifying the performance soil functions. The Soil Navigator comprises two models for quantifying the performance of soil biodiversity – one for grasslands and one for croplands (Van Leeuwen et al., 2019). These models were adapted into an integrated spatial model as described below.

A4 – 4.5.3.1 Cropland model

Figure A4 - 1 presents a decision-tree illustrating how attribute values in tier 6 are grouped into classes at higher tier levels.

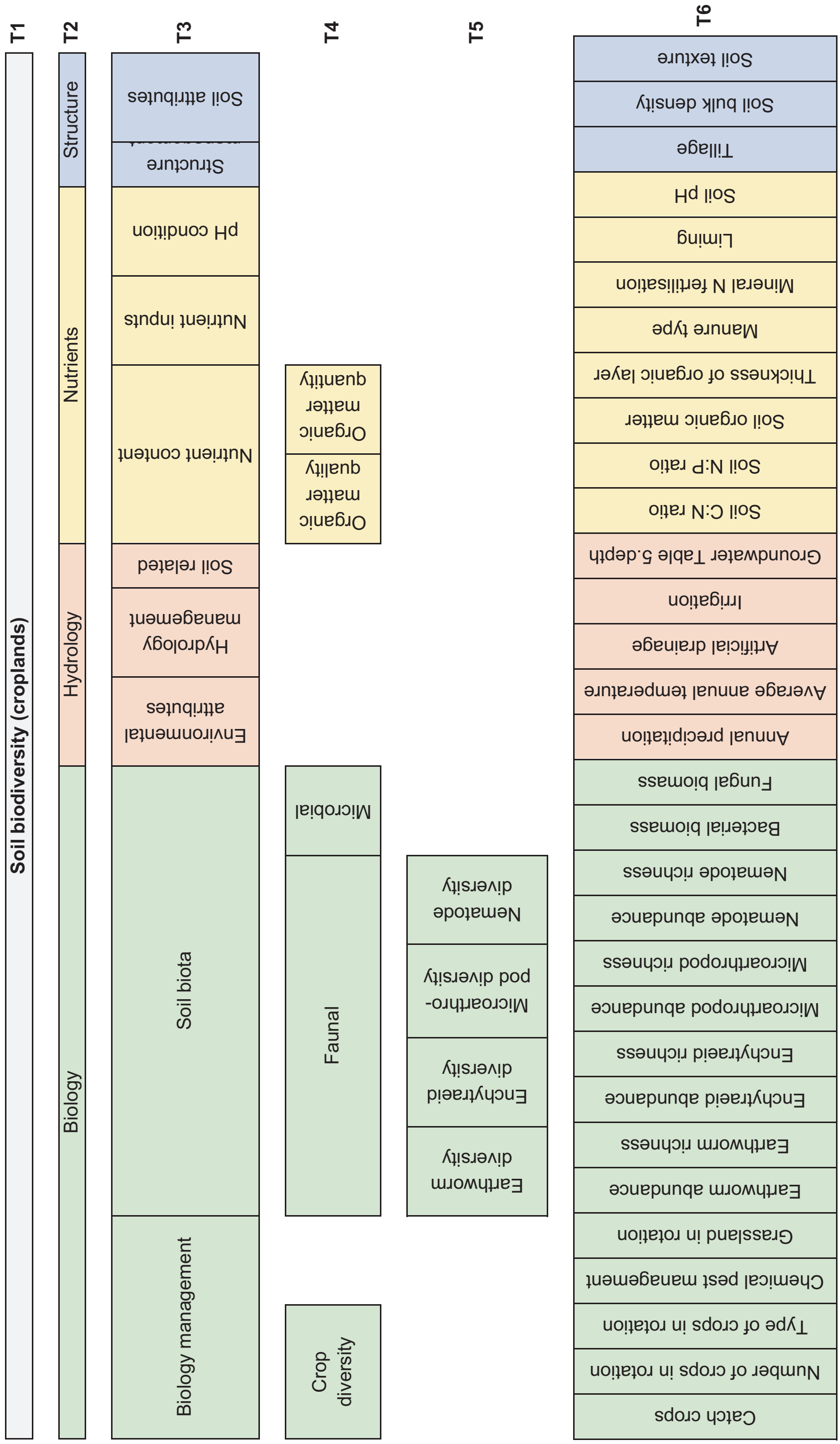

Figure A4 - 1: Decision tree for the classification of attributes within the cropland model into classes at higher tier levels, by use of look-up tables (Soil Navigator, 2020)

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Tier 6

Table A4 - 12 presents threshold values from the Soil Navigator that reclassify attribute values in tier 6 into performance scores. Performance scores were assigned numerical values (i.e., 1, 2, or 3) for integration within look-up tables for calculating scores at higher levels. Table A4 - 13 provides information on the type of data that was implemented to represent each attribute. Where the use of spatial data was not necessary (e.g., due to low spatial heterogeneity), attributes were assigned reference scores (e.g., overall poor, moderate, good performance). Where spatial data was not available, attributes were assigned a reference score (i.e., moderate), following the mechanism behind the Soil Navigator.

*Table A4 - 12: Attributes, descriptions, and threshold values that determine the performance score assigned to attributes (1, 2, or 3) within the cropland model (source: http://www.soilnavigator.eu/). * = excluding cover crops/green manure following the main crop in the present year.*

Attributes (Tier 6)	Description	1	2	3
Annual precipitation	Average yearly precipitation	<650	650-900	>950
Artificial drainage	Presence of tile drains, ditches, furrows or pipes	no	yes	
Average annual temperature	Average yearly temperature	<9	9-11	>11
Bacterial biomass	Bacterial biomass	<50	50-100	>100
Catch crops	Share of catch crops/cover crops/green manure (last 5 years *)	0	1-3 times in last 5 years	4+ times in last five years
Chemical pest management	Application of chemical pesticides	no	yes	
Earthworm abundance	Number of earthworms per m²	<100	100-200	>200
Earthworm richness	Number of earthworm species per 100 individuals	<3	3-5	>5
Enchytraeid abundance	Number of enchytraeids per m²	<3000	3000-30000	>30000
Enchytraeid richness	Number of enchytraeids species per m²	<6	6-12	>12
Fungal biomass	Fungal biomass	<20	20-50	>50
Grassland in rotation	Inclusion of grassland in rotation	no	yes	
Groundwater table depth	Groundwater table depth	<0.4	0.4-2.0	>2.0
Irrigation	Presence of sprinklers, drippers or ditches for providing water	no	yes	
Liming	Has liming been applied to the field during the past 5 years?	no	yes	

Manure type	Type of manure applied (slurry, manure, compost etc.)	no manure	slurry, sludge	solid manure, compost
Microarthropod abundance	Number of microarthropods per m^2	<10000	10000-20000	>20000
Microarthropod richness	Number of microarthropod families present	<10	10-20	>20
Mineral N fertilisation	Amount of plant-available N applied per ha per year	<50	50-100	>100
Nematode abundance	Number of nematodes per 100 g fresh soil	<1500	1500-3000	>3000
Nematode richness	Number of nematode genera per 150 individuals	<25	25-35	>35
Number of crops in rotation	Number of crop types in rotation	<3	3-5	>5
Soil bulk density	Soil bulk density	<1.1	1.1-1.5	>1.50
Soil C:N ratio	Soil C:N ratio	<10	10-30	>30
Soil N:P ratio	Soil N:P ratio	<10	10-20	>20
Soil organic matter	Soil organic matter content in the topsoil	<2	2-5	>5
Soil pH	Soil pH, measured as pH (KCl)	<4.5, >7.2	4.5-5.5, 6.5-7.2	5.5-6.5
Soil texture	3 classes: WRB classification system	fine: clay	medium: Loam	coarse: sand
Thickness of organic layer	Thickness of organic layer (A horizon)	<10	10-20	>20
Tillage	No tillage, non-inversion or intermittent tillage, or conventional tillage	no tillage	non-inversion, intermittent	conventional tillage
Type of crops in rotation	Cash crops, grass or grains, legumes, crop mixtures and intercropping	cash crops	grass or grains	legumes, crop mixtures, intercropping

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Table A4 - 13: Attributes and type of input data implemented for their representation (reference value or spatial data). Where reference values were used as input, the score assigned to a particular attribute is presented in column 3 (Ref. value). Where spatial data was used as input, the dataset source is stated in column 4 (Spatial data source).

Attributes (Tier 6)	Data type	Reference value	Spatial data source
Annual precipitation	reference value	2	
Artificial drainage	reference value	2	
Average annual temperature	reference value	2	
Bacterial biomass	reference value	2	
Catch crops	reference value	2	
Chemical pest management	reference value	2	
Earthworm abundance	spatial data		Soil characteristics
Earthworm richness	spatial data		Soil characteristics
Enchytraeid abundance	spatial data		Soil characteristics
Enchytraeid richness	spatial data		Soil characteristics
Fungal biomass	reference value	2	
Grassland in rotation	reference value	2	
Groundwater table depth	reference value	1	
Irrigation	reference value	2	
Liming	reference value	2	
Manure type	reference value	2	
Microarthropod abundance	spatial data		Soil characteristics
Microarthropod richness	spatial data		Soil characteristics
Mineral N fertilisation	reference value	2	
Nematode abundance	spatial data		Soil characteristics
Nematode richness	spatial data		Soil characteristics
Number of crops in rotation	reference value	1	
Soil bulk density	spatial data		Soil characteristics
Soil C:N ratio	spatial data		Soil characteristics
Soil N:P ratio	reference value	1	
Soil organic matter	spatial data		Soil characteristics
Soil pH	spatial data		Soil characteristics
Soil texture	spatial data		Soil biophysical units
Thickness of organic layer	reference value	1	
Tillage	reference value	3	
Types of crops in rotation	spatial data		

Tier 5

Hereunder, look-up tables that were implemented to group attributes in tier 6 into overarching classes in tier 5.

Table A4 - 14: Look-up table for reclassifying attribute performance scores into performance scores for the 'Earthworm diversity' class in tier 5

Input performance scores		**Output performance scores**
Earthworm richness	**Earthworm abundance**	**Earthworm diversity**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 15: Look-up table for reclassifying attribute performance scores into performance scores for the 'Enchytraeid diversity' class in tier 5

Input performance scores		**Output performance scores**
Enchytraeid richness	**Enchytraeid abundance**	**Enchytraeid diversity**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 16: Look-up table for reclassifying attribute performance scores into performance scores for the 'Microarthropod diversity' class in tier 5

Input performance scores		Output performance scores
Microarthropod richness	**Microarthropod abundance**	**Microarthropod diversity**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 17: Look-up table for reclassifying attribute performance scores into performance scores for the 'Nematode diversity' class in tier 5

Input performance scores		Output performance scores
Nematode richness	**Nematode abundance**	**Nematode diversity**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Tier 4

Hereunder, look-up tables that were implemented to group attributes and classes into overarching classes in tier 4.

Table A4 - 18: Look-up table for reclassifying attribute performance scores into performance scores for the 'Crop diversity' class in tier 4

Input performance scores			Output performance scores
Number if crops in rotation	**Type of crops in rotation**	**Catch crops**	**Crop diversity**
1	1	1	1
1	1	2	1
1	1	3	1
2	1	1	1
2	1	2	2
2	1	3	2
3	1	1	1
3	1	2	2
3	1	3	2
1	2	1	1
1	2	2	2
1	2	3	2
2	2	1	1
2	2	2	2
2	2	3	3
3	2	1	2
3	2	2	2
3	2	3	3
1	3	1	1
1	3	2	2
1	3	3	3
2	3	1	2
2	3	2	3
2	3	3	3
3	3	1	3
3	3	2	3
3	3	3	3

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Table A4 - 19: Look-up table for reclassifying attribute performance scores into performance scores for the 'Faunal' class in tier 4

Input performance scores				Output performance scores
Earthworm diversity	**Nematode diversity**	**Microarthropod diversity**	**Enchytraeid diversity**	**Faunal**
1	1	1	1	1
1	1	1	2	1
1	1	1	3	1
1	1	2	1	1
1	1	2	2	1
1	1	2	3	1
1	1	3	1	1
1	1	3	2	1
1	1	3	3	2
1	2	1	1	1
1	2	1	2	1
1	2	1	3	1
1	2	2	1	1
1	2	2	2	1
1	2	2	3	2
1	2	3	1	1
1	2	3	2	2
1	2	3	3	3
1	3	1	1	1
1	3	1	2	1
1	3	1	3	2
1	3	2	1	1
1	3	2	2	2
1	3	2	3	3
1	3	3	1	2
1	3	3	2	3
1	3	3	3	3
2	1	1	1	1
2	1	1	2	1
2	1	1	3	1
2	1	2	1	1
2	1	2	2	2
2	1	2	3	2
2	1	3	1	1
2	1	3	2	2
2	1	3	3	3
2	2	1	1	1
2	2	1	2	2
2	2	1	3	2
2	2	2	1	2
2	2	2	2	2
2	2	2	3	2
2	2	3	1	2
2	2	3	2	2
2	2	3	3	3

2	3	1	1	1
2	3	1	2	2
2	3	1	3	3
2	3	2	1	2
2	3	2	2	2
2	3	2	3	3
2	3	3	1	3
2	3	3	2	3
2	3	3	3	3
3	1	1	1	1
3	1	1	2	1
3	1	1	3	2
3	1	2	1	1
3	1	2	2	2
3	1	2	3	3
3	1	3	1	2
3	1	3	2	3
3	1	3	3	3
3	2	1	1	1
3	2	1	2	2
3	2	1	3	3
3	2	2	1	2
3	2	2	2	2
3	2	2	3	3
3	2	3	1	3
3	2	3	2	3
3	2	3	3	3
3	3	1	1	2
3	3	1	2	3
3	3	1	3	3
3	3	2	1	3
3	3	2	2	3
3	3	2	3	3
3	3	3	1	3
3	3	3	2	3
3	3	3	3	3

Table A4 - 20: Look-up table for reclassifying attribute performance scores into performance scores for the 'Microbial' class in tier 4

Input performance scores		**Output performance scores**
Bacterial biomass	**Fungal biomass**	**Microbial**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 21: Look-up table for reclassifying attribute performance scores into performance scores for the 'Organic matter quality' class in tier 4

Input performance scores		**Output performance scores**
Ratio C:N	**Ratio N:P**	**Organic matter quality**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	2
3	1	2
3	2	3
3	3	3

Table A4 - 22: Look-up table for reclassifying attribute performance scores into performance scores for the 'Organic matter quantity' class in tier 4

Input performance scores		**Output performance scores**
Soil organic matter	**Thickness of organic layer**	**Organic matter quantity**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Tier 3

Hereunder, look-up tables that were implemented to group attributes and classes into overarching classes in tier 3.

Table A4 - 23: Look-up table for reclassifying attribute performance scores into performance scores for the 'Biology management' class in tier 3

Input performance scores			Output performance scores
Grassland in rotation	**Crop diversity**	**Chemical pest management**	**Biology management**
2	1	2	1
2	1	1	1
2	2	2	1
2	2	1	1
2	3	2	2
2	3	1	3
1	1	2	1
1	1	1	2
1	2	2	2
1	2	1	3
1	3	2	2
1	3	1	3

Table A4 - 24: Look-up table for reclassifying attribute performance scores into performance scores for the 'Soil biota' class in tier 3]

Input performance scores		Output performance scores
Faunal	**Microbial**	**Soil biota**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 25: Look-up table for reclassifying attribute performance scores into performance scores for the 'Environmental attributes' class in tier 3

Input performance scores		Output performance scores
Average annual temperature	Annual precipitation	Environmental attributes
1	1	1
1	2	2
1	3	2
2	1	1
2	2	3
2	3	3
3	1	1
3	2	2
3	3	3

Table A4 - 26: Look-up table for reclassifying attribute performance scores into performance scores for the 'Hydrology management' class in tier 3

Input performance scores		Output performance scores
Irrigation	Artificial drainage	Hydrology management
2	2	3
2	1	2
1	2	2
1	1	1

Table A4 - 27: Look-up table for reclassifying attribute performance scores into performance scores for the 'Soil related' class in tier 3

Input performance scores	Output performance scores
Groundwater table depth	Soil related
1	1
2	2
3	3

Table A4 - 28: Look-up table for reclassifying attribute performance scores into performance scores for the 'Organic matter quantity' class in tier 3

Input performance scores		Output performance scores
Organic matter quantity	Organic matter quality	Nutrient content
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 29: Look-up table for reclassifying attribute performance scores into performance scores for the 'Nutrient inputs' class in tier 3

Input performance scores		Output performance scores
Mineral N fertilisation	**Manure Type**	**Nutrient inputs**
1	1	2
1	2	3
1	3	3
2	1	1
2	2	2
2	3	3
3	1	1
3	2	1
3	3	2

Table A4 - 30: Look-up table for reclassifying attribute performance scores into performance scores for the 'pH condition' class in tier 3

Input performance scores		Output performance scores
Liming	**Soil pH**	**pH condition**
2	1	2
2	1	1
2	2	3
2	2	2
2	3	3
1	1	1
1	1	1
1	2	2
1	2	2
1	3	3

Table A4 - 31: Look-up table for reclassifying attribute performance scores into performance scores for the 'Structure management' class in tier 3

Input performance scores	Output performance scores
Tillage	**Structure management**
3	1
2	2
1	3

Table A4 - 32: Look-up table for reclassifying attribute performance scores into performance scores for the 'Soil attributes' class in tier 3

Input performance scores		Output performance scores
Soil texture	**Soil bulk Density**	**Soil attributes**
1	1	3
1	2	3
1	3	2
2	1	3
2	2	2
2	3	1
3	1	2
3	2	1
3	3	1

Tier 2

Hereunder, look-up tables that were implemented to group attributes and classes into overarching classes in tier 2.

Table A4 - 33: Look-up table for reclassifying attribute performance scores into performance scores for the 'Biology' class in tier 2

Input performance scores		Output performance scores
Soil biota	**Biology management**	**Biology**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 34: Look-up table for reclassifying attribute performance scores into performance scores for the 'Hydrology' class in tier 2

Input performance scores			Output performance scores
Environmental attributes	**Soil related**	**Hydrology management**	**Hydrology**
1	1	1	1
1	1	2	1
1	1	3	1
1	2	1	1
1	2	2	1
1	2	3	2
1	3	1	1
1	3	2	2
1	3	3	3
2	1	1	1
2	1	2	1
2	1	3	2
2	2	1	1
2	2	2	2
2	2	3	3
2	3	1	2
2	3	2	3
2	3	3	3
3	1	1	1
3	1	2	2
3	1	3	3
3	2	1	2
3	2	2	3
3	2	3	3
3	3	1	3
3	3	2	3
3	3	3	3

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Table A4 - 35: Look-up table for reclassifying attribute performance scores into performance scores for the 'Nutrients' class in tier 2

Input performance scores			Output performance scores
Nutrient content	**Nutrient input**	**pH content**	**Nutrients**
1	1	1	1
1	1	2	1
1	1	3	1
1	2	1	1
1	2	2	1
1	2	3	2
1	3	1	1
1	3	2	2
1	3	3	3
2	1	1	1
2	1	2	2
2	1	3	2
2	2	1	1
2	2	2	2
2	2	3	3
2	3	1	2
2	3	2	2
2	3	3	3
3	1	1	2
3	1	2	2
3	1	3	2
3	2	1	2
3	2	2	3
3	2	3	3
3	3	1	3
3	3	2	3
3	3	3	3

Table A4 - 36: Look-up table for reclassifying attribute performance scores into performance scores for the 'Structure' class in tier 2

Input performance scores		Output performance scores
Soil attributes	**Structure management**	**Structure**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Tier 1

Hereunder, look-up tables that were implemented to group attributes and classes into overarching one overarching class in tier 1.

Table A4 - 37: Look-up table for reclassifying attribute performance scores into performance scores for the 'Soil biodiversity' class in tier 1

Input performance scores				Output performance scores
Nutrients	**Biology**	**Structure**	**Hydrology**	**Soil biodiversity**
1	1	1	1	1
1	1	1	2	1
1	1	1	3	1
1	1	2	1	1
1	1	2	2	1
1	1	2	3	1
1	1	3	1	1
1	1	3	2	1
1	1	3	3	2
1	2	1	1	1
1	2	1	2	1
1	2	1	3	1
1	2	2	1	1
1	2	2	2	2
1	2	2	3	2
1	2	3	1	2
1	2	3	2	2
1	2	3	3	2
1	3	1	1	1
1	3	1	2	2
1	3	1	3	2
1	3	2	1	2
1	3	2	2	2
1	3	2	3	2
1	3	3	1	2
1	3	3	2	3
1	3	3	3	3
2	1	1	1	1
2	1	1	2	1
2	1	1	3	2
2	1	2	1	1
2	1	2	2	2
2	1	2	3	2
2	1	3	1	2
2	1	3	2	2
2	1	3	3	2
2	2	1	1	2
2	2	1	2	2
2	2	1	3	2
2	2	2	1	2
2	2	2	2	2
2	2	2	3	3

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2	2	3	1	2
2	2	3	2	3
2	2	3	3	3
2	3	1	1	2
2	3	1	2	2
2	3	1	3	3
2	3	2	1	3
2	3	2	2	3
2	3	2	3	3
2	3	3	1	3
2	3	3	2	3
2	3	3	3	3
3	1	1	1	2
3	1	1	2	2
3	1	1	3	2
3	1	2	1	2
3	1	2	2	2
3	1	2	3	3
3	1	3	1	3
3	1	3	2	3
3	1	3	3	3
3	2	1	1	2
3	2	1	2	3
3	2	1	3	3
3	2	2	1	3
3	2	2	2	3
3	2	2	3	3
3	2	3	1	3
3	2	3	2	3
3	2	3	3	3
3	3	1	1	3
3	3	1	2	3
3	3	1	3	3
3	3	2	1	3
3	3	2	2	3
3	3	2	3	3
3	3	3	1	3
3	3	3	2	3
3	3	3	3	3

A4 – 4.5.3.2 Grassland model mechanism

Figure A4 - 2 presents a decision-tree illustrating how attribute values in tier 6 are grouped into classes at higher tier levels.

Tier 6

Table A4 - 38 presents threshold values from the Soil Navigator that reclassify attribute values in tier 6 into performance scores. Performance scores were assigned numerical values (i.e., 1, 2, or 3) for integration within look-up tables for calculating scores at higher levels. Table A4 - 39 provides information on the on the type of data that was implemented to represent each attribute. Where the use of spatial data was not necessary (e.g., due to low spatial heterogeneity), attributes were assigned reference scores (e.g., overall poor, moderate, good performance). Where spatial data was not available, attributes were assigned a reference score (i.e., moderate), following the mechanism behind the Soil Navigator.

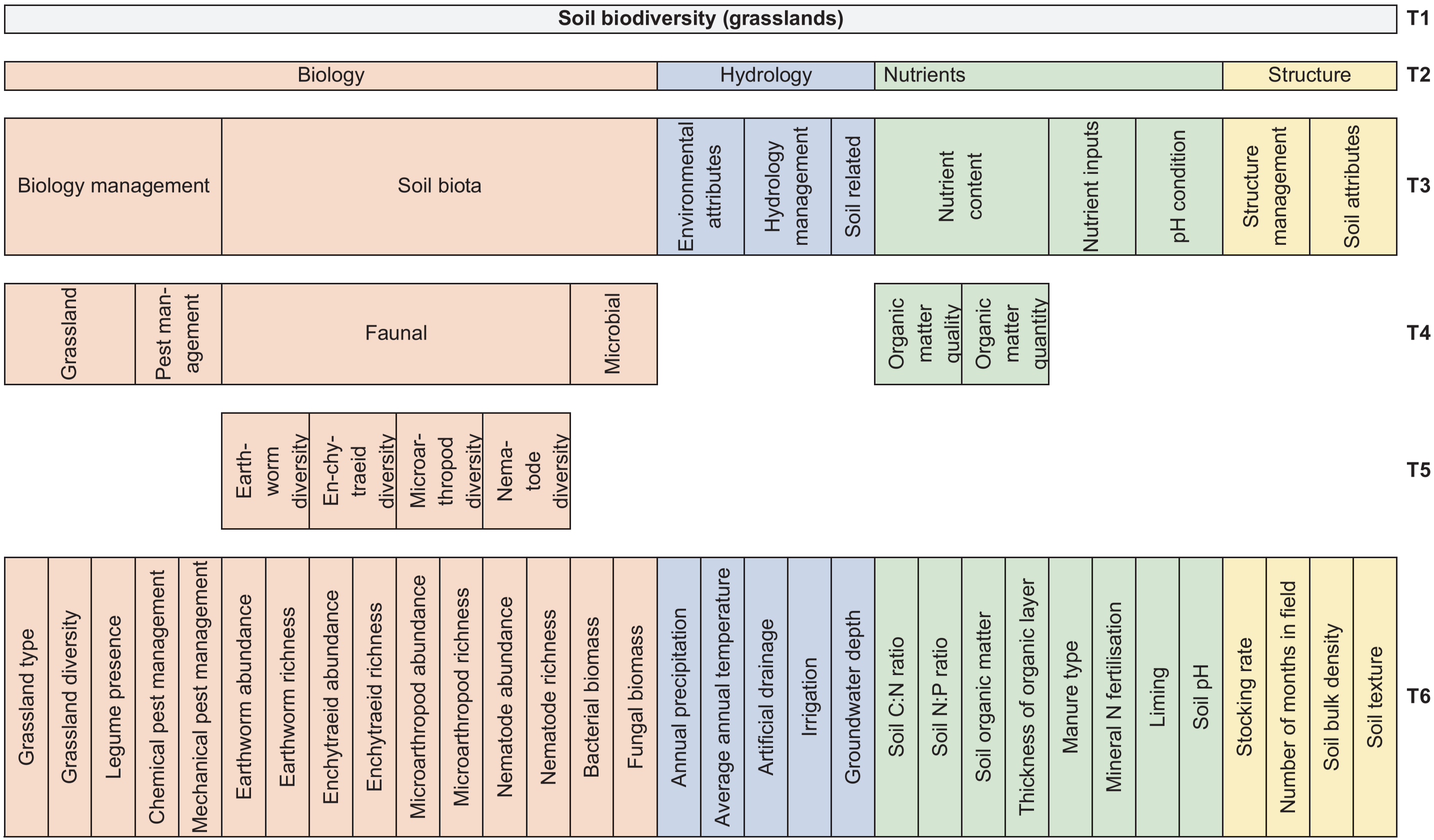

Figure A4 - 2: Decision tree for the classification of attributes within the grassland model into classes at higher tier levels, by use of look-up tables (Soil Navigator, 2020)

Table A4 - 38: Attributes, descriptions, and threshold values that determine the performance score assigned to attributes (1, 2, or 3) within the grassland model (source: http://www.soilnavigator.eu/). GL = grassland.

Attributes (Tier 6)	Description	1	2	3
Annual precipitation	Average yearly precipitation	<650	650-900	>950
Artificial drainage	Presence of tile drains, ditches, furrows or pipes	no	yes	
Average annual temperature	Average yearly temperature	<9	9-11	>11
Bacterial biomass	Bacterial biomass	<50	50-100	>100
Chemical pest management	Chemical pest management applied	no	yes	
Earthworm abundance	Number of earthworms per m^2	<100	100-200	>200
Earthworm richness	Number of earthworm species per 100 individuals	<3	3-5	>5
Enchytraeid abundance	Number of enchytraeids per m^2	<3000	3000-30,000	>30,000
Enchytraeid richness	Number of enchytraeids species per m^2	<6	6-12	>12
Fungal biomass	Fungal biomass	<20	20-50	>50
GL diversity	Number of grass/herb species sown	1	2	>2
GL type	GL type current	reseeded GL	permanent GL	
Groundwater table depth	Groundwater table depth	<0.4	0.4-2.0	>2.0
Irrigation	Presence of sprinklers, drippers or ditches for providing water	no	yes	
Legume presence	Percentage of legumes in GL	<10% of legumes in GL	>10% of legumes in GL	
Liming	Has liming been applied to the field during the past 5 years?	no	yes	
Manure type	Type of manure applied (slurry, manure, compost etc.)	no manure	slurry, sludge	solid manure, compost
Mechanical pest management	Mechanical pest management applied	no	yes	
Microarthropod abundance	Number of microarthropods per m^2	<10,000	10,000-20,000	>20,000
Microarthropod richness	Number of microarthropod families present	<10	10-20	>20
Mineral N fertilisation	Amount of plant-available N applied per ha per year	<50	50-100	>100
Nematode abundance	Number of nematodes per 100 g fresh soil	<1500	1500-3000	>3000
Nematode richness	Number of nematode genera per 150 individuals	<25	25-35	>35

Number of months in fields	Number of months in fields in winter (per year)	<4	4-8	>8
Soil bulk density	Soil bulk density	<1.1	1.1-1.5	>1.50
Soil C:N ratio	Soil C:N ratio	<10	10-30	>30
Soil N:P ratio	Soil N:P ratio	<10	10-20	>20
Soil organic matter	Soil organic matter content in the topsoil	<2	2-5	>5
Soil pH	Soil pH, measured as pH (KCl)	<4.5, >7.2	4.5-5.5, 6.5-7.2	5.5-6.5
Soil texture	3 classes: WRB classification system	clay	loam	coarse: sand
Stocking rate	Number of Livestock units per hectare per year	<1LSU/ha	1-2.5LSU/ha	>2.5LSU/ha
Thickness of organic layer	Thickness of organic layer	<10	10-20	>20

Table A4 - 39: Attributes and type of input data implemented for their representation (reference value or spatial data). Where reference values were used as input, the score assigned to a particular attribute is presented in column 3 (Ref. value). Where spatial data was used as input, the dataset source is stated in Column 4 (Spatial data source).

Attributes (tier 6)	Data type	Reference value	Spatial data source
Annual precipitation	reference value	2	
Artificial drainage	reference value	2	
Average annual temperature	reference value	2	
Bacterial biomass	reference value	2	
Chemical pest management	reference value	2	
Earthworm abundance	spatial data		Soil characteristics
Earthworm richness	spatial data		Soil characteristics
Enchytraeid abundance	spatial data		Soil characteristics
Enchytraeid richness	spatial data		Soil characteristics
Fungal biomass	reference value	2	
Grassland diversity	reference value	2	
Grassland type	reference value	2	
Groundwater table depth	reference value	1	Soil characteristics
Irrigation	reference value	2	
Legume presence	reference value	2	
Liming	reference value	2	
Manure type	reference value	2	
Mechanical pest management	reference value	2	
Microarthropod abundance	spatial data		Soil characteristics
Microarthropod richness	spatial data		Soil characteristics
Mineral N fertilisation	reference value	2	
Nematode abundance	spatial data		Soil characteristics
Nematode richness	spatial data		Soil characteristics
Number of months in field	reference value	1	
Soil bulk density	spatial data		Soil characteristics
Soil C:N ratio	spatial data		Soil characteristics
Soil N:P ratio	reference value	1	Soil characteristics
Soil organic matter	spatial data		Soil characteristics
Soil pH	spatial data		Soil characteristics
Soil texture	spatial data		Soil biophysical units
Stocking rate	reference value	1	
Thickness of organic layer	reference value	1	

Tier 5

Hereunder, look-up tables that were implemented to group attributes in tier 6 into overarching classes in tier 5.

Table A4 - 40: Look-up table for reclassifying attribute performance scores into performance scores for the 'Earthworm diversity' class in tier 5

Input performance scores		Output performance scores
Earthworm richness	**Earthworm abundance**	**Earthworm diversity**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 41: Look-up table for reclassifying attribute performance scores into performance scores for the 'Enchytraeid diversity' class in tier 5

Input performance scores		Output performance scores
Enchytraeid richness	**Enchytraeid diversity**	**Enchytraeid diversity**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 42: Look-up table for reclassifying attribute performance scores into performance scores for the 'Microarthropod diversity' class in tier 5

Input performance scores		Output performance scores
Microarthropod richness	**Microarthropod abundance**	**Microarthropod diversity**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 43: Look-up table for reclassifying attribute performance scores into performance scores for the 'Nematode diversity' class in tier 5

Input performance scores		Output performance scores
Nematode richness	**Nematode abundance**	**Nematode diversity**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Tier 4

Hereunder, look-up tables that were implemented to group attributes and classes into overarching classes in tier 4.

Table A4 - 44: Look-up table for reclassifying attribute performance scores into performance scores for the 'Crop diversity' class in tier 4

Input performance scores			Output performance scores
Grassland type	**Grassland diversity**	**Legume presence**	**Grassland**
1	1	1	1
1	1	2	1
1	2	1	1
1	2	2	2
1	3	1	2
1	3	2	3
2	1	1	1
2	1	2	2
2	2	1	2
2	2	2	3
2	3	1	3
2	3	2	3

Table A4 - 45: Look-up table for reclassifying attribute performance scores into performance scores for the 'Pest management' class in tier 4

Input performance scores		Output performance scores
Chemical pest management	**Mechanical pest management**	**Pest management**
2	2	1
2	1	2
1	2	3
1	1	4

Table A4 - 46: Look-up table for reclassifying attribute performance scores into performance scores for the 'Faunal' class in tier 4

Input performance scores				Output performance scores
Earthworm diversity	Nematode diversity	Microarthropod diversity	Enchytraeid diversity	Faunal
1	1	1	1	1
1	1	1	2	1
1	1	1	3	1
1	1	2	1	1
1	1	2	2	1
1	1	2	3	1
1	1	3	1	1
1	1	3	2	1
1	1	3	3	2
1	2	1	1	1
1	2	1	2	1
1	2	1	3	1
1	2	2	1	1
1	2	2	2	1
1	2	2	3	2
1	2	3	1	1
1	2	3	2	2
1	2	3	3	3
1	3	1	1	1
1	3	1	2	1
1	3	1	3	2
1	3	2	1	1
1	3	2	2	2
1	3	2	3	3
1	3	3	1	2
1	3	3	2	3
1	3	3	3	3
2	1	1	1	1
2	1	1	2	1
2	1	1	3	1
2	1	2	1	1
2	1	2	2	2
2	1	2	3	2
2	1	3	1	1
2	1	3	2	2
2	1	3	3	3
2	2	1	1	1
2	2	1	2	2
2	2	1	3	2
2	2	2	1	2
2	2	2	2	2
2	2	2	3	2
2	2	3	1	2
2	2	3	2	2
2	2	3	3	3

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2	3	1	1	1
2	3	1	2	2
2	3	1	3	3
2	3	2	1	2
2	3	2	2	2
2	3	2	3	3
2	3	3	1	3
2	3	3	2	3
2	3	3	3	3
3	1	1	1	1
3	1	1	2	1
3	1	1	3	2
3	1	2	1	1
3	1	2	2	2
3	1	2	3	3
3	1	3	1	2
3	1	3	2	3
3	1	3	3	3
3	2	1	1	1
3	2	1	2	2
3	2	1	3	3
3	2	2	1	2
3	2	2	2	2
3	2	2	3	3
3	2	3	1	3
3	2	3	2	3
3	2	3	3	3
3	3	1	1	2
3	3	1	2	3
3	3	1	3	3
3	3	2	1	3
3	3	2	2	3
3	3	2	3	3
3	3	3	1	3
3	3	3	2	3
3	3	3	3	3

Table A4 - 47: Look-up table for reclassifying attribute performance scores into performance scores for the 'Microbial' class in tier 4

Input performance scores		Output performance scores
Bacterial biomass	**Fungal biomass**	**Microbial**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 48: Look-up table for reclassifying attribute performance scores into performance scores for the 'Organic matter quality' class in tier 4

Input performance scores		Output performance scores
Ratio C:N	**Ratio N:P**	**Organic matter quality**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 49: Look-up table for reclassifying attribute performance scores into performance scores for the 'Organic matter quantity' class in tier 4

Input performance scores		Output performance scores
Soil organic matter	**Thickness of organic layer**	**Organic matter quantity**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Tier 3

Hereunder, look-up tables that were implemented to group attributes and classes into overarching classes in tier 3.

Table A4 - 50: Look-up table for reclassifying attribute performance scores into performance scores for the 'Biology management' class in tier 3

Input performance scores		Output performance scores
Grassland	**Pest management**	**Biology management**
1	1	1
1	2	1
1	3	2
1	4	2
2	1	1
2	2	1
2	3	2
2	4	3
3	1	2
3	2	3
3	3	3
3	4	3

Table A4 - 51: Look-up table for reclassifying attribute performance scores into performance scores for the 'Soil biota' class in tier 3

Input performance scores		Output performance scores
Faunal	Microbial	Soil biota
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 52: Look-up table for reclassifying attribute performance scores into performance scores for the 'Environmental attributes' class in tier 3

Input performance scores		Output performance scores
Average annual temperature	Annual precipitation	Environmental attributes
1	1	1
1	2	2
1	3	2
2	1	1
2	2	3
2	3	3
3	1	1
3	2	2
3	3	3

Table A4 - 53: Look-up table for reclassifying attribute performance scores into performance scores for the 'Hydrology management' class in tier 3

Input performance scores		Output performance scores
Irrigation	Artificial drainage	Hydrology management
1	1	3
1	2	2
2	1	2
2	2	1

Table A4 - 54: Look-up table for reclassifying attribute performance scores into performance scores for the 'Soil related' class in tier 3

Input performance scores	Output performance scores
Groundwater table depth	Soil related
1	1
2	2
3	3

Table A4 - 55: Look-up table for reclassifying attribute performance scores into performance scores for the 'Nutrient content' class in tier 3

Input performance scores		Output performance scores
Organic matter quantity	**Organic matter quality**	**Nutrient content**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 56: Look-up table for reclassifying attribute performance scores into performance scores for the 'Nutrient inputs' class in tier 3

Input performance scores		Output performance scores
Mineral N fertilisation	**Manure type**	**Nutrient inputs**
1	1	2
1	2	3
1	3	3
2	1	1
2	2	2
2	3	3
3	1	1
3	2	1
3	3	2

Table A4 - 57: Look-up table for reclassifying attribute performance scores into performance scores for the 'pH condition' class in tier 3

Input performance scores		Output performance scores
Liming	**Soil pH**	**pH condition**
2	1	2
2	1	1
2	2	3
2	2	2
2	3	3
1	1	1
1	1	1
1	2	2
1	2	2
1	3	3

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Table A4 - 58: Look-up table for reclassifying attribute performance scores into performance scores for the 'Structure management' class in tier 3

Input performance scores		Output performance scores
Stocking rate	**Number of months in field**	**Structure management**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 59: Look-up table for reclassifying attribute performance scores into performance scores for the 'Soil attributes' class in tier 3

Input performance scores		Output performance scores
Soil texture	**Bulk density**	**Soil attributes**
1	1	3
1	2	3
1	3	2
2	1	3
2	2	2
2	3	1
3	1	2
3	2	1
3	3	1

Tier 2

Hereunder, look-up tables that were implemented to group attributes and classes into overarching classes in tier 2.

Table A4 - 60: Look-up table for reclassifying attribute performance scores into performance scores for the 'Biology' class in tier 2

Input performance scores		Output performance scores
Soil biota	**Biology management**	**Biology**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Table A4 - 61: Look-up table for reclassifying attribute performance scores into performance scores for the 'Hydrology' class in tier 2

Input performance scores			Output performance scores
Environmental attributes	**Soil related**	**Hydrology management**	**Hydrology**
1	1	1	1
1	1	2	1
1	1	3	1
1	2	1	1
1	2	2	1
1	2	3	2
1	3	1	1
1	3	2	2
1	3	3	3
2	1	1	1
2	1	2	1
2	1	3	2
2	2	1	1
2	2	2	2
2	2	3	3
2	3	1	2
2	3	2	3
2	3	3	3
3	1	1	1
3	1	2	2
3	1	3	3
3	2	1	2
3	2	2	3
3	2	3	3
3	3	1	3
3	3	2	3
3	3	3	3

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Table A4 - 62: Look-up table for reclassifying attribute performance scores into performance scores for the 'Nutrients' class in tier 2

Input performance scores			Output performance scores
Nutrient content	**Nutrient inputs**	**pH condition**	**Nutrients**
1	1	1	1
1	1	2	1
1	1	3	1
1	2	1	2
1	2	2	2
1	2	3	2
1	3	1	2
1	3	2	2
1	3	3	2
2	1	1	1
2	1	2	2
2	1	3	2
2	2	1	1
2	2	2	2
2	2	3	2
2	3	1	2
2	3	2	2
2	3	3	3
3	1	1	1
3	1	2	2
3	1	3	3
3	2	1	2
3	2	2	2
3	2	3	3
3	3	1	2
3	3	2	3
3	3	3	3

Table A4 - 63: Look-up table for reclassifying attribute performance scores into performance scores for the 'Structure' class in tier 2

Input performance scores		Output performance scores
Soil attributes	**Structure management**	**Structure**
1	1	1
1	2	1
1	3	2
2	1	1
2	2	2
2	3	3
3	1	2
3	2	3
3	3	3

Tier 1

Hereunder, look-up tables that were implemented to group attributes and classes into overarching one overarching class in tier 1.

Table A4 - 64: Look-up table for reclassifying attribute performance scores into performance scores for the 'Soil biodiversity' class in tier 1

Input performance scores				Output performance scores
Nutrients	Biology	Structure	Hydrology	Soil biodiversity
1	1	1	1	1
1	1	1	2	1
1	1	1	3	1
1	1	2	1	1
1	1	2	2	1
1	1	2	3	1
1	1	3	1	1
1	1	3	2	1
1	1	3	3	1
1	2	1	1	1
1	2	1	2	1
1	2	1	3	1
1	2	2	1	1
1	2	2	2	1
1	2	2	3	1
1	2	3	1	1
1	2	3	2	1
1	2	3	3	2
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1	3	1	3	2
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1	3	3	2	2
1	3	3	3	3
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2	1	1	2	1
2	1	1	3	1
2	1	2	1	1
2	1	2	2	1
2	1	2	3	1
2	1	3	1	1
2	1	3	2	1
2	1	3	3	2
2	2	1	1	1
2	2	1	2	1
2	2	1	3	2
2	2	2	1	1
2	2	2	2	2
2	2	2	3	2

A

2	2	3	1	2
2	2	3	2	2
2	2	3	3	3
2	3	1	1	2
2	3	1	2	2
2	3	1	3	3
2	3	2	1	2
2	3	2	2	3
2	3	2	3	3
2	3	3	1	3
2	3	3	2	3
2	3	3	3	3
3	1	1	1	1
3	1	1	2	2
3	1	1	3	3
3	1	2	1	2
3	1	2	2	2
3	1	2	3	2
3	1	3	1	2
3	1	3	2	2
3	1	3	3	3
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3	2	1	2	3
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3	2	2	1	3
3	2	2	2	3
3	2	2	3	3
3	2	3	1	3
3	2	3	2	3
3	2	3	3	3
3	3	1	1	3
3	3	1	2	3
3	3	1	3	3
3	3	2	1	3
3	3	2	2	3
3	3	2	3	3
3	3	3	1	3
3	3	3	2	3
3	3	3	3	3

A4 – 4.5.4 Model procedure

A4 – 4.5.4.1 Run cropland model

1. All input spatial data must be converted to raster format and to a 10 m resolution. In order to perform calculations in PCRaster, it is also necessary that all spatial data has the same extent.
2. Create a Soil C:N ratio, CN_{crop}, layer by dividing Total C by Total N maps from the Soil (biological, physical, and chemical) characteristics, BoBi, dataset.
3. Create a *Soil texture, ST_{crop},* layer by reclassifying *Soil biophysical units*, *BOFEK2012,* conform Table A4 – 65.

Table A4 - 65: Look-up table for reclassifying units from the soil biophysical units map into soil textures, based on classes from the Soil Navigator

BOFEK code	Texture code	Model code	Soil texture
401	2	1	clay
402	2	1	clay
403	2	1	clay
407	2	1	clay
410	2	1	clay
412	2	1	clay
413	2	1	clay
418	2	1	clay
419	2	1	clay
420	2	1	clay
421	2	1	clay
201	6	1	clay
405	6	1	clay
415	6	1	clay
408	3	2	loam
416	3	2	loam
325	5	2	loam
323	8	3	sand

4. Spatial data used as input to represent attribute values is reclassified into *Cropland attribute performance scores*, based on the threshold values assigned for tier 6 (Table A4 - 13). For attributes that do not make use of spatial data as input, reference values are assigned. Where data is missing, a score of 1 is assigned. The lowest value is assigned to avoid possible overestimation of final values in areas where data is missing.
5. Run model by integration of remaining look up tables in Section A4 – 4.5.3.1.
6. Save Croplands soil biodiversity, BD_{crop}, map.

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A4 – 4.5.4.2 Run grasslands model

7. All input spatial data must be in raster format, at a 10 m resolution, and have the same spatial extent
8. Spatial data used as input to represent attribute values is reclassified into *Grassland attribute performance scores*, based on the threshold values assigned for tier 6 (Table A4 - 39). For attributes that do not make use of spatial data as input, reference values are assigned. Where data is missing, a score of 1 is assigned. The lowest value is assigned to avoid possible overestimation of final values in areas where data is missing.

9. Run model by integration of look up tables in Section A4 – 4.5.3.2
10. Save Grasslands soil biodiversity, BD_{grass}, map.

A4 – 4.5.4.3 Develop a land cover map

11. Create a *Reclassified agricultural crop parcels, BRP_{rec}*, layer by reclassifying the *Agricultural crop parcels, BRP,* map into land cover classes conform Table A4 - 66, where 1 = croplands, 2 = grasslands, 3 = semi-natural areas
12. Create a *Reclassified land use, EUM_{rec},* layer by reclassifying the *Land use map of the Netherlands, EUM,* map into land cover classes conform Table A4 - 67, where 1 = croplands, 2 = grasslands, 3 = semi-natural areas, 4 = built-up and paved areas, 5 = water
13. Create a *Reclassified field margins, FM_{rec}*, layer by assigning a value of 3 (= semi-natural areas) to cells where field margins are present within the *Akkerranden Hoeksche Waard 2017* map
14. Create a *Combined land cover, LC,* map by performing an overlay containing
    - all positive values from the FM_{rec} layer
    - where FM_{rec} is 0, all positive values from the BRP_{rec} layer
    - where FM_{rec} and BRP_{rec} are 0, all positive values from the EUM_{rec}

Table A4 - 66: Look-up table for reclassification of land cover classes from the agricultural crop parcels map (BRP) into three land cover classes (1 = croplands, 2 = grasslands, 3 = semi-natural areas)

BRP code	Model land cover type code
2025	1
2014	1
2015	1
2016	1
2706	1
1949	1
1926	2
1095	1
1096	1
2325	1
256	1
257	1
3504	1
247	1
2795	1
2797	1
174	1
242	1
853	1
854	1
311	1
863	1
1936	3
2719	1
511	1
2723	1
3506	1
244	1
3510	1

428	1
235	1
236	1
265	2
336	2
331	2
332	2
266	2
383	2
238	1
3512	1
670	1
241	1
2328	1
799	1
3524	1
2725	1
1923	1
2741	1
2743	1
258	1
316	1
259	1
335	3
2645	1
1927	1
1006	1
992	1
991	1
427	1
2791	1
2792	1
2793	1
2794	1
1044	1
1023	1
1097	1
1098	1
2747	1
1025	1
2735	1
2799	1
1870	1
1022	1
2755	1
2756	1
344	1
370	2
334	2
3803	1
3804	1
372	2

2759	1
237	1
2761	1
1074	1
1075	1
2767	1
2771	1
1876	1
665	1
382	1
2773	1
2774	1
2775	1
2777	1
2779	1
2781	1
233	1
234	1
3802	3
3801	3
1931	1
262	1
1094	1
666	1
2785	1
2789	1

Table A4 - 67: Look-up table for reclassification of land cover classes from the Land use map of the Netherlands (EUM) into five land cover classes (1 = croplands, 2 = grasslands, 3 = semi-natural areas, 4 = built-up and paved areas, 5 = water)

EUM code	Model land cover type code
1	1
2	1
3	1
4	2
5	2
6	1
21	3
22	3
23	3
24	3
25	3
26	3
27	3
28	3
29	3
41	4
42	4
43	4
44	4
45	4

46	4
47	4
48	4
51	5
52	5
53	5

A4 – 4.5.4.4 Assign soil biodiversity performance values to different land cover classes

15. Create a *Soil Biodiversity, BD*, layer where
    - Values from BD_{crop} are assigned to *croplands*, namely land cover type 1 from the *LC* layer
    - Values from BD_{grass} are assigned to *grasslands* and semi-natural areas, namely land cover types 2 and 3 from the *LC* layer
    - Assign a value of 0 to *built-up and paved areas* and *water*, namely land cover types 4 and 5 from the *LC* layer

A4 – 4.6: ‘Sociocultural values’ model specifications

This Section describes the process required for modelling ecosystem service supply and use indicators presented in this study by use of the ‘Sociocultural Values’ model. Below, a stepwise procedure is provided guiding implementation of the model.

A4 – 4.6.1 Spatial data requirements

- Agricultural crop parcels (BRP)
- Basic registry of addresses and buildings (BAG)
- Elevation map of the Netherlands (AHN2)
- High resolution aerial photograph of the Netherlands (Luchtfoto)
- Land use map of the Netherlands (EUM)

A4 – 4.6.2 Non-spatial data requirements

- Values assigned by stakeholders to cultural service indicators
- Data on linkages between cultural service indicators and semi-natural elements

A4 – 4.6.3 Model mechanism

To assess sociocultural values, a survey was distributed by interviewed local representatives (no. of respondents = 87, locals = 83, visitors = 4, men = 55%, women = 45%). Respondents must then assign a score to sociocultural indicators based on the individual’s perceived importance of the value (0 = not important, 5 = very important; Table A4 - 68). Based on the scores assigned to a sociocultural value indicator, a low standard deviation was considered a robust choice for consideration of a proxy indicator. Respondents were then requested to link proxy indicators to natural landscape elements that were deemed important for the delivery of each sociocultural value (maximum choice of three elements; Table A4 - 69). Based on results obtained, scores were assigned to different landscape elements, for each cultural service. First, weights (0-1) were assigned to each element, based on the number of times an element was linked to a particular cultural service (Table A4 - 70). Subsequently, weights were multiplied by the score assigned to each particular cultural service to obtain weighted scores for each element, for each cultural service (Table A4 - 71). Within a final stage, different land cover maps were generated for each semi-natural landscape element. Finally, scores were assigned to semi-natural landscape element typologies. Since some typologies overlap, the highest score assigned to an element is always prioritised.

Table A4 - 68: Score (0 = not important, 5 = important) assigned to each cultural service, based on average scores assigned to sub-indicators for each value (respondents = 87).

Cultural service	Sub-indicator	Average score (respondents)	Score (average sub-indicator scores)
Cultural history and identity	History of area development	4.3	4.3
	Local products	4.2	
	The farmer life	4.3	
Habitability	Accessibility	3.6	3.6
	Low population density	4.3	
	Public festivities	3.0	
Intrinsic values	Rich nature	4.8	4.7
	Protected nature areas	4.7	
	Bird species diversity	4.6	
Knowledge	Sustainable agriculture	4.6	4.4
	Agricultural knowledge	4.2	
Landscape aesthetics	The open landscape	4.7	4.7
	Calmness	4.7	
Recreational potential	Biking and hiking	4.4	4.6
	Greenness of the area	4.7	

Table A4 - 69: Score assigned to each cultural value, based on average scores assigned to sub-indicators for each value

Cultural service indicator	Trees, hedges, wood- walls	Grasslands and shrubs	Large water bodies	Ditches and creeks	Agricultural fields	Polder structures
Cultural history and identity	10	7	24	25	59	42
Habitability	3	4	13	4	5	12
Intrinsic values	72	75	63	86	10	39
Educational and scientific knowledge	13	2	1	9	64	7
Landscape aesthetics	35	49	44	37	33	37
Recreational potential	41	29	13	15	6	27

Table A4 - 70: Weight assigned to each semi-natural element typology for each ecosystem service

Cultural service indicator	Trees, hedges, wood walls	Grasslands and shrubs	Large water bodies	Ditches and creeks	Agricultural fields	Polder structures
Cultural history and identity	0.2	0.1	0.4	0.4	1.0	0.7
Habitability	0.2	0.3	1.0	0.3	0.4	0.9
Intrinsic values	0.8	0.9	0.7	1.0	0.1	0.5
Educational and scientific knowledge	0.2	0.0	0.0	0.1	1.0	0.1
Landscape aesthetics	0.7	1.0	0.9	0.8	0.7	0.8
Recreational potential	1.0	0.7	0.3	0.4	0.1	0.7

Table A4 - 71: Weighted scores of semi-natural element typologies for each ecosystem service

Cultural service indicator	Trees, hedges, wood walls	Grasslands and shrubs	Large water bodies	Ditches and creeks	Agricultural fields	Polder structures
Cultural history and identity	0.7	0.5	1.7	1.8	4.3	3.1
Educational and scientific knowledge	0.9	0.1	0.1	0.6	4.4	0.5
Habitability	0.8	1.1	3.6	1.1	1.4	3.3
Intrinsic values	3.9	4.1	3.4	4.7	0.5	2.1
Landscape aesthetics	3.4	4.7	4.2	3.5	3.2	3.5
Recreational potential	4.6	3.3	1.5	1.7	0.7	3.0

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A4 – 4.6.4 Model procedure

A4 – 4.6.4.1 Create land cover typology maps based on the EUM map

1. Convert the *Land use map of the Netherlands, EUM,* feature class to raster format and to a 10 m resolution.
2. Create a *Trees, hedges, woodwalls, THW_{EUM}*, layer by reclassifying land cover typologies from *EUM* following Table A4 - 72 (column 2).
3. Create a *Grasslands and shrubs, GS_{EUM}*, layer by reclassifying land cover typologies from *EUM* following Table A4 - 72 (column 3).
4. Create a *Large water bodies, LWB*, layer by reclassifying land cover typologies from *EUM* following Table A4 - 72 (column 4).
5. Create a *Ditches and creeks, DC*, layer by reclassifying land cover typologies from *EUM* following Table A4 - 72 (column 5).
6. Create an *Agricultural fields, AF_{EUM},* layer by reclassifying land cover typologies from *EUM* following Table A4 - 72 (column 6).

A4 – 4.6.4.2 Create land cover typology maps based on the BRP map

7. Convert the *Agricultural crop parcels, BRP,* map to raster format and to a 10 m resolution.
8. Create a *Trees, hedges, woodwalls, THW_{BRP}*, layer by reclassifying land cover typologies from *BRP* following Table A4 - 73 (column 2).
9. Create a *Grasslands and shrubs, GS_{BRP}*, layer by reclassifying land cover typologies from *BRP* following Table A4 - 73 (column 3).
10. Create an *Agricultural fields, AF_{BRP}*, layer by reclassifying land cover typologies from *BRP* following Table A4 - 73 (column 4).

Table A4 - 72: Reclassification of land cover typologies from the Land use map of the Netherlands (EUM) layer into six semi-natural element typologies (0 = not present, 1 = present)

EUM code	Trees, hedges, woodwalls	Grasslands and shrubs	Large water bodies	Ditches and creeks	Agricultural fields
1	0	0	0	0	1
2	0	0	0	0	1
3	0	0	0	0	1
4	0	1	0	0	1
5	1	1	0	0	1
6	0	0	0	0	1
11	0	0	0	0	0
12	0	0	0	0	0
13	0	0	0	0	0
21	0	0	0	0	0
22	0	0	0	0	0
23	0	0	0	0	0
24	0	0	0	0	0
25	0	0	0	0	0
26	0	0	1	0	0
27	0	1	0	0	0
28	0	0	0	0	0
29	0	0	0	0	0
41	0	0	0	0	0
42	0	0	0	0	0
43	0	0	0	0	0
44	0	0	0	0	0
45	0	0	0	0	0
46	0	0	0	0	0
47	0	0	0	0	0
48	0	0	0	0	0
51	0	0	1	0	0
52	0	0	1	0	0
53	0	0	0	1	0

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Table A4 - 73: Reclassification of land cover typologies from the Agricultural crop parcels (BRP) map into three semi-natural element typologies (0 = not present, 1 = present)

BRP code	Trees, hedges, woodwalls (Trees_BRP)	Grasslands and shrubs (Grass_BRP)	Agricultural fields (Agri_BRP)
174	0	0	1
233	0	0	1
234	0	0	1
235	0	0	1
236	0	0	1
237	0	0	1
238	0	0	1
241	0	0	1
242	0	0	1
244	0	0	1
247	0	0	1
256	0	0	1
257	0	0	1
258	0	0	1
259	0	0	1
262	0	0	1
265	0	1	0
266	0	1	0
311	0	0	1
316	0	0	1
331	0	1	0
332	0	1	0
334	0	1	0
335	0	0	0
336	0	1	0
344	0	0	1
370	0	1	0
372	0	1	0
382	0	0	1
383	0	1	0
427	0	0	1
428	0	0	1
511	0	0	1
665	0	0	1
666	0	0	1
670	0	0	1
799	0	0	1
853	0	0	1
854	0	0	1
863	1	0	1
991	0	0	1
992	0	0	1
1006	0	0	1
1022	0	0	1
1023	0	0	1
1025	0	0	1

1044	0	0	1
1074	0	0	1
1075	0	0	1
1094	0	0	1
1095	0	0	1
1096	0	0	1
1097	0	0	1
1098	0	0	1
1870	0	0	1
1876	0	0	1
1923	0	0	1
1926	0	1	0
1927	0	0	1
1931	0	0	1
1936	1	0	0
1949	0	0	1
2014	0	0	1
2015	0	0	1
2016	0	0	1
2025	0	0	1
2325	0	0	1
2328	0	0	1
2645	0	0	1
2706	0	0	1
2719	0	0	1
2723	0	0	1
2725	0	0	1
2735	0	0	1
2741	0	0	1
2743	0	0	1
2747	0	0	1
2755	0	0	1
2756	0	0	1
2759	0	0	1
2761	0	0	1
2767	0	0	1
2771	0	0	1
2773	0	0	1
2774	0	0	1
2775	0	0	1
2777	0	0	1
2779	0	0	1
2781	0	0	1
2785	0	0	1
2789	0	0	1
2791	0	0	1
2792	0	0	1
2793	0	0	1
2794	0	0	1
2795	0	0	1
2797	0	0	1
2799	0	0	1

3504	0	0	1
3506	0	0	1
3510	0	0	1
3512	0	0	1
3524	0	0	1
3801	0	0	0
3802	0	0	0
3803	0	0	1
3804	0	0	1

A4 – 4.6.4.3 Create remaining land cover typology maps

11. Vegetation layers at a 10 m resolution are created, conform Appendix 3 – 2 (Section A2 – 2.1.2.1), using the *AHN2, BAG*, *Luchtfoto,* and *BRP* layers as input. Vegetation layers include trees *(fr_{tree})*, *bushes/shrubs (fr_{shrub})*, and *low vegetation ($fr_{loweveg}$)*. Each layer shows the *fraction per grid cell (0-1) covered by each vegetation type.*
12. Create a *Trees* layer by assigning a value of 1 to all cells from the fr_{tree} layer containing values higher than 0. All other cells are assigned a value of 0.
13. Create a *Shrub* layer by assigning a value of 1 to all cells from the fr_{shrub} layer containing values higher than 0. All other cells are assigned a value of 0.
14. Create a *Polder, POL,* layer by assigning a value of 1 to all cells in *AHN2* containing a value higher than 0.5 m.
15. Create a *Trees, hedges, and woodwalls, THW,* layer by adding layers created in Steps 2, 8, and 12. All cells with a positive value are assigned a value of 1. All other cells are assigned a value of 0.
16. Create a *Grassland and shrubs, GS,* layer by adding layers created in Steps 3, 9, and 13. All cells with a positive value are assigned a value of 1. All other cells are assigned a value of 0.
17. Create an *Agricultural fields, AF,* layer by adding layers created in Steps 6 and 10. All cells with a positive value are assigned a value of 1. All other cells are assigned a value of 0.
18. This should result in the layers *LWB* (Step 4), *DC* (Step 5), *POL* (Step 14), *THW* (Step 15), *GS* (Step 16), and *AF* (Step 17).

A4 – 4.6.4.4 Create a 'Cultural history and identity' map

19. Create six layers displaying the *Cultural history and identity score* for six landscape typologies, by reclassifying the layers LWB, DC, POL, THW, GS, and AF, following Table A4 - 71 (row 2). This should result in six new layers (LWB_{CHI}, DC_{CHI}, POL_{CHI}, THW_{CHI}, GS_{CHI}, and AF_{CHI}) displaying the distribution of the cultural history and identity score for each semi-natural element typology across the landscape.
20. Create a *Cultural history and identity, CHI,* map by performing an overlay where the highest score per cell for the layers LWB_{CHI}, DC_{CHI}, POL_{CHI}, THW_{CHI}, GS_{CHI}, and AF_{CHI}, is prioritised, then the second highest score, and so on (Agricultural fields, followed by Polder structures, and so on).

A4 – 4.6.4.5 Create an 'Educational and scientific knowledge' map

21. Create six layers displaying the *Educational and scientific Knowledge score* for six landscape typologies, by reclassifying the layers LWB, DC, POL, THW, GS, and AF, following Table A4 - 71 (row 3). This should result in six new layers (LWB_{ESK}, DC_{ESK}, POL_{ESK}, THW_{ESK}, GS_{ESK}, and AF_{ESK}) displaying the distribution of the educational and scientific knowledge score for each semi-natural element typology across the landscape.

22. Create an *Educational and scientific Knowledge, ESK,* map by performing an overlay where the highest score per cell for the layers LWB_{ESK}, DC_{ESK}, POL_{ESK}, THW_{ESK}, GS_{ESK}, and AF_{ESK}, is prioritised, then the second highest score, and so on.

A4 – 4.6.4.6 Create a 'Habitability' map

23. Create six layers displaying the *Habitability score* for six landscape typologies, by reclassifying the layers LWB, DC, POL, THW, GS, and AF, following Table A4 - 71 (row 4). This should result in six new layers (LWB_{HAB}, DC_{HAB}, POL_{HAB}, THW_{HAB}, GS_{HAB}, and AF_{HAB}) displaying the distribution of the habitability score for each semi-natural element typology across the landscape.
24. Create a *Habitability, HAB,* map by performing an overlay where the highest score per cell for the layers LWB_{HAB}, DC_{HAB}, POL_{HAB}, THW_{HAB}, GS_{HAB}, and AF_{HAB}, is prioritised, then the second highest score, and so on.

A4 – 4.6.4.7 Create an 'Intrinsic values' map

25. Create six layers displaying the *Intrinsic values score* for six landscape typologies, by reclassifying the layers LWB, DC, POL, THW, GS, and AF, following Table A4 - 71 (row 5). This should result in six new layers (LWB_{IV}, DC_{IV}, POL_{IV}, THW_{IV}, GS_{IV}, and AF_{IV}) displaying the distribution of the intrinsic values score for each semi-natural element typology across the landscape.
26. Create an *Intrinsic values, HAB,* map by performing an overlay where the highest score per cell for the layers LWB_{IV}, DC_{IV}, POL_{IV}, THW_{IV}, GS_{IV}, and AF_{IV} is prioritised, then the second highest score, and so on.

A4 – 4.6.4.8 Create a 'Landscape aesthetics' map

27. Create six layers displaying the *Landscape aesthetics score* for six landscape typologies, by reclassifying the layers LWB, DC, POL, THW, GS, and AF, following Table A4 - 71 (row 6). This should result in six new layers (LWB_{LA}, DC_{LA}, POL_{LA}, THW_{LA}, GS_{LA}, and AF_{LA}) displaying the distribution of the landscape aesthetics score for each semi-natural element typology across the landscape.
28. Create a *Landscape aesthetics, LA,* map by performing an overlay where the highest score per cell for the layers LWB_{LA}, DC_{LA}, POL_{LA}, THW_{LA}, GS_{LA}, and AF_{LA} is prioritised, then the second highest score, and so on.

A4 – 4.6.4.9 Create a 'Recreational potential' map

29. Create six layers displaying the *Recreational potential score* for six landscape typologies, by reclassifying the layers LWB, DC, POL, THW, GS, and AF, following Table A4 - 71 (row 7). This should result in six new layers (LWB_{RP}, DC_{RP}, POL_{RP}, THW_{RP}, GS_{RP}, and AF_{RP}) displaying the distribution of the recreational potential score for each semi-natural element typology across the landscape.
30. Create a *Recreational potential, RP,* map by performing an overlay where the highest score per cell for the layers LWB_{RP}, DC_{RP}, POL_{RP}, THW_{RP}, GS_{RP}, and AF_{RP} is prioritised, then the second highest score, and so on.

A4 – 4.7: 'Property value' model specifications

This Section describes the process required for modelling ecosystem service supply and use indicators presented in this study by use of the 'Property Value' model, conform the NC-Model (Paulin et al., 2020b; Remme et al., 2018). A stepwise procedure guiding the model's application is available in Paulin et al. (2020b).

A4 – 4.7.1 Spatial data requirements

- Agricultural crop parcels (BRP)
- Basic registry of addresses and buildings (BAG)
- Elevation map of the Netherlands (AHN2)
- High resolution aerial photograph of the Netherlands (Luchtfoto)
- Key neighbourhood statistics (Wijk- en buurtkaart)
- Land use map of the Netherlands (EUM)
- Topographic land use map of the Netherlands (Top10NL)
- Real estate value (WOZ-Waarde)

A4 – 4.7.2 Non-spatial data requirements

- Various reference values found in Appendix 3 – 2.3.

Appendix 4 – 5

Table A4 - 74: Descriptive statistics for spatially quantified ecosystem service indicator. CP = Crop Production; AQR = Air Quality Regulation; HH = Human Health; PC = Pest Control; SB = Soil Biodiversity; LS = Landscape Services; PV = Property Value.

Ecosystem service indicator	Unit	Minimum	Maximum	Range	Mean	Std. dev.
CP - Volume of harvested crops	million €/yr	18	771	753	322	237
CP - Net output of harvested crops	million kg/yr	9	623	614	82	126
CP - Value added of harvested crops	million €/yr	6	393	387	54	80
AQR - PM_{10} retention	thousand kg/yr	0.00	0.27	0.27	0.05	0.01
AQR - Reduced health costs	million €/yr	0.0	11.7	11.6	2.2	0.5
HH - Reduced health costs	million €/yr	0	297	297	38	24
HH - Reduced labour costs	million €/yr	133	145,465	145,332	18,545	11,598
HH - Reduced visits to general practitioners	thousand visits/yr	0.00	0.34	0.34	0.04	0.03
PC - Yearly effective pest control	average score 0 - 1.5	0.75	1.50	0.75	0.90	0.13
SB - Average performance of soil biodiversity	performance classes	1.00	3.00	2.00	2.59	0.54
LS - Cultural identity and heritage	average score 0 - 5	0.5	4.3	3.8	1.6	0.3
LS - Educational and scientific knowledge	average score 0 - 5	0.1	4.4	4.3	3.1	1.8
LS - Habitability	average score 0 - 5	0.8	3.6	2.8	1.5	0.6
LS - Intrinsic value	average score 0 - 5	0.5	4.7	4.2	2.3	0.4
LS - Landscape aesthetics	average score 0 - 5	3.2	4.7	1.5	3.7	0.6
LS - Recreational potential	average score 0 - 5	0.7	4.6	3.9	1.7	1.3
PV - Contribution to property value	million €	60	50,880	50,820	10,291	6,396

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Table A4 - 75: Pearson correlations among spatially quantified ecosystem service indicators. CP = Crop Production; AQR = Air Quality Regulation; HH = Human Health; PC = Pest Control; SB = Soil Biodiversity; LS = Landscape Services; PV = Property Value.

	CP - Volume of harvested crops	CP - Net output of harvested crops	CP - Value added of harvested crops	AQR - PM_{10} retention	AQR - Reduced health costs	HH - Reduced health costs	HH - Reduced labour costs	HH - Reduced visits to general practitioners	PC - Yearly effective pest control	SB - Average performance of soil biodiversity	LS - Cultural identity and heritage	LS - Educational and scientific knowledge	LS - Habitability	LS - Intrinsic value	LS - Landscape aesthetics	LS - Recreational potential	PV - Contribution to property value
CP - Volume of harvested crops	1.00	0.39	0.41	0.50	0.50	-0.03	-0.03	-0.03	0.29	0.51	0.59	0.61	0.36	-0.05	0.37	0.06	-0.03
CP - Net output of harvested crops		1.00	1.00	0.31	0.31	-0.02	-0.02	-0.02	0.17	0.32	0.37	0.38	0.23	-0.02	0.24	0.05	-0.02
CP - Value added of harvested crops			1.00	0.32	0.32	-0.02	-0.02	-0.02	0.18	0.33	0.38	0.40	0.24	-0.02	0.24	0.05	-0.02
AQR - PM_{10} retention				1.00	1.00	-0.00	-0.00	-0.00	0.59	0.85	0.81	0.79	0.75	0.39	0.79	0.57	-0.01
AQR - Reduced health costs					1.00	-0.00	-0.00	-0.00	0.59	0.85	0.81	0.79	0.75	0.39	0.79	0.57	-0.01
HH - Reduced health costs						1.00	1.00	1.00	-0.01	-0.04	-0.04	-0.04	0.01	0.05	0.03	0.09	0.47
HH - Reduced labour costs							1.00	1.00	-0.01	-0.04	-0.04	-0.04	0.01	0.05	0.03	0.09	0.47

HH - Reduced visits to general practitioners	1.00	-0.01	-0.04	-0.04	-0.04	0.01	0.05	0.03	0.09	0.47
PC - Yearly effective pest control		1.00	0.57	0.54	0.54	0.50	0.25	0.53	0.40	0.00
SB - Average performance of soil biodiversity			1.00	0.82	0.83	0.68	0.29	0.72	0.48	-0.04
LS - Cultural identity and heritage				1.00	0.97	0.77	0.39	0.84	0.50	-0.03
LS - Educational and scientific knowledge					1.00	0.64	0.25	0.74	0.42	-0.03
LS - Habitability						1.00	0.63	0.89	0.66	0.02
LS - Intrinsic value							1.00	0.79	0.88	0.06
LS - Landscape aesthetics								1.00	0.84	0.05
LS - Recreational potential									1.00	0.11
PV - Contribution to property value										1.00

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Appendix 4 – 6

This Appendix displays spatial data used as input for Soil Biodiversity model (i.e., soil characteristics maps), developed by the Dutch National Institute for Public Health and the Environment (RIVM; Rutgers et al., 2009; Van Wijnen et al., 2012). Unshaded areas comprise areas where no value has been assigned.

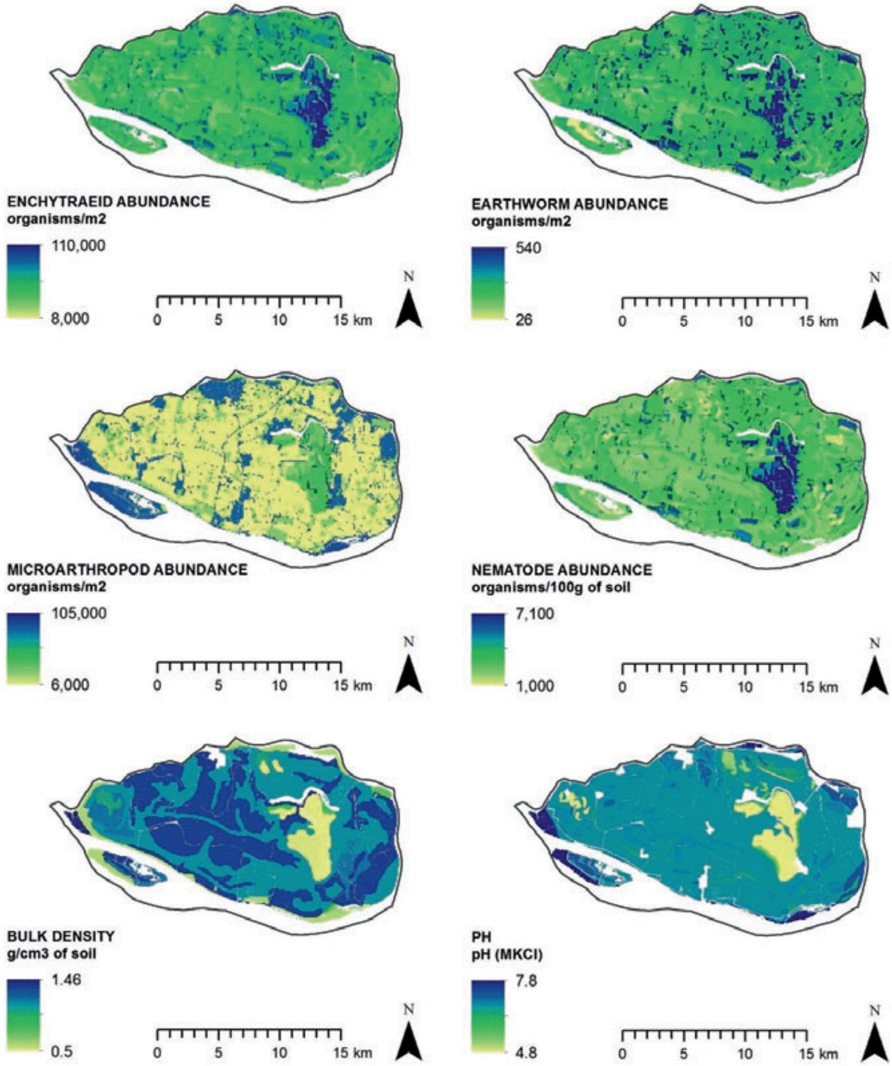

Figure A4 - 3: Biotic and abiotic soil characteristics in the Hoeksche Waard (cell size = 10 x 10 m). Data is classified by use of the standard deviation stretch type, which applies a linear stretch between the values defined by the standard deviation (n) value (https://desktop.arcgis.com/). Source:

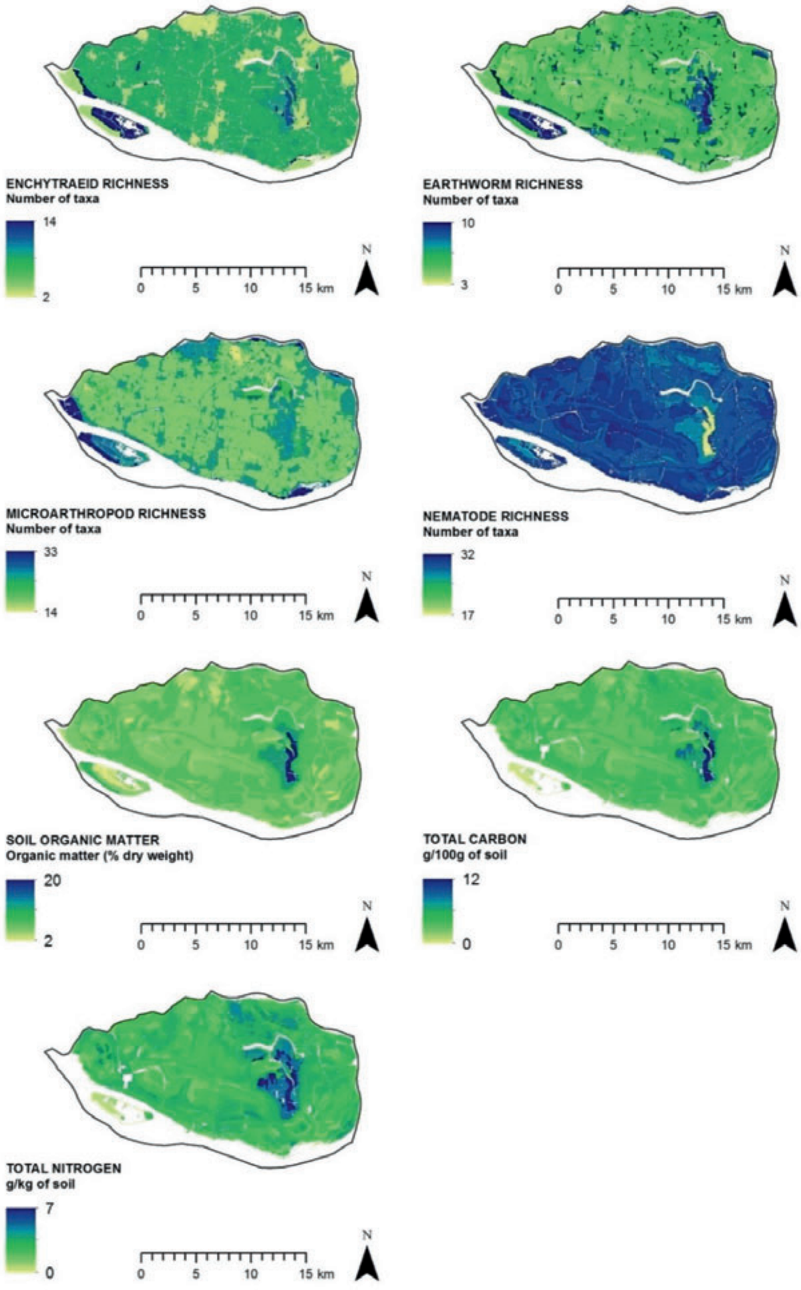

Figure A4 - 4: Continued - Biotic and abiotic soil characteristics in the Hoeksche Waard (cell size = 10 x 10 m). Data is classified by use of the standard deviation stretch type, which applies a linear stretch between the values defined by the standard deviation (n) value (https://desktop.arcgis.com/).

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Appendix 4 – 7

This Appendix displays all maps displaying ecosystem service use indicators modelled by use of the Landscape Services model. Unshaded areas comprise areas where no value has been assigned.

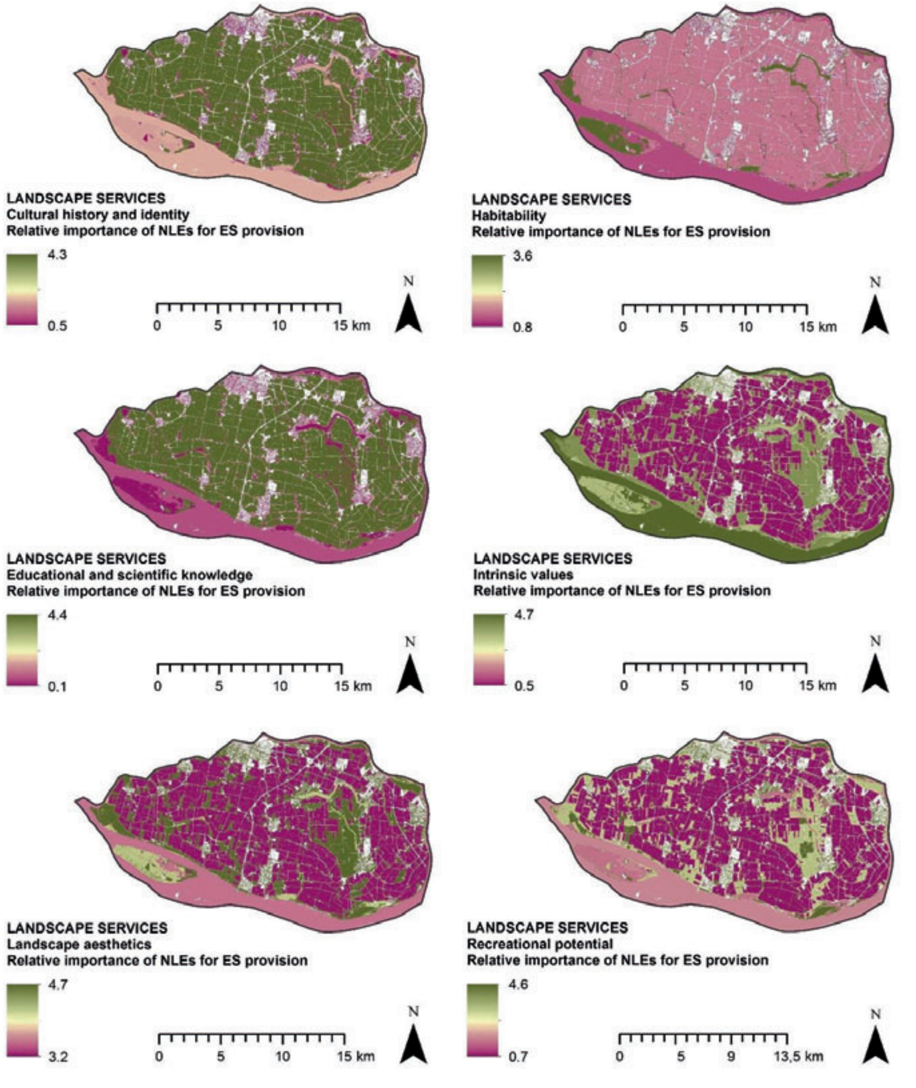

Figure A4 - 5: Cultural ecosystem services in the Hoeksche Waard, modelled by use of the Landscape Services model (cell size = 10 x 10 m).

Appendix 4 – 8

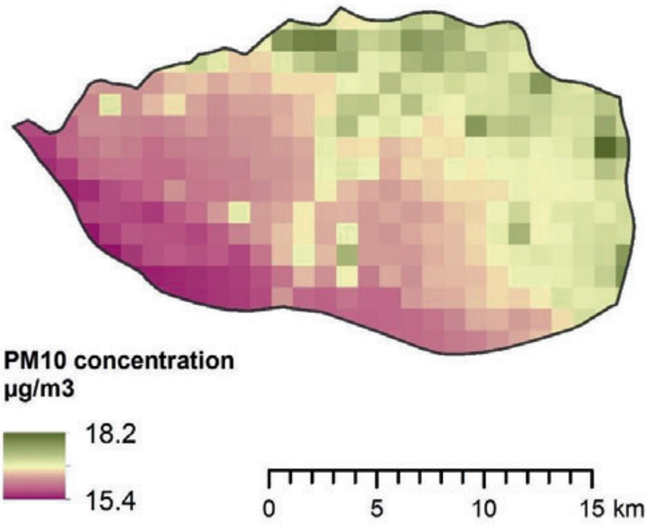

Figure A4 - 6: PM_{10} concentration levels in the Hoeksche Waard (source: https://www.rivm.nl)

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