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Ecosystem-functioning approaches for assessing and managing ecosystem services

Methodes voor evaluatie en beheer van ecosysteemdiensten op basis van ecosysteemfunctioneren

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“ ‘Now I have to think very carefully,’ Pluk said to himself. ‘If I whistle three times, a ferry comes to cary me cross the river. But the boatman is a wolf… a *tell-me-where-wolf*. So I don’t do it. It would be incredibly stupid to whistle. I’d rather go back. I drive back along the river to the big ferry and there I wait for the fog to disappear. Which can ofcourse take a long time… it could take days. And then it’s too late, then it’s no longer needed… I have to cross the river now! Today! But I don’t dare to whistle. How I wish somebody would be here to give me a little courage. I lost heart… maybe my heart sank…’

Pluk looked into the water, but there was nothing special to see. ‘Come on,’ he said to himself. ‘Now you must choose. Or leave, or whistle, and call the wolf. Is a wolf actually something to be afraid of? All animals have always been nice to me. And if it gets scary… then I run to my car and I drive off.’

Pluk put his fingers to his mouth and whistled. It cut through the silence. Again and again… three times he whistled. ‘All right. Now we have to wait and not be afraid.’ Nothing happened for a long time. He stood on the deck with shivering knees and a pounding heart in the throat. The fog was so dense that he could only see a small part of the grayish water, near the river bank. Then he heard the splashing sound of rowing oars… ”

Atmospheric image from “Pluk van de Petteflet”, Annie M.G. Schmidt, free translation

Dank

2010. Pas getrouwd en op huwelijksreis voor zes maanden in Noord- en Zuid-Amerika. We hadden een rustperiode in het rondtjokken ingelast, een momentje om even na te denken over wat we gaan doen als we terugkeren naar België. 's Morgens in het internetcafé zag ik de aankondiging dat er een doctoraatspositie open stond bij ECOBE, 's avonds was de sollicitatiebrief verstuurd. Dat was het gewoon, ecosysteemdiensten! Ik was meteen fan.

En dat was dan het begin van mijn doctoraat waar ik acht jaar zoet mee ben geweest. Ja, het klopt dat een goede dosis zelfstandigheid nodig is om een doctoraat afgerond te krijgen. Maar zonder de hulp en ondersteuning van een aantal personen was het me hoe dan ook nooit gelukt.

Patrick, het spreekt voor zich dat ik dankbaar ben dat ik de kans heb gekregen hieraan te beginnen. Maar ook om uw kritische oog dat ervoor zorgde dat het werk de goede richting uitging. Geen 'handje vasthouden', maar er wel zijn wanneer het nodig was, en steeds met het juiste advies dat me hielp tot nieuwe inzichten te komen. Jan, de draaiende motor van ons ecosysteemdienstengroepje, die zorgt voor continuïteit, die het voor mij mogelijk heeft gemaakt om het acht jaar uit te zingen. Dank om me uit de soms woelige branding van de beginjaren te sleuren. En gewoon om een fijne collega te zijn.

Mijn bureaugenoten en andere ecosysteemdiensten-collega's. Annelies, fijn dat wanneer ik het ook maar vraag - en ik heb altijd veel vragen - jij er bent om tips te geven en presentaties en papers te overlopen. Dirk, master of python, bedankt voor het delen van je programmeerkennis en je immer nuchtere blik om zaken op tijd en stond te relativeren. Chumping, bedankt voor de hulp die je altijd spontaan aanbiedt. Lotte, bedankt voor het opkrikken van de sfeer met feestjes en barbecues, van ons bureau tot op departementsniveau. Steven, Martijn en Apostolos, jullie maken het groepje compleet met raad en daad vanuit andere ooghoeken dan deze die ik als geograaf aangeleerd heb kregen. Aan mijn talrijke andere collega's: jullie maken de koffiepauzes hoe een koffiepauze moet zijn – ontspannend. Een aantal collega's van het eerste uur - Floor, Alanna, Sander, ... - hebben ondertussen hun vleugels uitgeslagen en zullen dit dankwoord misschien nooit lezen, maar toch: bedankt voor de leuke sfeer op het bureau, in het koffiekot, op feestjes, op congressen.

Mijn lieve familie en vrienden, jullie waren steeds oprecht geïnteresseerd in wat ik deed en wat die ecosysteemdiensten nu precies zijn. Dit was voor mij een leerrijke oefening om de o zo belangrijke vertaalslag te maken van droge wetenschap naar een interessant verhaal voor breed publiek.

Mama en papa, zonder jullie hulp had ik er allicht nog vier jaar langer over gedaan. Woorden doen tekort aan mijn dankbaarheid. Jullie staan al-tijd klaar, voor mij, voor Siebe en Nellis, en voor Serdar. Jonge kinderen combineren met een doctoraat of met een job is niet altijd simpel, maar een deel van de uitdaging is mee door jullie gedragen. Ook babacu is een hele grote hulp geweest. Serdar, bedankt om me de vrijheid te geven om mijn ding te doen. Het vergt ook van jou een hele inspanning, vooral wanneer ik voor meerdere dagen of weken in het buitenland ben, en jij er alleen voor staat om voor Siebe en Nellis te zorgen. Mijn lieve, zachte man van mijn leven. Al moet je voor de letterlijke betekenis die plaats nu wel delen met twee andere kereltjes. Siebe en Nellis, twee stukken van mijn hart die armpjes en beentjes hebben gekregen.

2018. Het is me dan toch gelukt. Voor mij gaat het boek nu toe, voor jullie gaat het open.

Summary

Ecosystem services are a general term for all the benefits that nature provides to humans. These include food, clean drinking water, control of greenhouse gasses, protection against floods as well as potential for recreation, a sense of well-being, … . The concept has emerged in response to the unabated loss of biodiversity and as a tool to increase support for nature and biodiversity conservation. Originally it had a main purpose of awareness raising and the most commonly used approach to assess them was to attach a value – often monetary – to land use types for their capacity to deliver certain ecosystem services. With increasing interest to include ecosystem services in spatial planning and using it to support decision-making, criticism started to grow on the lack of ecological underpinning of land-use based indicators. Because of the wide array of disciplines that the concept of ecosystem services covers, developing scientifically rigorous yet user-friendly and transparent methodologies poses a huge challenge. The main aim of this research is to develop practical methodologies to improve the consideration of ecosystem functioning in ecosystem services assessments and thus advance the implementation of ecosystem services in practice.

As a first step (Chapter 2), it was tested if a more detailed land use classification that includes information on the biophysical environment and on land management allows to more accurately assess ecosystem services. By comparing with estimates of biophysical models, it was demonstrated that land use is insufficiently precise to predict multiple ecosystem services, even when using a highly detailed thematic resolution.

Next, the potential of Bayesian networks to deal with the accuracy – practicability trade-off was explored in a case-study with three conflicting ecosystem services (Chapter 3). The main advantages of Bayesian networks in this respect are their capacity to combine knowledge from different sources, including process-based models as well as expert elicitation, and the possibility to optimize service delivery based on local ecosystem conditions. However, the difficulty to include feedbacks makes them less suitable to model dynamic ecosystem processes and interactions among ecosystem services.

The study on the role of sand transport to support ecosystem services of coastal dunes (Chapter 4) highlights the importance of taking into account dynamic processes in ecosystem services assessments. Five ecosystem services were quantified in different dune habitats that rely on sand transport in varying degrees. Coastal safety maintenance and recreation were found to be economically the most valuable ecosystem services of coastal dunes and were especially high in the dynamic dune habitats. Based on these insights, two novel process-based approaches were developed in which ecosystem processes are the central focus of the assessment. The first approach (Chapter 5) enables to find common ground between biodiversity and ecosystem services based on

an identification of the ecosystem processes that underlie the development of habitats and ecosystem services. The contribution of processes to the development of habitats and the production of ecosystem services was derived using expert elicitation. This allowed to take into account all relevant processes (biophysical, ecological and anthropogenic), including those that are difficult to quantify. Processes are ranked according to the conflicts and synergies they create and taking into account stakeholders' preferences. The results guide in the debate on choosing over inevitable trade-offs and support the creation of a shared vision in spatial planning. In a next chapter (Chapter 6), this methodology is further elaborated to develop a supportive tool to integrate ecosystem services into impact assessments. For each of the processes, the parameters are identified that lead to changes in the process and ultimately affect production of ecosystem services. Evaluating impacts on processes improves the consideration of cross-sectoral effects and of effects taking place beyond the boundaries of the individual project. The method is based on expert elicitation and can be used as a pre-screening tool to identify the ecosystem services and habitats that are potentially affected by a project development.

This work stems from applied research projects and several of the methodologies were used to support decision-making processes in different cases in Flanders. To conclude, this research underpins the need to more prominently include dynamic ecosystem processes in the next-generation ecosystem services tools and aims to provide inspiration for the development of such tools.

Samenvatting

Ecosysteemdiensten zijn een algemene term voor de voordelen die de natuur aan de mens levert. Het kan hierbij gaan om voedsel, proper drinkwater, beperken van broeikasgassen in de atmosfeer, beschermen tegen overstromingen, maar ook recreatie, een gevoel van welzijn, … . Het concept is ontstaan als reactie op het aanhoudende verlies van biodiversiteit en als een middel om meer draagvlak te creëren voor de bescherming van natuur en biodiversiteit. Oorspronkelijk had het concept vooral een focus op bewustmaking. De meest gebruikte methode hiervoor was het toekennen van een waarde – vaak monetair – aan ieder landgebruikstype voor de levering van bepaalde ecosysteemdiensten. Maar met de toenemende interesse om ecosysteemdiensten mee te nemen bij de inrichting van landschappen en als beleidsondersteuning kwam kritiek op het gebrek aan onderbouwing op basis van kennis van ecosysteemfunctioneren. Ecosysteemdiensten zijn echter gelinkt aan zeer uiteenlopende disciplines waardoor de ontwikkeling van wetenschappelijk gefundeerde methodes die tegelijk praktisch zijn in gebruik een belangrijke uitdaging vormt. Dit onderzoek heeft als doel om dergelijke methodes te ontwikkelen en de toepassing van ecosysteemdiensten in ruimtelijke planning en beheer te stimuleren.

In een eerste fase (Hoofdstuk 2) is onderzocht of een meer gedetailleerde landgebruiksclassificatie, die ook informatie bevat over de biofysische omgeving en beheer, toelaat om ecosysteemdiensten beter in te schatten. Op basis van een vergelijking met de resultaten van biofysische modellen werd aangetoond dat landgebruik onvoldoende accuraat is om meerdere ecosysteemdiensten te beoordelen, zelfs wanneer een hoge thematische resolutie wordt gebruikt.

Vervolgens werden de mogelijkheden van Bayesiaanse netwerken voor het ontwikkelen van ecosysteemdienstenmodellen die zowel wetenschappelijk onderbouwd als praktisch zijn nader onderzocht (Hoofdstuk 3). De belangrijkste voordelen van Bayesiaanse netwerken zijn de mogelijkheid om kennis van verschillende bronnen, zowel proces-gebaseerde modellen als expert-gebaseerde kennis, te integreren, en om optimale scenario's te ontwikkelen voor meerdere ecosysteemdiensten op basis van plaatselijke omgevingskenmerken. De moeilijkheid om feedbacks te integreren maakt hen echter minder geschikt om dynamische ecosysteemprocessen en interacties tussen ecosysteemdiensten te modelleren.

Het onderzoek rond de rol van zandtransport in het leveren van diensten in kustduinen (Hoofdstuk 4) onderstreept het belang om dynamische processen mee in rekening te brengen in ecosysteemdiensten modellen. Vijf ecosysteemdiensten werden berekend in duinhabitats die in verschillende mate afhankelijk zijn van zandtransport. Het onderzoek toont aan dat kustbescherming en recreatie economisch gezien de meest waardevolle ecosysteemdiensten van kustduinen zijn en deze zijn vooral belangrijk in de dynamische habitats. Op basis van deze

inzichten werden twee nieuwe proces-gebaseerde methodes ontwikkeld. De eerste methode (Hoofdstuk 5) laat toe om op basis van onderliggende ecosysteemprocessen een gemeenschappelijke basis te identificeren tussen ecosysteemdiensten en biodiversiteit. De manier waarop processen bijdragen tot de ontwikkeling van habitats en het leveren van diensten is gebaseerd op expertbeoordeling. Dit laat toe om alle belangrijke processen in rekening te brengen (biofysische, ecologische en antropogene), inclusief de processen die moeilijk te kwantificeren zijn. Processen worden gerangschikt op basis van de conflicten en de synergiën die ze creëren en rekening houdend met de voorkeuren van lokale stakeholders. De resultaten vormen een hulpmiddel bij de ontwikkeling van een gedragen ruimtelijke visie en het maken van keuzes wanneer trade-offs onvermijdelijk zijn. In een volgende hoofdstuk (Hoofdstuk 6) wordt deze methode verder uitgewerkt tot een instrument om ecosysteemdiensten mee te nemen bij de beoordeling van milieueffecten. Voor ieder proces worden de parameters geïdentificeerd die leiden tot een verandering in het proces en zo de productie van ecosysteemdiensten beïnvloeden. Door veranderingen in processen te beschouwen worden cumulatieve effecten beter in rekening gebracht en ook de effecten die zich afspelen voorbij de grenzen van het individuele project. De methode is gebaseerd op expertbeoordeling en kan gebruikt worden als een quick-scan om de ecosysteemdiensten en habitats te identificeren die door een project beïnvloed kunnen worden.

Dit werk vloeit voort uit toegepaste onderzoeksprojecten in Vlaanderen en meerdere van de voorgestelde methodes werden gebruikt voor beleidsondersteuning. De bevindingen uit dit onderzoek onderstrepen het belang om dynamische ecosysteemprocessen beter te integreren in de volgende generatie ecosysteemdienstenmodellen en bieden inspiratie bij de ontwikkeling van dergelijke instrumenten.

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General introduction

1.1. The concept of ecosystem services

The notion of ecosystem services comes forth from the consciousness of the dependency of mankind on nature, which is obviously as old as mankind itself. Ancient civilizations such as the classical Greeks already noticed that human activities could cause destruction of natural resources of which they are dependent (Gómez-Baggethun et al. 2010). Attempts to capture this dependency in measurable units appeared only in the 20th century, with the observations that human activities had a detrimental impact on biodiversity and that services provided by nature were given too little weight in decisions compared to economic services (Costanza et al. 1997). The label 'ecosystem services' – the benefits people obtain from ecosystems (MEA 2005) – can be traced back in literature as early as the 1970s-80s (Gosselink et al. 1974; Westman 1977; Braat et al. 1979; Maltby 1986). With the publications of the Millenium Ecosystem Assessment (MEA 2005), The Economics of Ecosystems and Biodiversity (TEEB 2010) and papers such as Costanza et al. (1997), ecosystem services started to become more mainstream. This has led to an exponential increase in research in the field since the mid 2000s (Figure 1.1), accompanied by a proliferation of assessment methodologies (Seppelt et al. 2011; Crossman et al. 2013) and the development of multiple classification schemes (MEA 2005; Wallace 2007; TEEB 2010; CICES, EEA 2016; Böhnke-Henrichs et al. 2013).

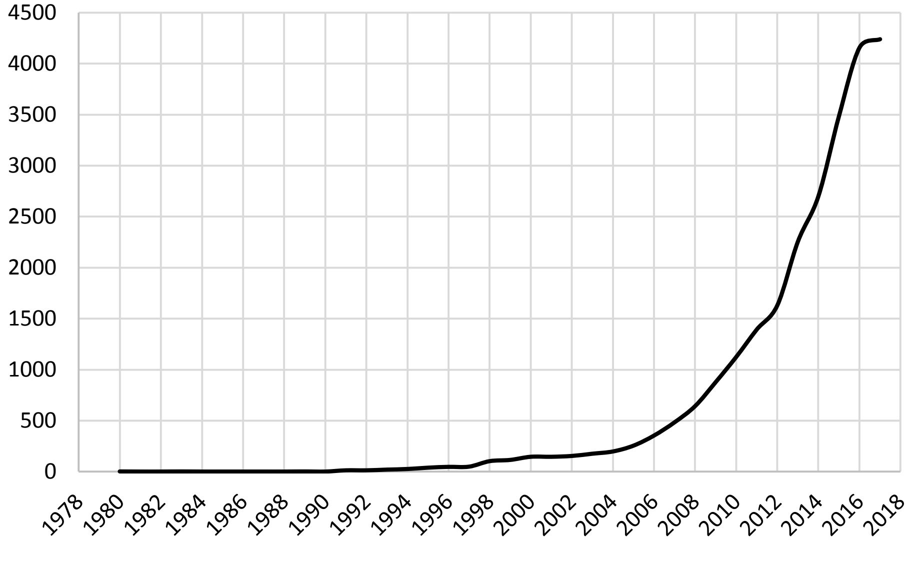

Figure 1.1 – Evolution of the number of citations on the topic 'ecosystem services' in Web of Science between 1980 and 2017.

Classification systems of ecosystem services are used to guide the identification and selection of ecosystem services in an assessment. Many systems have been developed, where the inclusion of supporting services constitutes the main point of discussion. The initial classification of the Millennium Ecosystem Assessment (MEA 2005) and TEEB (2010) distinguish 4 categories: provisioning, regulating, supporting and cultural (Figure 1.2). This is argued to be ambiguous since the ecosystem services as well as the processes which underlie services (referred to as supporting or intermediate services) were included at the same level (Wallace 2007), causing inconsistency (Nahlik et al. 2012) and a risk of double-counting (Boyd and Banzhaf 2007; Fisher et al. 2009). The European Environmental Agency (EEA) came up with an alternative system, called the Common International Classification of Ecosystem Services – CICES (EEA 2016). This classification comes forth from work on environmental accounting and distinguishes 3 categories: provisioning, regulating and cultural services (Table 1.1).

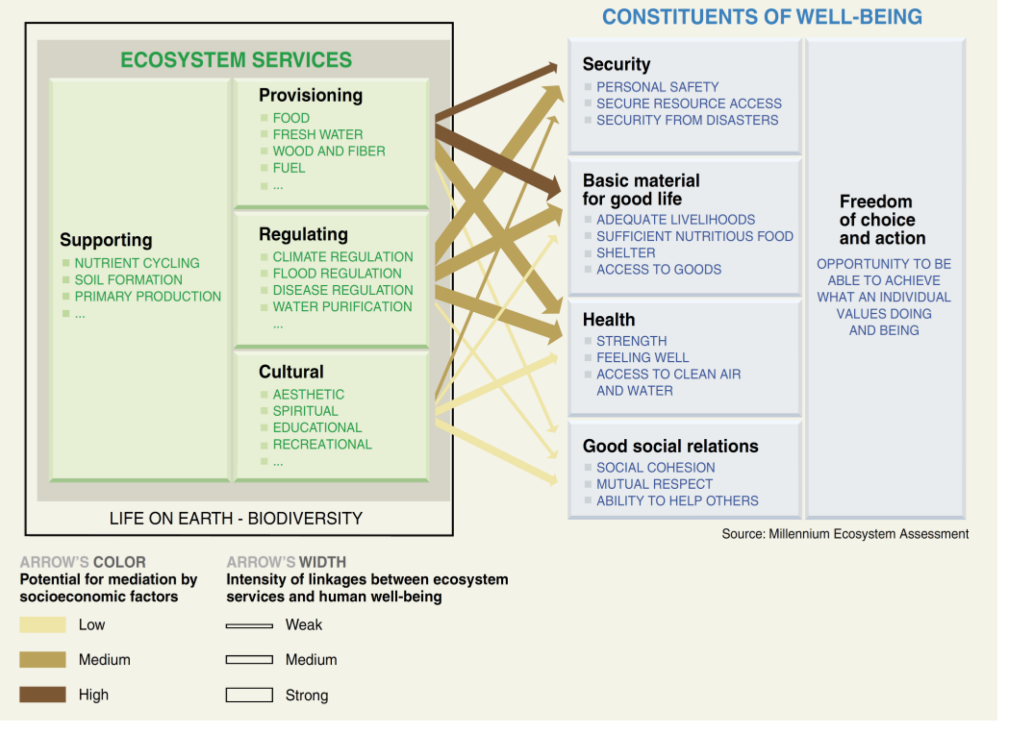

Figure 1.2 – The Millenium Ecosystem Assessment scheme showing links between different categories of ecosystem services and human well-being (MEA 2005)

Table 1.1 – Extract of the CICES V4.3 classification system (EEA 2016)

Category	**Division**	**Class**
Provisioning	Nutrition	Cultivated crops
		Reared animals and their outputs
		Animals from in-situ aquaculture
		Surface water for drinking
		…
	Materials	Fibres and other materials from plants, algae and animals for direct use or processing
		Materials from plants, algae and animals for agricultural use
		Genetic materials from all biota
		…
	Energy	Plant-based resources
		…
Regulation and Maintenance	Mediation of waste, toxics and other nuisances	Bio-remediation by micro-organisms, algae, plants, and animals
		Filtration/sequestration/storage/accumulation by ecosystems
		…
	Mediation of flows	Mass stabilisation and control of erosion rates
		Flood protection
		Storm protection
		…
	Maintenance of physical, chemical, biological conditions	Pollination and seed dispersal
		Maintaining nursery populations and habitats
		Decomposition and fixing processes
		Global climate regulation by reduction of greenhouse gas concentrations
		…
Cultural	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes	Scientific
		Educational
		Heritage, cultural
		…
	Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes	Symbolic
		Sacred and/or religious
		…

The choice of classification system is mainly driven by the endpoint of the assessment and the methodological approach. Although the central focus of this thesis is not on accounting, it builds on the CICES categories as these emphasize on final services, allowing to make a clear distinction between the underlying role of ecosystem functioning and the socio-economic benefits for humans (Figure 1.3). For the research in the coastal zone (Chapters 5 and 6), the typology of marine ecosystem services of Böhnke-Henrichs et al. (2013) was used to select additional marine-focused ecosystem services that were found to be lacking in CICES.

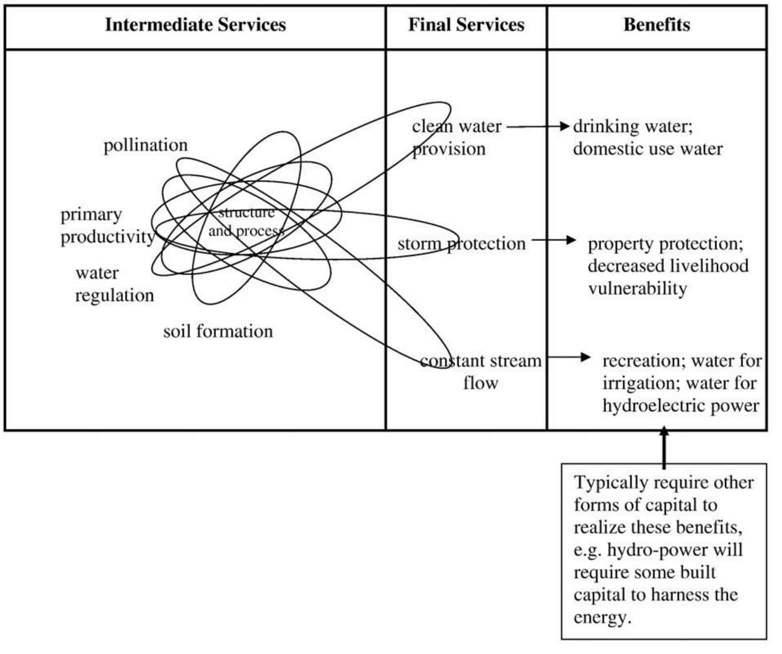

Figure 1.3 – Scheme of linkages between intermediate services (ecosystem functions), final services and benefits (Fisher et al. 2009)

A well-established way of representing the ecosystem services paradigm is the ecosystem services 'cascade' (Figure 1.4), originally developed by Haines-Young and Potschin (2010) and later revised by several authors (e.g. van Oudenhoven et al. 2012; Boerema, Rebelo et al. 2017). Ecosystems can be simplified as an entity with certain characteristics (ecosystem structures) in which things happen (ecological processes), together they constitute the ecosystem's properties. The products of ecological processes are changes in the ecosystem's structures or changes to what enters or leaves the ecosystem. Ecosystem structures and processes result in benefits for humans by improving or maintaining the quality of the environment in which humans live, by producing goods that humans can extract from the ecosystem or by enhancing the quality of those goods. This framework thus emphasizes the role of ecosystem services as a bridge between the ecological and the socio-economic system.

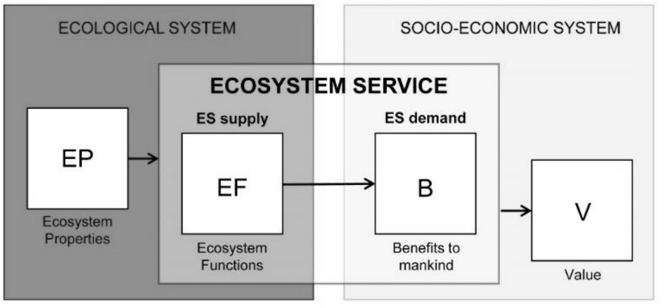

Figure 1.4 – Conceptual framework illustrating the steps in the production chain of ecosystem services (Boerema, Rebelo et al. 2017)

In the ecological field, ecosystem functions are a quite general term covering ecosystem structures as well as processes (Hooper et al. 2011). In the cascade, ecosystem functions refer to "those ecosystem structures and processes which humans find useful" (Haines-Young and Potschin 2010). A function may have the capacity to provide something useful to humans, but only when humans consider the function as a benefit, it is called an ecosystem service. Coniferous trees, for example, have a large capacity to adsorb fine dust particles onto their needles and branches. However, only in urbanized areas where there is a source of pollution and where people live or work, the trees will actually deliver the service of air purification. The words 'supply' and 'demand' are also often used in an ecosystem services context. 'Supply' refers to what is present in the ecosystem and can be useful to humans (ecosystem function). The 'demand' for an ecosystem service represents the amount of the service which humans require and may be larger than the 'supply'. Only that part of the 'supply' that is covered by the 'demand' is to be considered an ecosystem service.

The advantage of using this framework, is that it breaks up the production chain of ecosystem services into quantifiable entities for which indicators can be identified (Boerema, Rebelo et al. 2017). An indicator in ecology and environmental planning can be defined as "a measure of environmentally relevant phenomena used to depict or evaluate environmental conditions or changes or to set environmental goals" (Heink and Kowarik 2010). In case of ecosystem services, they can be used as signals of changes in complex human-enviornmental systems (Müller et al. 2012). They can either be a single variable (e.g. the value of climate regulation is directly linked to land use), or, when multiple variables are used, they can be combined into a composite indicator (e.g. the value of climate regulation is defined by a combination of land use, soil texture, groundwater depth and age of the ecosystem). Indicators can be based on structural properties (e.g. land use) or on ecological processes or ecosystem functioning (e.g. sand transport). The focus in this thesis is on elaborating methods that link ecosystem functioning to human well-being.

1.2. Endpoints of an ecosystem services assessment

An ecosystem services assessment is a qualitative, quantitative or monetary evaluation of the ecosystem services provided by an ecosystem or of the changes in ecosystem services related to changes in the ecosystem. As mentioned earlier, ecosystem services assessments originally had a strong focus on raising awareness on people's dependency of nature (Norgaard 2009), which, with industrial and technological advances, had increasingly been ignored. The exponential increase in research in the field since the mid 2000's (Figure 1.1) has led not only to a proliferation of assessment methodologies and classification systems, but also to a wide exploration of the potential applications where an ecosystem services assessment can add value. Four main objectives can be identified of performing an ecosystem services assessment: (1) raising awareness on the human dependency of nature, (2) financial or social motivations to account for losses or gains of interventions or changes in an ecosystem, (3) spatial management to optimally allocate ecosystem services and (4) methodological advantages over other approaches such as the variety of effects than can be considered. In practice, most ecosystem services assessments will have multiple objectives.

One of the most well-known examples of ecosystem services assessments with a main purpose of awareness raising is the publication of Costanza et al. (1997), which has been cited over 19,000 times (Google Scholar, November 2017). Other authoritative studies that have been successful in demonstrating the global or national benefits of biodiversity and the costs associated with loss of biodiversity are TEEB (2010) and National Ecosystem Assessments (e.g. UK, Flanders, Netherlands, …). Awareness raising remains a crucial and valuable objective of many ecosystem services assessments today (Guerry et al. 2015; Maes et al. 2016). The target audience for these types of analyses can vary from the political and decision-making level to different stakeholders and the general public. Environmental and natural capital accounting (e.g. System of Environmental-Economic Accounting, UN 2017; Natural Capital Coalition 2016) take awareness raising a step further by offering an instrument by which the value of ecosystem services can be integrated into national accounting and business accounting, which are overlooked in traditional economic accounting (de Groot 1987; Boyd and Banzhaf 2007).

Examples of ecosystem services assessment with financial or social purposes are cost-benefit analyses or cost-effectiveness analyses, payments for ecosystem services and compensations for environmental damages. Cost-benefit analyses make a comparison between the investment costs in and the economic benefits of a project or a management option (Pearce et al. 2006). Cost-effectiveness analyses differ from cost-benefit analyses in the sense that they do not require the use of monetary values (Balana et al. 2011). Cost-benefit and cost-effectiveness analyses can be useful to underpin the value of investing in the ecosystem's functioning instead of using technical solutions, and similar to environmental accounting pinpoint benefits of nature that are overseen in traditional cost-benefit and cost-effectiveness analyses. Payments for ecosystem services are compensations to landowners for managing their property to provide certain ecosystem services (Kallis et al. 2013). The German producer of organic non-alcoholic beverages ‘Bionade’, for example, paid the costs to convert conifers to broadleaves to protect their drinking water resource. A well-known example in Belgium are governmental subsidies for grass buffer strips along agricultural fields to reduce erosion and runoff. Such schemes may increase the willingness of owners to protect natural resources from which they otherwise may not directly benefit or even suffer economic losses, and thus lack conservation incentive. Compensation for environmental damages, or also known as the polluter-pays principle (Balana et al. 2011), is another example to apply the ecosystem services framework.

One of the challenges in spatial planning is to truly incorporate biodiversity values (Geneletti 2008). Biodiversity conservation has long been perceived as the sole purpose of protected areas, and nature values outside these areas were ignored (Geneletti 2008; Agardy et al. 2011). This has led to loss of biodiversity assets outside protected areas and in certain cases undermined support for nature conservation as it was perceived to conflict with economic development (Beunen et al. 2013). The inclusion of ecosystem services in spatial planning and in protected areas designation has increasingly been acknowledged to improve the consideration of biodiversity in spatial planning (Ingram et al. 2012). Ecosystem services allow to make socio-economic benefits of nature, whether expressed in monetary or in non-monetary terms (e.g. health effects), explicit, both inside and outside protected areas. They thus provide a means to align socio-economic and biodiversity conservation targets and support the development of multi-functional landscapes. Integration of ecosystem services is possible at different levels of planning processes: in early phases to define strategic objectives (Adriaensens et al 2005; Geneletti et al. 2011; Partidário and Gomes 2013), to guide and advice on-site management to implement strategic plans (de Groot et al. 2010) or to assess impacts of a project development in environmental impact assessments and develop alternative nature-based solutions (Benson 2012; Partidário and Gomes 2013).

The concept of ecosystem services is also used for more methodological reasons related to advantages over alternative ecosystem assessment approaches. For example as a framework for facilitating interdisciplinary research (Boerema 2016), as a tool to support stakeholder participation and communication, to balance multiple objectives and choose between trade-offs (Fisher et al. 2009; Jennings and Rice 2011) or because of its holistic character including a wide range of socio-economic and ecological aspects (Ingram et al. 2012).

1.3. Outputs of an ecosystem services assessment

There are several reasons why the dissemination of the ecosystem services concept in research and management has led to a proliferation of assessment methodologies. Amongst others are the variety in the purpose of an ecosystem services assessment, the perspective from which the assessment is carried out (supply or demand side), the availability of data and research, the diversity of ecosystem services and the field of expertise in which the study is performed (Seppelt et al. 2011; Crossman et al. 2013; Wong et al. 2015). Methods can be categorized into three broad groups, according to the unit of the output: qualitative, quantitative and monetary assessments (Figure 1.5).

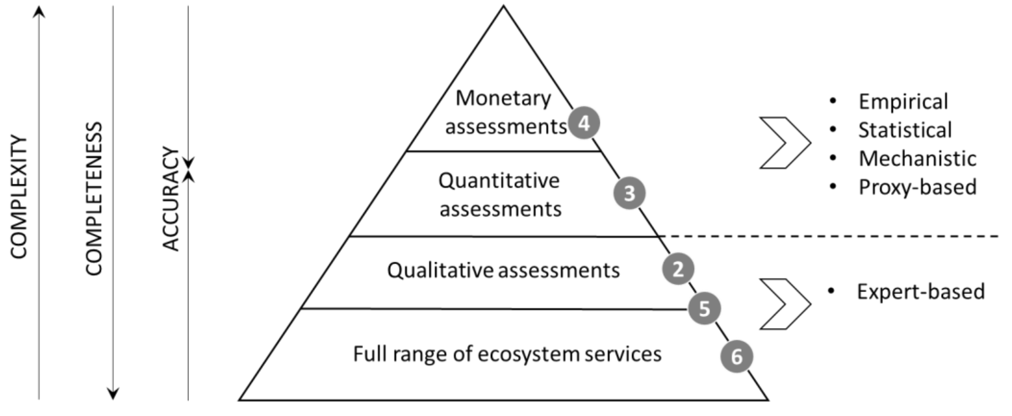

Figure 1.5 – Indication of the change in number of services which can be evaluated per assessment type (pyramid adapted after Kettunen et al. 2009) and commonly used assessment methods. A trade-off exists between completeness and complexity of the method and between completeness and accuracy. The numbers in circles indicate the thesis chapter where the method is discussed (see 1.7 Reading guide).

The outcome of a qualitative assessment is generally a score or a description which expresses the amount of a service that is or that can be delivered given a certain environmental condition (e.g. habitat) or a combination of conditions (e.g. land use and soil type). The assessment is based on expert judgement which can be done by scientists, stakeholders or locals. This method can be expected to be the least accurate, as it strongly depends on the knowledge of the experts, it is prone to subjectivity and results are rarely validated. It has however advantages that it can provide an estimate for almost all services and it generally doesn't require large investments in research (Burkhard et al. 2009). Qualitative assessments are often used in data scarce environments, to be able to include all relevant ecosystem services, for large-scale assessments, for a quick identification of affected services or for regularly repeated assessments.

Quantitative assessments express the delivery of an ecosystem service in a unit which is most meaningful for that service. Examples are the amount of carbon sequestered in the soil per hectare per year, or the number of avoided casualties by natural flood defence. Quantitative assessments can be based on empirical data (e.g. field measurements, surveys, …), where an appropriate indicator is directly measured in the field. Empirical data can also be used to develop statistical models (e.g. regression models) that describe relationships between sampled data of environmental properties (explanatory variables) and sampled data of ecosystem services (response variables).

Mechanistic models (e.g. hydrological models) quantify ecosystem services based on mathematically defined or well-known causal relationships between environmental variables and ecosystem services (Martínez-Harms and Balvanera 2012). The most accurate predictions are obtained using empirical measurements of ecosystem services, or otherwise called primary data (Eigenbrod et al. 2010). This is also the most labour-intensive and costly method, which explains why it is used less frequently (Martínez-Harms and Balvanera 2012). Statistical and mechanistic models are able to partially overcome this problem as they allow to cover larger areas than the sampling area, based on an extrapolation of statistical relationships or mathematical formula describing ecosystem processes. The advantage is that explanatory variables are often more commonly studied or modelled than ecosystem services in itself (e.g. groundwater levels) or they can be derived from readily available ancillary maps (e.g. soil map). Such models are generally more generic and simplified, but should build on well-established knowledge and consensus.

Monetary assessments attach an economic value to the delivery of a service (e.g. € per hectare per year). Common valuation methods include market price (mostly applied to provisioning services), replacement costs (e.g. restoration of flood defence by beach nourishments), costs of alternatives (e.g. costs of waste water treatment stations), avoided damage costs (e.g. avoided flood damage to houses), revealed preference (e.g. costs to travel to a recreational site) (TEEB 2010). Most often they are derived from a quantitative estimate of the ecosystem service, by multiplying with its economic value per unit (e.g. production of 1 m^3 drinking water = 0.2 €). This value is derived from local data, or from data of other places with similar socio-economic and ecological conditions (cfr. 'benefit transfer'). Monetary assessments can also be based on direct measurements (e.g. using stated preference surveys), such as in social valuation of ecosystem services. In this case, monetary assessments can cover a much larger number of ecosystem services than reflected by the top of the pyramid, since preference statements of stakeholders do not require quantitative information to be available.

Monetary assessments seem to have the advantage that they allow comparison among different ecosystem services as they are all expressed in a single unit (e.g. €). However, this is not completely true as the social and environmental benefits are not always fully accounted for and the true value of a good or service can thus be underrated. For example, assuming a family on average consumes 3500 litre water per week and the maximum value for the ecosystem service water production is 0.2 € per m^3 (Broekx et al. 2013a). The monetary value for the service delivered to the family would be only 0.7 € per week. However, they would not be able to survive a week without water and the true value of the water is thus much higher than its economic value. For some services monetarisation is inadequate and misleading (TEEB 2010), such as the spiritual value of a landscape. Only taking into account those services which are (easily and adequately) measurable can result in biased conclusions (Seppelt et al. 2012). Many other critiques exist on the economic valuation of ecosystem services, such as the impossibility to include the intrinsic value of nature, the promotion of exploitative management, commodification of ecosystems and the high degree of uncertainty for some services and methods (Gómez-Baggethun and Ruiz-Pérez 2011; Schröter et al. 2014; Neuteleers and Engelen 2015). Nevertheless, monetary valuation in certain cases can be a sound choice, for example because of its influential power in some sectors (e.g. media), the potential to claim compensation for environmental damages and its awareness-raising capacity (Kettunen et al. 2009; Jax et al. 2013; Kallis et al. 2013; Boerema 2016).

A trade-off exists between complexity and completeness in ecosystem services assessments (Figure 1.5). Quantitative and monetary assessments generally require more data, time and means, resulting in a reduction in the number of services that can be included in an assessment (Kettunen et al. 2009). The base of the pyramid reflects the full range of acknowledged ecosystem services, including those services which cannot be evaluated, for example due to lack of knowledge or adequate methods. A trade-off also exists between completeness and accuracy. In case monetary assessments are performed by multiplying a monetary value with the outcome of a quantitative assessment, their accuracy will decrease because of error propagation.

1.4. Ecosystem services and biodiversity in spatial planning

The integration of biodiversity in spatial planning traditionally revolved around species and habitat conservation (e.g. NATURA2000 since 1992 in Europe). With the development of concepts such as ecosystem services, the rationale behind nature conservation started to have a secondary objective besides protecting 'nature for itself' or for its intrinsic value, i.e. protecting nature for its life-insurance value (Schneiders and Müller 2017) – 'nature for people' (Mace 2014). The utilitarian framing of nature however has created dispute among conservationist (Schröter et al. 2014) and has even been argued to result in biodiversity loss (Cardinale et al. 2012). Another argument often used to protect nature for itself is that efforts to increase biodiversity values by definition increase ecosystem services (Mace et al. 2012). The relationship between ecosystem services and biodiversity however has been subject of debate in the past decades. Although evidence supports a positive link in general (Hooper et al. 2011; Harrison et al. 2014), many cases exist where they are not or are negatively correlated (Mace et al. 2012; Bennett et al. 2015; Sandbrook and Burgess 2015; Manhaes et al. 2016; Ricketts et al. 2016). The precise relationships depend, amongst others, on the biodiversity facet that is considered, i.e. species richness, functional diversity, … (Díaz et al. 2007); the aspect in the ecosystem services cascade that is used, i.e. underlying ecosystems functions or the final socio-economic benefits (Duncan et al. 2015); whether or not ecosystem disservices are included such as increased diversity of disease-causing bacteria (Sandbrook and Burgess 2015) and whether long-term resilience of ecosystem functioning is taken into account (Oliver et al. 2015).

It is now increasingly recognized by science and policy that including socio-economic benefits together with biodiversity values improves effectiveness of environmental planning and biodiversity conservation (Adriaensens et al. 2005; Leenhardt et al. 2015; Paudyal et al. 2016).

International environmental policies and platforms that adopt ecosystem services include the EU Biodiversity Strategy to 2020, the global Strategic Plan for Biodiversity (2011-2020) of the Convention of Biological Diversity (CBD) and the Integrated Platform on Biodiversity and Ecosystem Services established in 2012 (IPBES). Examples in spatial planning practice however are still scarce (Potts et al. 2013; Ortiz-Lozano et al. 2017). A lack of transdisciplinary thinking (Leenhardt et al. 2015) and tools (Grêt-Regamey et al. 2017) allowing for a holistic approach including social-ecological interactions, and a lack of guidance on prioritizing objectives and dealing with inevitable trade-offs (Jennings and Rice 2011; Lester et al. 2013) are mentioned as important reasons why this remains challenging.

Also in impact assessments and strategic environmental assessments there is a growing interest in using ecosystem services as a tool to increase environmental consideration in decision-making (Geneletti 2016). An ecosystem services approach has the potential to deal with several shortcomings in current impact assessments (Geneletti 2011; Honrado et al. 2013), such as a shift of emphasis on avoiding negative impacts to creating opportunities (Goodstadt et al. 2010; Baker et al. 2013), a more clear representation of the added value of preventing environmental risks (Bowd et al. 2015; Mandle et al. 2016; Hattam et al. 2017), a cross-sectoral consideration of effects (Baker et al. 2013; Hattam et al. 2017) and an integration of stakeholders. However, analytical tools and frameworks to facilitate this are lacking (Helming et al. 2013; Geneletti 2015)

1.5. Need for practical tools based on ecosystem functioning

To support the integration of ecosystem services into different management and policy processes, practitioners need user-friendly instruments (Broekx et al. 2013b; Wissen-Hayek et al. 2015; Grêt-Regamey et al. 2017). Proxies of static ecosystem structures (e.g. land use) have been widely applied to assess and map ecosystem services because of their ease of use (Eigenbrod et al. 2010b; e.g. Sutton and Costanza 2002, Burkhard et al. 2009, Vihervaara et al. 2010, Schneiders et al. 2012). Such proxies make a direct link between a structural ecosystem property and the (potential) delivery or the value of an ecosystem service and entirely discard ecosystem functioning. Their over-generalization makes it easy to apply the estimates in other locations and/or contexts for which primary data is lacking (Landsberg et al. 2011; Vrebos et al. 2015), or otherwise called ‘benefit transfer’ (Plummer 2009). Many studies however point to the risks associated with using simplified proxies (Kienast et al. 2009; Eigenbrod et al. 2010a; Lautenbach et al. 2011; Geijzendorffer and Roche 2013) or generalization errors related to benefit transfer (Plummer 2009; Eigenbrod et al. 2010b). It has even been argued that ignoring the causal relationships between ecosystem functioning and ecosystem services may lead to losses of what the ecosystem services concept actually aims to conserve (Eigenbrod et al. 2010b; Wong et al. 2015).

Following the exponential increase in ecosystem services research in the 2000s (Figure 1.1) and the proliferation of assessment methods, **the integration of knowledge on ecosystem functioning into ecosystem services assessments was identified as key research priority** (Kremen 2005; Nicholson et al. 2009; Verburg et al. 2009; Norgaard 2010; Potschin and Haines-Young 2011; Seppelt et al. 2011; Maes et al. 2012; Wong et al. 2015). Recognizing, understanding and modelling the underlying processes has increasingly been acknowledged as key to predicting changes in ecosystem services (Nicholson et al. 2009; Lavorel et al. 2017). Several reasons are cited why ecosystem services assessments should be based on ecological theory: to avoid ineffective conservation efforts (Lavorel et al. 2011), to reduce uncertainty of the estimates (Lavorel et al. 2017), to avoid biased outcomes (Menzie et al. 2012) and to create credibility of the results (Wong et al. 2015).

Although a wide variety of specialised ecological models exists that allow to make a detailed prognosis of the delivery of a single ecosystem service based on ecosystem functioning, the extensive data requirements and specific modelling skills needed for each model type form an obstacle for their application in institutional contexts (Villa et al. 2014) where interest generally lies in multiple ecosystem services (Bennett et al. 2009). Also, ecological models mostly don't consider interactions with other ecosystem services which makes it difficult to look at the system from a more integral socio-ecological perspective. It is therefore crucial to condense the complex ecological information and integrate it with aspects of the sociological system into a practical and flexible tool, to ensure that it is actually used in practice, but without compromising on scientific integrity. A user-friendly tool also allows repeated analyses, supporting iterative science-policy processes in spatial planning (Ruckelshaus et al. 2015; Grêt-Regamey et al. 2017). **An important trade-off thus exists between accuracy and practicability, impeding the integration of ecosystem services at different levels of environmental management and challenging the development of effective management tools.**

To avoid a 'black box' impression among practitioners caused by a lack of understanding and confidence it is important that the tool is transparent. Another aspect which is important in this sense is the recognition and communication of uncertainty on ecosystem services predictions. Although it may strongly affect decision-makers' choices (Grêt-Regamey et al. 2013), less than a third of ecosystem services studies actually quantifies uncertainty (Seppelt et al. 2011; Boerema, Rebello et al. 2017). A challenge for decision-makers lies in making sound decisions under data scarcity and uncertainty. A practical tool should therefore be able to inform decision-makers on uncertainty and guide them in coping with uncertainty when choices need to be made.

Finally, the interdisciplinary nature of the ecosystem services concept requires involvement of specialists from different research niches, including experts in socio-economic domains, to elaborate the tools. Ecosystem services assessment tools should allow stakeholder involvement in different stages of the development of the instrument (Seppelt et al. 2011) and, depending on the purpose of the assessment, also in the application phase to support balanced decision-making and create shared goals. An important aspect in this matter is the capacity of the tool to **identify conflicts and synergies between ecosystem services and with biodiversity values based on understanding of underlying processes, and guide in the selection among trade-offs**.

Around the start of this doctoral research in 2010, Bayesian networks were recently introduced in ecosystem services modelling (Landuyt et al. 2013). One of the reasons why they gained increasing attention is the possibility to make predictions when information on the state of an input variable is missing, whereas unavailable data in numerical models results in ineffective or inaccurate reasoning (CharlesRiver Analytics Inc. 2008). Another major strength is the possibility to take into account uncertainty (Aguilera et al. 2011; Grêt-Regamey et al. 2013), which is a shortcoming of many ecosystem services studies. They also allow to combine multiple assessment methods (Figure 1.5), making them especially interesting for ecosystem services that cover a wide variety of knowledge domains (soils, plants, water flow, health, etc.). However, only one out of 48 papers mentioned in the review of Landuyt et al. (2013) considers more than 2 ecosystem services in a single Bayesian network. In order to spatially allocate land use and management measures to optimize ecosystem service delivery it is necessary to include all relevant ecosystem services as well as the underlying mechanisms and structures that produce ecosystem services and cause trade-offs among ecosystem services. Given the advantages of Bayesian networks in ecosystem services assessments related to uncertainty and integration of knowledge from different sources as demonstrated in earlier research, **the potentials of Bayesian networks to integrate multiple ecosystem services and to use for scenario development remained largely unexplored**.

1.6. General aim and research questions

This dissertation aims to develop practical methodologies to improve the consideration of ecosystem functioning in ecosystem services assessments and to support decision-making processes in Flanders through implementation of the methodologies in practice. This is done by testing the accuracy of ecosystem services assessment methods based on structural indicators, by exploring the potentials of Bayesian networks to integrate ecosystem-functioning knowledge for multiple ecosystem services and by developing two novel process-based approaches for spatial planning and impact assessments (Figure 1.7). The methodologies, with exception of the first, respond to shortcomings at different levels of current environmental management (local management, spatial planning and impact assessments), but have in common that they are based on ecosystem functioning, practical in use, allow to involve stakeholders and support the development of multifunctional landscapes providing a diversity of ecosystem services.

Three main research questions have been formulated to guide the study:

Research question 1

Does a highly detailed classification increase the accuracy of land use as a single structural indicator of multiple ecosystem services?

Research question 2

How useful are Bayesian networks to develop practical and scientifically underpinned ecosystem services assessment tools?

Research question 3

How can a focus on ecosystem processes advance the integration of ecosystem services at different levels of environmental management?

1.7. Reading guide

The research was performed in different case-studies in Flanders, the north of Belgium (Figure 1.6). These include a lowland landscape with wetlands, brooks and land dune relicts in the Grote Nete catchment (Chapter 3), the Westhoek nature reserve of coastal dunes on the border with France (Chapter 4), the entire coastal zone with dunes, estuaries and tidal flats on the land side and the continental shelf with sandbanks on the sea side (Chapters 5-6), and the whole of the Flemish region (Chapter 2). Two main approaches are elaborated upon (Figure 1.7): a structural indicator-based approach (Chapter 2) and an ecosystem-functioning approach (Chapters 3-6). The variety of spatial scales of the study-areas results from the fact that this research has evolved from different applications in management rather than having a purely science-driven objective. Elaborating the methodologies on different spatial scales allows to show the advantages and challenges of integrating ecosystem services at different stages of decision-making, going from spatial allocation of management choices to more strategic purposes.

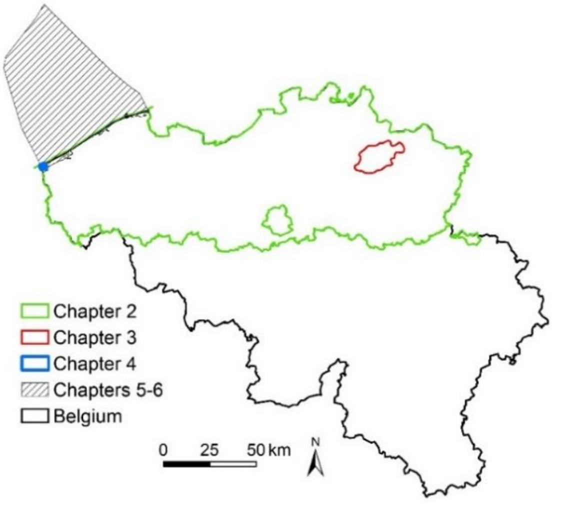

Figure 1.6 – Study areas of the different chapters

The introduction provided a state-of-the-art in ecosystem services research related to tools supporting the inclusion of ecosystem services in environmental management and policy and identifies current shortcomings and challenges.

In search for appropriate ways to apply ecosystem services in environmental planning, the accuracy of one of the most commonly used assessment methods, i.e. land use as a structural indicator of ecosystem services, is tested in Chapter 2 (Figure 1.7). This is done by a statistical comparison of ecosystem services maps of Flanders based on land-use with maps created using biophysical models.

The potentials of Bayesian networks as a modelling environment to develop practical tools for ecosystem services optimization on a landscape scale are explored in Chapter 3 for a case-study in the upstream part of the Grote Nete catchment. Three ecosystem services are included based on knowledge derived from biophysical models. The capacities of Bayesian networks to integrate ecosystem-functioning knowledge from different sources and to develop scenarios for optimized service delivery are discussed.

Chapters 4-6 focus on the role of ecosystem processes in the management of ecosystem services. In Chapter 4 the use of ecosystem processes as indicators of ecosystem services is evaluated. This is based on a monetary valuation of the contribution of sand transport in maintaining ecosystem services in coastal dunes in Belgium.

Chapters 5 and 6 present two novel methodologies. Chapter 5 elaborates upon a method to align biodiversity conservation and ecosystem services targets using ecosystem processes as a common denominator. Its capacities to deal with shortcomings in current spatial planning are discussed based on a case of the Belgian coastal zone.

Chapter 6 expands on a methodology to integrate ecosystem services in impact assessments and discusses its merits and constraints using the Belgian coastal zone as a case-study.

Chapter 7 frames the performed research in relation to trends in ecosystem services research and summarizes the main achievements. Finally, some management recommendations and opportunities for further research are discussed.

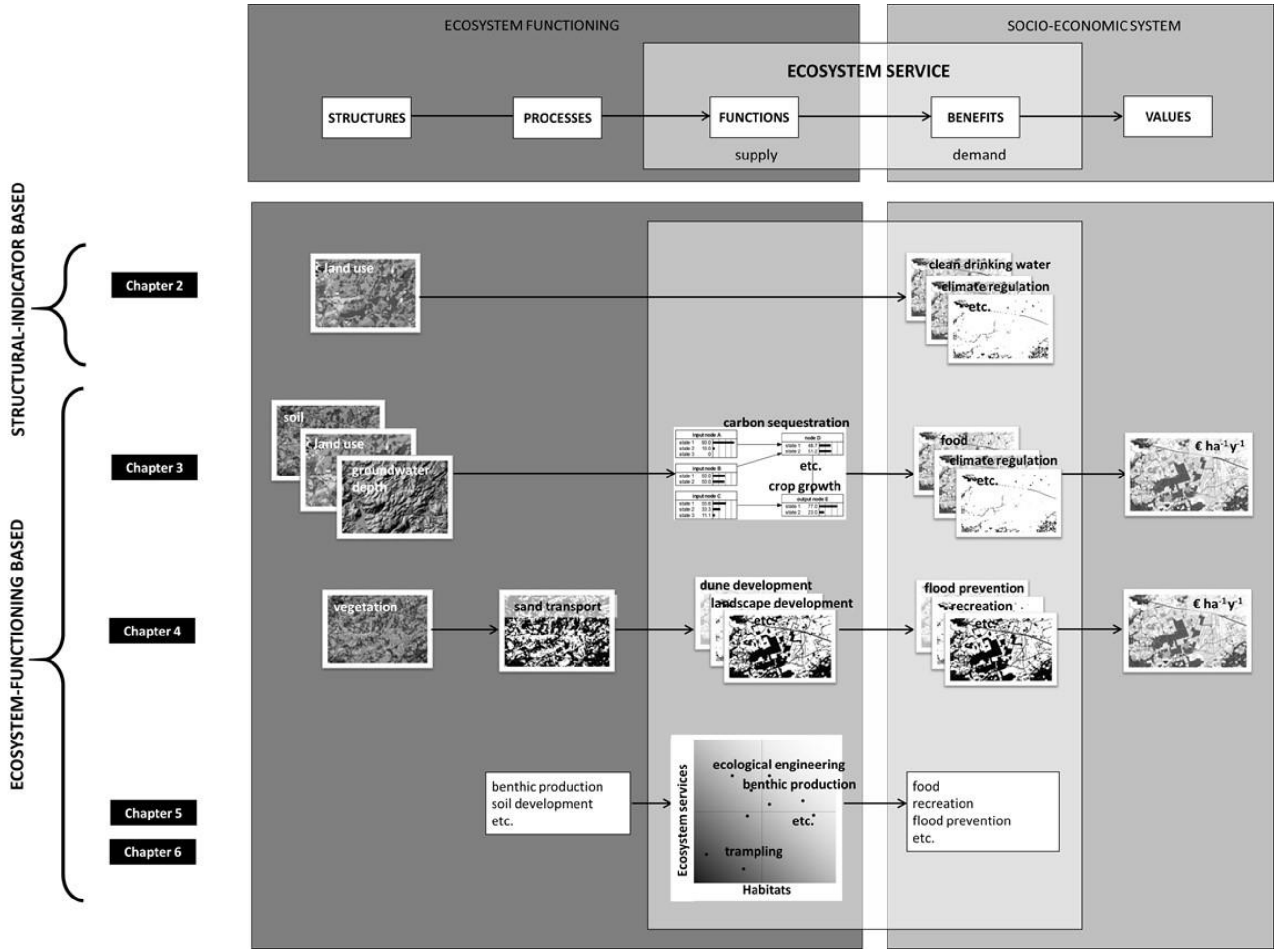

Figure 1.7 - Schematized overview of the dissertation with reference to the ecosystem services cascade as presented in Boerema, Rebelo et al. (2017)

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2

Mapping ecosystem services based on structural indicators

Published as Van der Biest K., Vrebos D., Staes J., Boerema A., Bodí M.B., Fransen E. and Meire P. (2015). Evaluation of the accuracy of land-use based ecosystem service assessments for different thematic resolutions. Journal of Environmental Management (156), 41-51

2.1. Abstract

The demand for pragmatic tools for mapping ecosystem services has led to the widespread application of land-use based proxy methods, mostly using coarse thematic resolution classification systems. Although various studies have demonstrated the limited reliability of land use as an indicator of service delivery, this does not prevent the method from being frequently applied on different institutional levels. It has recently been argued that a more detailed land use classification system may increase the accuracy of this approach. This research statistically compares maps of predicted ecosystem services delivery based on land use scoring for three different thematic resolutions (number of classes) with maps of ecosystem services delivery produced by biophysical models. Our results demonstrate that using a more detailed land use classification system does not significantly increase the accuracy of land-use based ecosystem services assessments for the majority of the considered ecosystem services. Correlations between land-use based assessments and biophysical model outcomes are relatively strong for provisioning services, independent of the classification system. However, large discrepancies occur frequently between the score and the model-based estimate. We conclude that land use, as a simple indicator, is not effective enough to be used in environmental management as it cannot capture differences in abiotic conditions and ecological processes that explain differences in service delivery. Using land use as a simple indicator will therefore result in inappropriate management decisions, even if a highly detailed land use classification system is used.

2.2. Introduction

With the concept of ecosystem services finding its way into impact assessments, spatial planning and national nature and environment monitoring (de Groot et al. 2010; Landsberg et al. 2011; Maes et al. 2011), the need for straightforward assessment tools is clear (Kienast et al. 2009; Van der Biest et al. 2014). These tools should allow for consistent ecosystem services evaluation through time and across different levels of practice. The demand for such methods has stimulated the emergence of land-use based proxy methods for assessing the capacity of a landscape to deliver services, mostly using coarse thematic resolution classification systems (Burkhard et al. 2009; Burkhard et al. 2012). While these methods are powerful awareness-raising instruments, applying them on the level of decision-making may have adverse effects.

Despite the many studies showing the errors caused by using simplified proxies for ecosystem services delivery (Kienast et al. 2009; Eigenbrod et al. 2010; Lautenbach et al. 2011; Geijzendorffer and Roche 2013; Hou et al. 2013), land use is still frequently applied as an indicator of service delivery (Maes et al. 2011; Maes et al. 2012; Nedkov et al. 2012; Schneiders et al. 2012). Although proxy-based methodologies are generally developed to be used on large spatial scales (Naidoo et al. 2008; Haines-Young et al. 2012; Maes et al. 2012), for awareness raising or as a starting point for more thorough assessments (Burkhard et al. 2012; Vihervaara et al. 2012), there is a danger that their easy application leads to the concepts being used outside of these contexts (Landsberg et al. 2011).

It has recently been argued that a more detailed land use classification system may increase the accuracy of land-use based ecosystem services assessments (Vihervaara et al. 2010; Burkhard et al. 2012; Vihervaara et al. 2012). Explanations for this are that coarse thematic resolution classification systems may not identify small, region-specific habitats that are of particular importance for the delivery of certain ecosystem services (Vihervaara et al. 2010; Schulp and Alkemade 2011). Other arguments are the limited delineation of too-general classes (Vihervaara et al. 2010). Schulp and Alkemade (2011) argue that taking information on the spatial organization of the landscape into account would increase the accuracy of ecosystem services assessments. They also suggest including information on land cover as well as land use in the classification system.

In this study, we investigate the accuracy of existing land use classification systems for predicting ecosystem services delivery by comparing the results of a land-use based scoring method with the results of quantitative biophysical models. We then verify whether or not a finer thematic resolution and delineation of land use categories for the purpose of ecosystem services research improves the precision of land-use based ecosystem services assessments. Lastly, we analyze the extent to which land use can be used to make predictions on ecosystem services delivery and whether differences exist between the predictive capacity for provisioning and regulating services.

2.3. Materials and methodology

2.3.1. Study area

The analysis was carried out in the Central Campine ecoregion, an area of ~ 1,100 km² located in North East Belgium (Figure 2.1). The area is a typical low-relief lowland (about 30m maximum elevation gain). Its soils mainly consist of sand and loamy sand, with loamy and boggy soils on alluvial plains and coarse sand on a series of inland dunes. The high population density (470 inhabitants/km²) and historic land use make it a highly fragmented landscape (Antrop 2004) with a median parcel size of less than 0.1ha (residential property included). The western part of the study area covers the densely-populated suburban area around the city of Antwerp. In the East, buildings are scattered and follow typical ribbon development patterns, connecting villages throughout a landscape that is dominated by agriculture and forestry. Several small nature reserves are found within the study area.

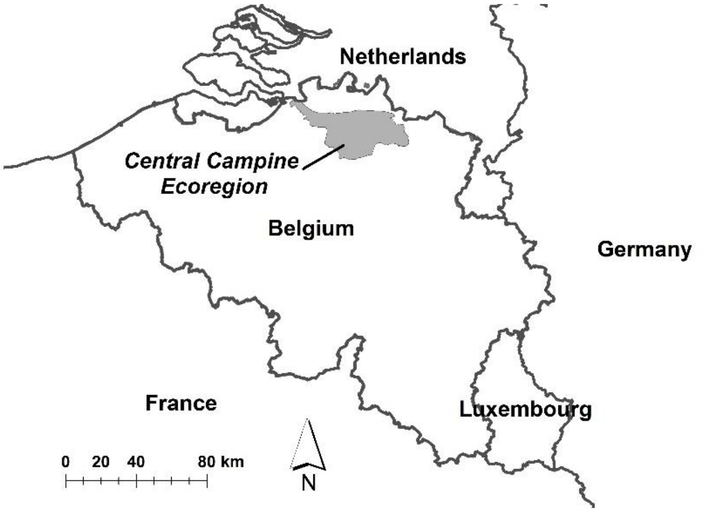

Figure 2.1 – Map of Belgium indicating the study area (Sevenant et al. 2002)

2.3.2. Land use classification systems

The strength of correlation between land-use based qualitative scoring and process-based quantitative modeling of ecosystem services is analyzed for three land use classification systems with different thematic resolutions and mapping targets. These three land use classification systems differ in two crucial aspects, i.e. number of classes (thematic resolution) and purpose of classification.

The **CORINE classification system** (Coordination of Information on the Environment, EEA 2007) was selected because of its frequent use in ecosystem services assessments (Burkhard et al. 2009; Vihervaara et al. 2010; Maes et al. 2011; Burkhard et al. 2012; Maes et al. 2012; Zulian et al. 2014). It was originally developed in 1994 to represent the different European ecosystems and to enable European-level cross-border investigation and comparison (EEA 1999). This study used the 2000 update of the classification system (CLC2000). Table 2.1 gives an overview of the original number of classes per main land use category for the corresponding version of the 2007 raster map (CLC2000 100m x 100m - version 9/2007).

The system developed by Gobin et al. (2009), hereafter referred to as the **governmental classification system**, was selected because of its frequent use in policy development in the Flemish region (Gobin et al. 2009; Schneiders et al. 2012; Staes et al. 2014). The main way it differs from CORINE is in its combination of land cover information with management-related aspects (e.g. environmental targets, multifunctional targets, agricultural management, biodiversity management). The number of classes per main land use category distinguished on the original map (100m x 100m) of the study area is summarized in Table 2.1.

Table 2.1 – Number of classes occurring within the study area per main land use category for the three different original land use maps

	Number of classes					
Land use map	**Urban**	**Agricultural**	**Forest & semi-natural**	**Wetland**	**Water body**	**Total**
CORINE	9	5	5	1	2	22
Governmental	16	7	5	2	1	32
ESLUC	18	11	23	8	5	65

A third classification system was developed for the purposes of this study, specifically for ecosystem services mapping. It is hereafter referred to as the **Ecosystem Services Land Use Classification (ESLUC)**. It aims to provide a high level of detail, which is expected to allow for more accurate estimates of ecosystem services. A small group of experts (5) involved in ecosystem services research were asked to identify all the land cover classes they believed to differ at the level of service delivery. Since the majority of ecosystem services are controlled not only by land use or vegetation type but also by the abiotic environment, socio-economic factors and spatial relationships (Lautenbach et al. 2011; Nedkov et al. 2012; Hou et al. 2013), an 'ideal' land use classification system for ecosystem services should include distinctive categories reflecting the differences between these elements. The level of detail of the classification system, however, had to be adapted so that the system could be used on different institutional levels (Kienast et al. 2009; Hou et al. 2013). A classification system with a very high level of detail is impractical and may lead to erroneous interpretations when the classification system is used by people other than its developers.

A raster map with a resolution of 5m x 5m was created for the study area, using the ESLUC system. The map was constructed using the best and most detailed geographical data available, covering the entire study area and allowing for mapping of the ESLUC classes. The following data sources were used to create the map: the biological valuation map of Flanders (INBO 2010) providing highly detailed information on vegetation type, the National Geographical Institute topographic vector database (NGI 2007), the areas under nature management map (INBO 2008), the intertidal flats map (INBO 2007), the green map of Flanders (ANB 2012), that distinguishes between high (> 3m) and low (< 3m) vegetation, and the agricultural land use map of Flanders (VLM 2011). Table 2.2 gives an overview of the original number of classes per main land use category occurring in the study area according to the ESLUC map.

The main differences between the ESLUC, the governmental classification system and CORINE are to be found in the high number of natural land use categories in the ESLUC and the distinction between paved surfaces and green infrastructure. In particular, different nature types are distinguished according to habitat type, moisture status, dominant species and vegetation development. Attention is also given to the importance of green infrastructure in cities, with a distinction made between categories such as gardens, parks and cemeteries.

Starting from the ESLUC system raster map, the ESLUC classes were reclassified according to the CORINE and governmental classification system respectively (Appendix A Table A.1). Using the

ESLUC map as a basis retains the spatial heterogeneity typical of the study area in the three maps, as the CORINE and governmental map are only available in coarser resolution (100m x 100m). It also eliminates the effect of spatial resolution (Dendoncker et al. 2008; Anderson et al. 2009; Schulp and Alkemade 2011) caused by using maps with different cell sizes, as well as the effect of spatial discrepancies in land use amongst the maps caused by the use of different mapping techniques. The effects of different classification systems on ecosystem services mapping as analyzed in this study are thus merely a reflection of differences in thematic resolution.

Of the 22 classes occurring in the study area according to the original CORINE map, 3 did not reoccur after converting the ESLUC map to fit the CORINE classification system (Table 2.2). Of the 32 classes found in the study area on the original governmental map, 17 remain after reclassifying the ESLUC according to the governmental classification system (Table 2.3).

Table 2.2 – Overview of the lost classes (column 'Original CORINE map') after reclassification of the ESLUC classes according to the CORINE classification system, including % of the total surface of the study area and final classification into which the lost classes were reclassified (column 'CORINE map after reclassification')

Original CORINE map	% total surface area	CORINE map after reclassification
Industrial or commercial units	3%	Continuous urban fabric
Complex cultivation patterns	18%	Non-irrigated arable land Fruit trees and berry plantations Pastures
Land principally occupied by agriculture, with significant areas of natural vegetation	10%	Non-irrigated arable land Fruit trees and berry plantations Pastures Broad-leaved forest Coniferous forest Mixed forest Moors and heathland Natural grassland Transitional woodland-shrub Inland marshes

Table 2.3 – Overview of the lost classes (column 'Original governmental map') after reclassification of the ESLUC classes according to the governmental classification system, including the % of the total surface of the study area and final class into which the lost classes were reclassified (column 'Governmental map after reclassification')

Original governmental map	% total surface area	Governmental map after reclassification
Light industry	< 1%	Business and industrial site
Heavy industry	< 1%	Business and industrial site
Waste water treatment	< 1%	Business and industrial site
Mining	< 0.1%	Business and industrial site
Energy	< 0.1%	Business and industrial site
Wholesale and transport	< 1%	Business and industrial site
Other industrial/commercial	5%	Business and industrial site
Retail and catering	< 1%	Residential and commercial fabric
Health and education	< 1%	Residential and commercial fabric
Military infrastructure	< 1%	Residential and commercial fabric
Non-registered agricultural land	7%	Agricultural grassland
Cropland with nature targets	< 0.1%	Cropland with environmental targets
Marshland (no biodiversity management)	3%	Marshland (biodiversity management)
Heathland (no biodiversity management)	< 0.1%	Heathland (biodiversity management)
Non-identified land use	4%	all

2.3.3. Qualitative assessment of ecosystem services

For each category in every classification system, a score was given to express the capacity of the land use to provide ecosystem services. The definition of the scores is based on the system used by Burkhard et al. (2009) in which a score of 0 means the land use type has no relevant capacity to provide the particular ecosystem services, 1 = low relevant capacity, 2 = relevant capacity, 3 = medium relevant capacity, 4 = high relevant capacity and 5 = very high relevant capacity.

The scoring of the ESLUC classes (Appendix A Table A.2) is based on the methodology used by Burkhard et al. (2009) who used expert opinions to estimate the capacity of the land use types to provide services. Scoring was carried out during a round of expert discussions involving representatives from different ecological research and management fields (10 experts in total). To avoid the risk of bias, none of the experts that developed the ESLUC were involved in the scoring experiment. In order to prevent the results from being influenced by differences in expert opinion, it was decided that the CORINE and governmental system classes' (Appendix A Table A.3 and A.4 resp.) scores would be awarded based on the score matrix for the ESLUC classes, instead of using the scores provided in Burkhard et al. (2009). The same general conclusions that Burkhard et al. (2009) came to can also be drawn from the score matrices developed in this study. That is: forest and semi-natural land use types have an overall high capacity to deliver a broad range of services and urbanized land use classes have a very low capacity for service-delivery. Agricultural classes in the different classification systems have a low capacity to provide biochemical regulating services (global climate regulation, nutrient retention and denitrification) while they perform well for provisioning services for crops and livestock.

2.3.4. Quantitative assessment of ecosystem services

The quantitative assessment was carried out for eight different services: 3 provisioning services (crop production, livestock production, wood production) and 5 regulating services (global climate regulation through soil organic carbon sequestration, groundwater recharge through infiltration, nutrient regulation through denitrification, nutrient regulation through nutrient soil retention and pollination). For each service, a quantitative map, that was developed for ecosystem services assessment in different governmental and research projects in Flanders (Meersmans et al. 2011; Liekens et al. 2013; Staes et al. 2014; Van der Biest et al. 2014; European Interreg IVB project GIFT-T!, GIFT-T! 2014; Nature Outlook Flanders 2030, INBO 2014) was used. Table 2.4 gives an overview of the type of data that was used to develop the models, with a distinction between (1) models based on measurement data from within the study area, (2) models that partly rely on measurements from within the study area and partly on causal relationships from existing literature, either based on field data from other study areas or on logical assumptions and (3) models that are entirely based on literature. Pollination is exceptional in comparison with the other services as it is only based on one input parameter: that is, vegetation type. All the models were developed in ArcGIS 10.2 (ESRI 2013) and produce maps on a 5m x 5m or 25m x 25m resolution. For more background information on the variables and processes driving these models and the quantification of ecosystem services supply please see Appendix B.

Table 2.4 – Types of data used for the quantitative modeling of ecosystem services

Based on field data from within the study area

		Spatially explicit input data	
Ecosystem service	**Source**	**a) Land use**	**b) Geophysical environment**
Crop production	Bollen 2012; Moens et al. 2008; Brouwer en Huininck 2002	Agricultural land use map Flanders (VLM 2011) ESLUC	Soil texture (AGIV 2001) Soil wetness class (AGIV 2001) Profile development (AGIV 2001) Substrate (AGIV 2001) Topography - Digital Elevation Model Flanders (AGIV 2011)
Livestock	Bollen 2012; Moens et al. 2008; Brouwer en Huininck 2002	Agricultural land use map Flanders (VLM 2011) ESLUC	Soil texture (AGIV 2001) Soil wetness class (AGIV 2001) Profile development (AGIV 2001) Substrate (AGIV 2001) Topography - Digital Elevation Model Flanders (AGIV 2011)
Climate regulation	Meersmans et al. 2011; Altor and Mitsch 2008	ESLUC	Soil texture (AGIV 2001) Soil wetness class (AGIV 2001) Topography - Digital Elevation Model Flanders (AGIV 2011)

Based on field data from within the study area and literature on causal relationships between ecosystem variables

		Spatially explicit input data	
Ecosystem service	**Source**	**a) Land use**	**b) Geophysical environment**
Wood production	Moonen et al. 2011; De Vos 2009; Jansen et al. 1996	ESLUC	Soil texture (AGIV 2001) Soil wetness class (AGIV 2001) Topography - Digital Elevation Model Flanders (AGIV 2011)
Groundwater recharge	Batelaan and Desmedt 2007	ESLUC	Soil texture (AGIV 2001) Soil wetness class (AGIV 2001) Topography - Digital Elevation Model Flanders (AGIV 2011)
Nutrient retention	Liekens et al. 2013; Meersmans et al. 2011	Agricultural land use map Flanders (VLM 2011) Biological valuation map Flanders (INBO 2010) ESLUC	Manure standards for Flanders (VLM 2012) Soil texture (AGIV 2001) Soil wetness class (AGIV 2001) Topography - Digital Elevation Model Flanders (AGIV 2011)

Based on literature on causal relationships between ecosystem variables or expert judgment

		Spatially explicit input data	
Ecosystem service	**Source**	**a) Land use**	**b) Geophysical environment**
Pollination	Staes et al. 2014	Biological valuation map Flanders (INBO 2010)	-
Denitrification	Pinay et al. 2007	-	Soil texture (AGIV 2001) Soil wetness class (AGIV 2001) Topography - Digital Elevation Model Flanders (AGIV 2011)

2.3.5. Analytical settings

The qualitative maps are based on an expert score matrix and the quantitative maps are based on spatially explicit process-based models, resulting in 4 different raster maps per ecosystem service (Figure 2.2): one with the quantitative estimate for the service and one with the qualitative score for the service for each of the three land use maps. Each of these 3 qualitative maps was correlated individually with the corresponding quantitative map. The strength of the correlation gives information on the performance of land use as an indicator for service delivery.

Due to the large extent (~ 1,100km²) and the small cell size (5m x 5m) of the raster datasets, the statistical analysis was carried out for a systematic subsample of the raster data sets rather than the entire data set. For each quadrant of a regular grid of 100m x 100m, only the central cell (5m x 5m) was used. This resulted in a subsample of 108,712 cells for the entire study area. The information attached to the qualitative and quantitative maps for the subsample was abstracted using the Combine tool in the ArcGIS 10.2 (ESRI 2013) Spatial Analyst toolset.

The statistical analysis was carried out using R version 3.1.0 (RStudio Inc. 2012). Each of the 8 qualitative score maps (one for each ecosystem service studied) per land use classification system was correlated with its corresponding quantitative version (Figure 2.2). Normality was checked for using the Shapiro-Wilk test (Shapiro and Wilk 1965). As not all data was normally distributed, we applied a non-parametric Spearman rank correlation.

Violin plots were created to get an insight into the nature of potential discrepancies between score-based and model-based ecosystem services assessment that may not come out of the correlation analysis. A violin-plot is a combination of a boxplot and a kernel density plot where the width of the box corresponds to the number of observations of the quantitative value (y-axis) at each score value (x-axis). Linear regression lines were drawn of the quantitative value versus the score value. The violin plots were created using the lattice package in R.

Boxplots were created to analyze visually whether or not the capacity of land use to predict ecosystem services delivery differs between provisioning and regulating services. The x-axis represents the sum of the scores of all the provisioning services (agricultural production score + livestock production score + wood production score) and of all the regulating services (climate regulation score + denitrification score + infiltration score + pollination score + nutrient retention score) respectively. The y-axis represents the sum of the standardized quantitative values of all the provisioning services and of all the regulating services respectively. Standardizing was applied to rescale quantitative values of different services onto a shared scale and to allow the calculation to be performed. Additionally, linear regression lines were drawn on the graphs.

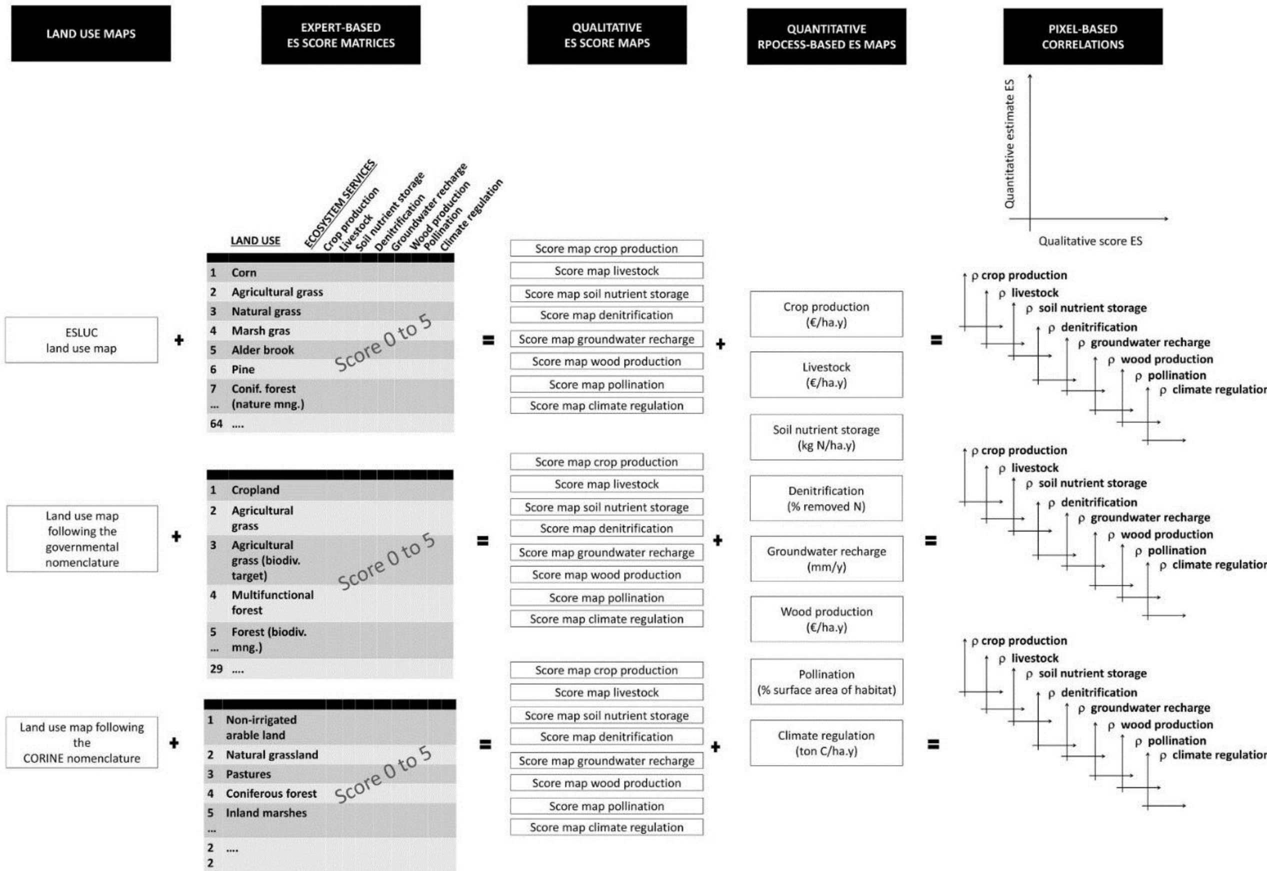

Figure 2.2 – Analytical settings. Each land use map was connected with a score table (score of 0 to 5 for the capacity of each land use type to deliver ecosystem services). For every ecosystem service, a map was made, giving the qualitative score for service delivery, resulting in a set of 24 qualitative maps (8 services x 3 land use maps). Each qualitative service delivery map was then correlated with its corresponding quantitative process-based map (one for each service), resulting in 24 correlation coefficients (8 services x 3 land use maps).

2.4. Results

In the three systems, the correlation coefficients between qualitative score and quantitative estimate for the provisioning services are considerably higher than the correlation coefficients for the regulating services. Correlations for the regulating services are generally very weak or even illogically negative. Illogical negative correlations are obtained for three of the five regulating services in CORINE and for two of the five regulating services in the governmental system. The ESLUC does not show any negative correlation.

The ESLUC generally has higher correlation coefficients than CORINE and the governmental system (Table 2.5), except with denitrification and livestock which score better using CORINE. The governmental system has the lowest correlation coefficients for all services.

The differences between the correlation coefficients over the different land use classification systems are most pronounced for the regulating services (infiltration, nutrient retention, pollination, climate regulation). This difference in predictive capacity of land-use based scoring for regulating services compared with provisioning services is also illustrated in Figure 2.3. The provisioning services show a clear positive correlation between qualitative score and quantitative estimate in all 3 of the classification systems. For the regulating services, the positive correlations are more pronounced in the ESLUC than in CORINE and in the governmental system, indicating that regulating services especially benefit from a refined land use classification system.

Although the correlation coefficients for the provisioning services are relatively high in all three classification systems (Table 2.5), the violin graphs in Figure 2.4 show the significant inconsistencies that may occur between quantitative value and qualitative score. For wood production in the ESLUC and CORINE, for example, the risk of making false predictions is high, in terms of likelihood or frequency of misclassifications (widening of the violin graphs near value 0 at each score value different from 0), as well as in terms of the magnitude of the error (widening of the violin graphs near value 0 at the highest score values).

Table 2.5 – Spearman's rank correlation coefficients between qualitative score and quantitative estimate per ecosystem service per land use classification system, with correction for spatial autocorrelation (n = 108,712). Highest correlation per ecosystem service in grey.

	Spearman rank's ρ correlation coefficient		
Ecosystem service	**CORINE**	**Governmental system**	**ESLUC**
wood production	0,78	0,69	0,83
crop production	0,70	0,68	0,73
livestock production	0,83	0,68	0,81
denitrification	0,17	0,10	0,09
infiltration	-0,04	-0,03	0,12
nutrient retention	-0,18	-0,02	0,28
pollination	-0,11	0,02	0,22
climate regulation	0,29	0,28	0,38

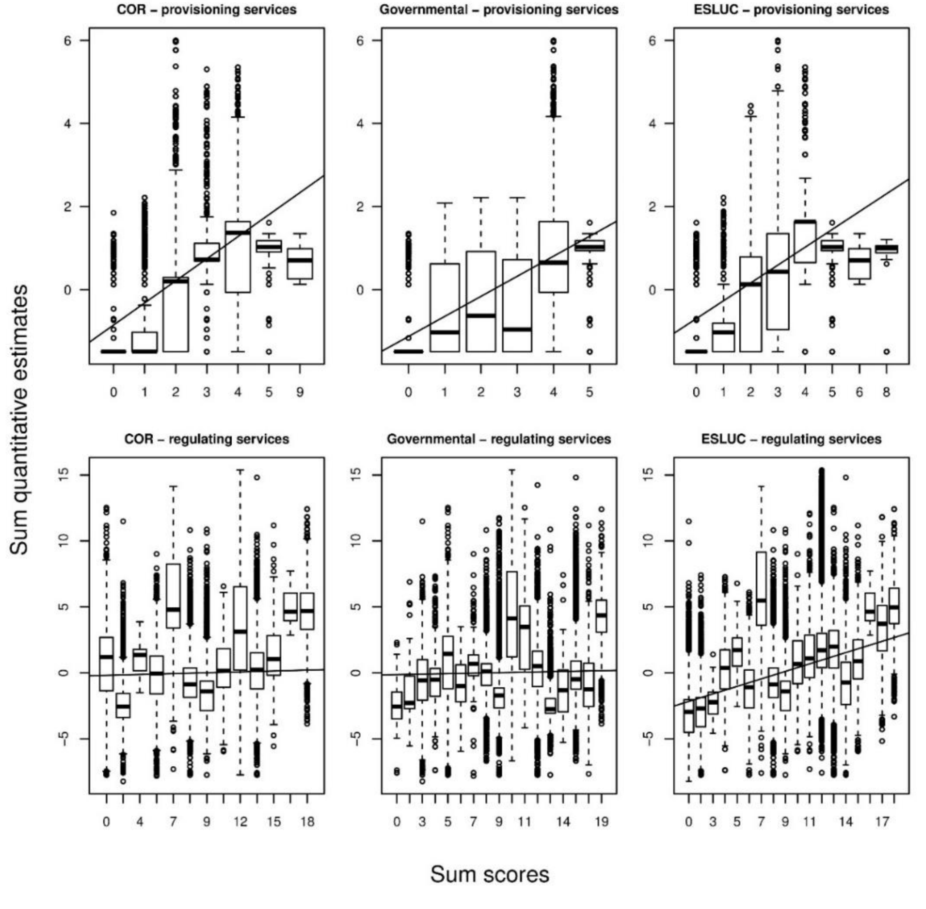

Figure 2.3 – Boxplots and regression lines illustrating the relation of land-use based scoring versus quantitative modeling of provisioning (crop production, livestock production, wood production) and regulating (denitrification, infiltration, nutrient retention, pollination, climate regulation) ecosystem services using a coarse land use classification system (CORINE and governmental classification system) vs. a refined system intended for ecosystem services assessments (ESLUC). The X-axis represents the sum of the qualitative scores for the different services, the Y-axis represents the sum of the standardized (mean = 0; standard deviation = 1) quantitative values for the different services.

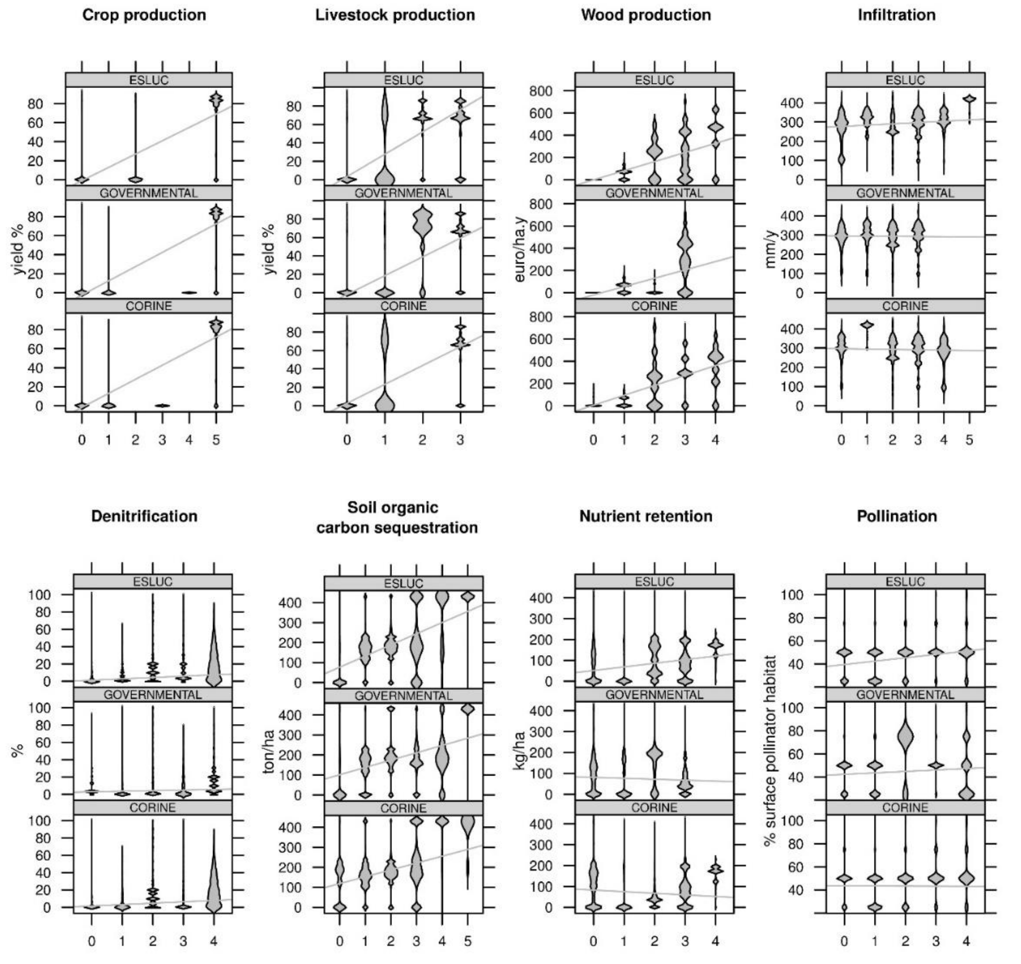

Figure 2.4 – Violin plots and linear regression lines for the relationship between qualitative score (x) and quantitative estimate (y) per ecosystem service, per classification system. The width of the "violins" represents the probability density of the quantitative data at each score value.

2.5. Discussion

2.5.1. The impact of land use reclassification

Reclassifying the CORINE and governmental land use classes according to the corresponding categories of the ESLUC led to a reduction in the number of classes by 3 and 15 respectively. This, however, is expected not to influence the results significantly for several reasons.

The lost categories are mostly artificial surfaces. As service delivery for these areas scored 0 in all classification systems, this did not cause any information loss.

The "complex cultivation patterns" class in CORINE, which is defined as a 'juxtaposition of small parcels of diverse annual crops, pasture and/or permanent crops' (EEA 1994), was reclassified into a more specific type of agricultural cultivation. The class of "land principally occupied by agriculture, with significant areas of natural vegetation", which EEA (1994) defines as 'areas principally occupied by agriculture, interspersed with significant natural areas' was reclassified into a more specific agricultural or (semi-)natural land use type. These classes disappeared because the cell size of the ESLUC is smaller than the minimum size of a parcel. This information gain may slightly increase correlation coefficients obtained using the CORINE classification system, which could potentially disprove the hypothesis that a more detailed classification system leads to more accurate ecosystem services assessments. The same applies to the "non-identified land use" class in the governmental classification system which was reclassified as any other type of land use.

"Non-registered agricultural land" from the governmental map was reclassified as "agricultural grassland". This may lead to a minor loss of information as service provision may slightly differ amongst both categories. Finally, some very rare land use classes from the governmental classification system ("cropland with nature targets", "marshland with biodiversity management", "heathland with biodiversity management") were reclassified into more general classes. The total surface area of these classes covers less than 1%.

2.5.2. The effect of a more detailed land use classification system on the prediction of ecosystem services

Using coarse spatial resolution data engenders a significant risk of error in ecosystem services assessments. The true location of the more extreme values is masked by using average values on larger spatial scales (Eigenbrod et al. 2010). This will especially lead to erroneous estimates in highly fragmentized landscapes such as those in Western Europe. A similar effect is seemingly created when using a coarse thematic resolution (Kienast et al. 2009; Vihervaraa et al. 2010; Koschke et al. 2012; Hou et al. 2013): values in general land use classes will be smoothed out, masking the more extreme values that can be obtained using more detailed land use classes.

Our results indicate that, when applied to regulating ecosystem services, land-use based assessments become slightly more accurate when a more detailed land use classification system is used. The main factor explaining the better correlations is the greater level of nuance in environmental conditions amongst the ESLUC land use classes. Regulating services often strongly depend on these environmental characteristics. While they can be delivered through almost every type of land use, it is the combination of land use/vegetation with the abiotic state (hydrology, texture, nutrient supply, etc.) that determines to which extent they will be delivered. While CORINE and the governmental system, for example, identify respectively 1 and 2 classes in wet terrestrial areas (wetlands), the ESLUC has 8 different categories. Natural grasslands in CORINE include very wet as well as dry grasslands, while they are categorized into two separate classes in the ESLUC. The same categorization according to moisture status is also applied in other classes, thus leading to a better approximation of soil hydrological conditions as an input parameter in the biophysical models and thus to higher correlation coefficients for the ESLUC. Nevertheless, correlations for regulating ecosystem services remain weak, even when a detailed classification system is used. Taking abiotic conditions into account increases the accuracy of land-use based ecosystem services assessment, but it is undesirable to include all types of environmental conditions in the delineation of land use. This would lead to an explosion in the number of classes and produce an unmanageable classification system.

The relatively high correlation coefficients for the provisioning services in all classification systems seem to indicate that the accuracy of land-use based assessments of provisioning services does not increase with thematic resolution. This can be explained by provisioning services' strong dependence on land use. Whereas food from crops can only be delivered in areas covered with cropland, for example, carbon is sequestered under nearly all types of land use. Within the upper score categories for crop production, differences in service delivery are explained by other factors than land use, such as abiotic conditions or management. These differences, however, may be too small to be distinguishable using a rather rough score system (0-5).

Our results suggest that land use may be used as a coarse, nearly binary predictor of service delivery, in the sense that the service may or may not be delivered. This is especially the case for provisioning services that strongly depend on land use. In this sense, a more detailed land use classification system does not necessarily provide additional accuracy in land-use based ecosystem services assessments.

2.5.3. The accuracy of land use as a proxy for ecosystem service delivery

Although a more refined system increases the accuracy of proxy-based assessment of regulating ecosystem services and correlation coefficients seem to show that land use is a good proxy for provisioning services, the violin graphs (Figure 2.4) illustrate that the method is prone to significant errors. The ESLUC infiltration diagram, for example, shows that areas with a high total groundwater recharge (around 300mm/y) are roughly as frequently classified into very low capacity for infiltration (score 1) as they are into high capacity for infiltration (score 4). Some services (e.g. wood production) show a relatively high correlation between proxy-based and model-based ecosystem services assessment, but from Figure 2.4 it becomes clear that a relatively high number of pixels are classified erroneously (0 wood delivery has a high frequency in the majority of the score categories).

The land-use based score method inherently assumes that land use is in accordance with the abiotic environment, as it does not distinguish between different environmental properties. In very densely populated areas with limited available space, discrepancies may exist between land use and the ecosystem's biophysical suitability (Kienast et al. 2009). Within the study area, for example, increasing population density and growing demand for food over the past centuries has caused agriculture to expand to less suitable grounds such as wet valley soil. Figure 2.5 illustrates potential errors that arise from using land use as a proxy to map agricultural production. While according to Figure 2.5a, the capacity of the land along the river banks to deliver agricultural production is very high, the biophysical model (Figure 2.5 b) predicts yield losses of up to more than 50% (dark orange) in the wettest soils (Figure 2.5 c). Protection of nature and ecosystem services especially is very important in densely populated areas with large demands for ecosystem services. When assessing the state of ecosystem services and how they evolve over time, such as in nature monitoring programs, not taking into account these areas with inappropriate land use choices may lead to adverse management decisions.

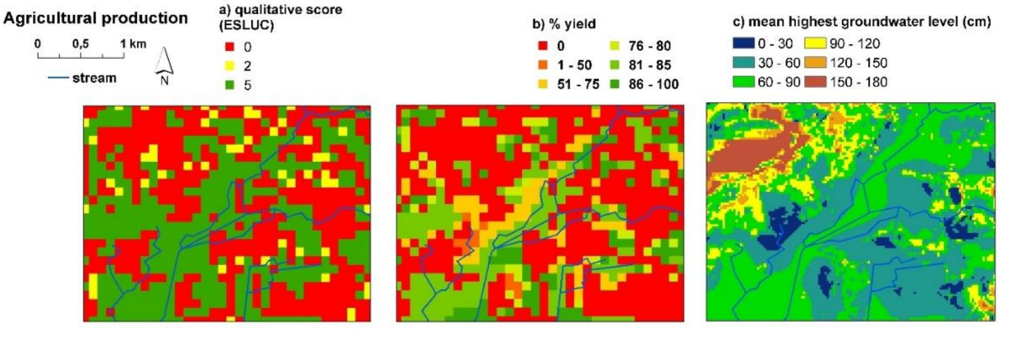

Figure 2.5 – Illustration of the discrepancies between qualitative scores using the ESLUC system (a) and quantitative estimate (b) of agricultural production (% of yield), and comparison with mean highest groundwater level (c).

2.5.4. Alternatives for land-use based ecosystem services assessment

Recent literature has been largely critical of the use of land use as a proxy for ecosystem services delivery (Naidoo et al. 2008; Eigenbrod et al. 2010; Seppelt et al. 2011; Geijzendorffer and Roche 2013). The results of the analyses carried out in this study further underpin the poor quality of land use as a proxy for ecosystem services, and in particular for regulating services. Policy makers, however, need tools that allow for periodic assessment of ecosystem services in an efficient and pragmatic way. One option would be to make use of composite rather than simple indicators, allowing abiotic, socio-economic or spatial elements besides land use to be taken into account. Another option is to make use of one of the many user-ready tools that are available for assessing ecosystem services (e.g. InVEST – Kareiva et al. 2011; ARIES – Villa et al. 2014; etc.). Among the wide variety of tools that exist (Bagstad et al. 2013), some have been specifically developed to contend with the obstacles that favor the usage of land use as a proxy for service delivery, such as data availability and ecological complexity. Bayesian belief networks for example allow predictions to be made when information on input variables is partially missing and are increasingly being applied in ecosystem services research (e.g. Haines-Young 2011; Grêt-Regamey et al. 2013; Landuyt et al. 2013; Van der Biest et al. 2014; Villa et al. 2014). Other tools apply a TIER-based approach, which allows users to choose from different levels of model complexity, depending on the availability of data (e.g. Kareiva et al. 2011).

Although the analysis is carried out at regional level, it is expected that similar conclusions can be drawn on larger spatial scales, such as continental. The CORINE "moors and heathland" class, for example, includes dry heath types such as the Calluno-Ulicetea alliance as well as wet heath types such as the Ericion tetralicis alliance (EEA 1994). Their capacity for delivering services such as groundwater recharge, climate regulation and fire prevention, however, differs greatly. Taking additional indicators such as moisture status into account may thus increase the accuracy of ecosystem services assessments, even when used on large spatial scales.

2.5.5. Uncertainty

An important source of uncertainty in the analysis is caused by using expert opinions to make predictions on ecological functions (Burkhard et al. 2009; McBride et al. 2012). This may partially contribute to the low or negative correlations obtained for some services. As expert-based scoring was used for all of the classification systems and the scores for the governmental map and for CORINE are derived from the ESLUC scores, potential errors in the expert judgment are not expected to influence the results.

Another source of uncertainty in the analysis lies in the biophysical model estimates. Validation of such models is an essential step towards improvement of ecosystem services assessments (Seppelt et al. 2011; Rounsevell et al. 2012; Seppelt et al. 2012). Due to the wide variety of ecosystem services, this is not always possible on the level of the study area for which the assessment needs to be done (Lautenbach et al. 2010). Although research has demonstrated that modelled outputs of ecosystem services assessments do not always match field observations, they are expected to match primary data better than land use proxies, especially if they are based on sampling from within the study area (Eigenbrod et al. 2010). This justifies the use of biophysical models as a basis for comparison in this research.

2.6. Conclusions

The results of the analyses further underpin prior evidence of the poor quality of land use as an indicator for ecosystem services delivery and additionally indicate that the use of a more refined land use classification system only slightly increases the accuracy of land-use based ecosystem services assessments. For provisioning ecosystem services, correlation coefficients between qualitative land-use based and biophysical-model-based assessments are relatively high when using a coarse land use classification system as well as when using a refined system. The risk of making substantial errors in predicting service delivery based on land use, however, is high for both classification types. Correlations are weak for regulating ecosystem services, although a more detailed land use classification system slightly increases correlation coefficients. The results of this experiment highlight the need to go beyond using land use as a simple indicator of ecosystem services delivery in environmental management and to take additional factors such as environmental, socio-economic and spatial components that drive ecosystem services delivery into account.

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3

Bayesian belief networks to model ecosystem services

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3.1. Abstract

Integrating the ecosystem service concept into land use planning requires tools that allow rapid and transparent assessment of ecosystem services. The demand for simple indicators has stimulated the emergence of land use based proxy methods. Although these have been very powerful to create policy awareness on different levels, they are insufficient when it comes to land use and policy planning for ecosystem service delivery. Discarding the complex ecological reality or scientific uncertainty poses serious risks for adverse effects of policies. This explorative study constitutes the basis for the further development of a tool to link land use planning for ecosystem service bundle optimization, capturing inherent ecological complexity and uncertainty. Particular emphasis was placed on the biophysical potential of an ecosystem to deliver services and the link with the actual land use. The EBI – Ecosystem Service Bundle Index – builds on a Bayesian network model that allows integration of biophysical and socio-economic processes as well as land use planning policies driving the delivery of bundles of ecosystem services. The EBI prototype was tested in a pilot study area using three interacting ecosystem services. Incorporation of judicial land use claims, more intense involvement of stakeholders and other improvements are being developed.

3.2. Introduction

Global costs of technical measures to replace natural processes, restore ecological functions and reduce risks of climate and macro-economic shifts increase rapidly. This stresses the eminence of taking into account a wider scope of limited natural resources and the need for an integrative planning of further land and resource use (MEA, 2005; TEEB, 2010; Potschin and Haines-Young, 2006). Ecosystem services, the benefits humans derive from ecosystems, are claimed to allow such integrative resource use planning. A growing body of literature is being published on ecosystem services in widely different fields as ecology, agriculture, economy, sociology, politicology, etc. (Seppelt et al., 2011; Fisher et al., 2009). Meanwhile, the concept gains influence in academic, governmental as well as corporate circles. In lack of transparent and user-friendly techniques to rapidly and periodically assess ecosystem services, simple indicators (mostly land use proxies) are applied to map services. The demand for simple indicators however must not cause a discard of complex biophysical and socio-economic reality, since this entails non-quantified risks for decision-making. The few papers that have put ecosystem service quantification and mapping results to the test (Eigenbrod et al., 2010; Villa et al., 2009) have immediately pointed out serious biases, although these pragmatic mapping approaches are widely applied within an emerging market of mapping and valuation tools. Most ecosystem services originate from complex interactions between abiotic and biotic elements (e.g. Luck et al.,2003; Kremen, 2005; Luck et al., 2009) which are not straightforwardly integrated in land use categories. Proxy-based mapping has been very useful to generate wider awareness by demonstrating the presence and spatial variability of ecosystem services. However, in order to generate adequate maps for planning and decision-making, robust indicators need to be developed which capture the complexity of interactions between and within services and the related biophysical processes and land use effects.

Recent advances in ecosystem service modeling focus on linking ecosystem service delivery, their associated values and trade-offs across services (e.g. Integrated Valuation of Ecosystem Services and Trade-offs InVEST, and Artifical Intelligence for Ecosystem Services ARIES). None of these models however map ecosystem services in terms of their supporting systems, namely the biophysical potential for the delivery of services. Insight in biophysical supporting functions for ecosystem services is essential for optimizing land use and maximizing the extent to which the potential is realized. Bastian et al. (2012) make a clear distinction between potential supply of ecosystem services (based on the natural capacity of an ecosystem), and the actual delivery. This chapter incorporates these insights into a model estimating provision of three interacting ecosystem services, couples this model with spatially explicit data from a river basin in the north of Belgium and captures underlying trade-offs in a single index for delivery of the ecosystem service bundle. The presented approach offers a way to capture biophysical complexity in a pragmatic but scientifically transparent index applicable beyond awareness raising. Further extensions to incorporate additional services, socio-economic valuation and consensus methods are underway.

3.3. Materials and methodology

3.3.1. Pilot study area

The study area (261 km2) consists of the upstream area of the Grote Nete basin (Belgium), Figure 3.1. It is a typical lowland landscape with numerous brooks and small rivers, infiltration areas and dune relicts. The soil type varies from sand with sandy loam and loamy sand in the floodplains to loamy and clayey soils in the southern-most part. Land use mainly consists of agriculture (22% pasture and15% cropland), paved area (28%), forest (17%) and wetland (4%).

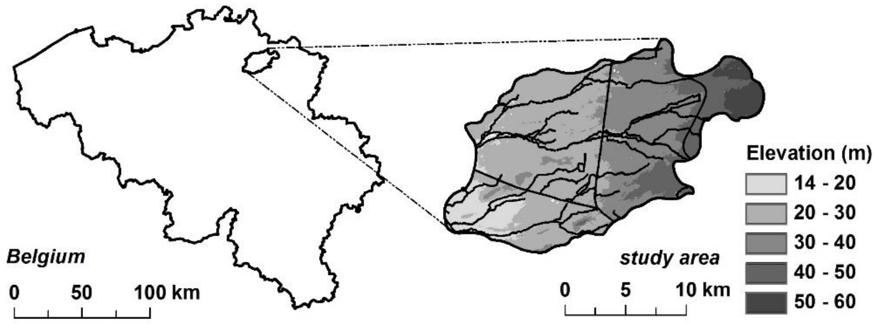

Figure 3.1 – Location, topography and major hydrology of the study area.

3.3.2. Selected ecosystem services

Three interacting ecosystem services which exhibit trade-offs and competitive land claims in the catchment were selected: provisioning services food production, wood production and regulating service climate regulation as long-term storage of soil organic carbon. Forests, extensively managed grasslands and wetlands, for example, are capable of storing carbon, but wetlands are incapable of providing substantial amounts of food or wood. On the other hand, intensively managed croplands and forests exhibit sub optimal capability to store large amounts of carbon. The area exhibits abroad range of hydrological and biophysical conditions which affect the potential for delivering these three services separately: well-drained loamy soils are very suitable for food production, but less for climate regulation, while wetter zones are more suitable for climate regulation but still partly capable of providing food (pasture) or wood. The complexity of varying feasibilities and land use related trade-offs between these three services provides an ideal pilot test-case.

3.3.3. Modelling and mapping approach

A Bayesian belief network modelling approach was applied. Bayesian belief techniques recently gained a lot of attention in ecosystem service modelling due to several model related advantages, e.g. the possibility to take into account uncertainties and to complement empirical data with expert knowledge (Aguilera et al.,2011; Landuyt et al., 2013). The limited availability of data, high complexity of natural processes and related uncertainties favour the use of Bayesian networks for the assessment of ecosystem services.

A Bayesian belief network is a multivariate statistical model comprising two structural components: a causal network (the directed acyclic graph) as the qualitative part, and conditional probability tables (CPTs) as the quantitative part. In the graph (Figure 3.2), statistical dependencies are indicated by arrows that connect parent nodes, e.g. node A and B, with the child nodes they affect, e.g. node D. Each node has a set of states to which its realized value can belong. The probability that the realized value of a variable X is manifested in a certain state depends on the state probability distributions of its parent nodes and is determined by the conditional probability P(X|parents(X)). These probabilities thus denote the strength of the relations among the graph's variables and are tabled in the model as CPTs, the quantitative component of the model. Input nodes are parentless and therefore do not contain CPTs. When the conditional probabilities of all nodes are known (prior knowledge), posterior probability distributions for all nodes can be determined given inserted evidence for some nodes (instantiated variables). For a detailed model description and statistical background we refer to Jensen (2001).

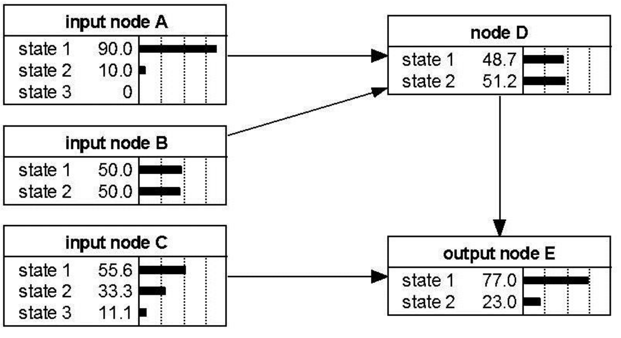

Figure 3.2 - A visual example of a Bayesian belief network. Causal relations among the systems variables can be derived from the graph. The probability distribution of child node D is for example affected by the probability distributions of both its parent nodes A and B.

A major strength of Bayesian modelling is the possibility to make predictions when information on the state of input variables is partially missing, whereas unavailable data in numerical models results in ineffective or inaccurate reasoning (Charles River Analytics Inc., 2008). Uncertainties resulting from missing information on the state of an input variable is dealt with following the principle of indifference and is automatically propagated through the model, increasing uncertainty to the models' predictions (Jensen, 2001; Marcot et al., 2006). A second type of uncertainty occurs in the causal relationships between parent and child nodes stored in the CPTs. These relationships can either be deterministic, so without uncertainty, or probabilistic. As the EBI is developed as a prototype, all relationships were kept deterministic, assigning a 100% probability to the each causal relationships (e.g. Appendix C Table C.4). These different uncertainty measures allow for determination of the level of confidence of the final EBI, which is essential for honest decision support.

3.4. Calculation

3.4.1. Influence diagram

Model development was based on Smith et al. (2007). Firstly, an influence diagram was constructed (Figure 3.3). This causal network combines GIS data-layers of biophysical characteristics (V1–V3) and land use (LU) as inputs, to calculate the delivery of ecosystem services as an integrated ecosystem service bundle index (EBI). Existing feasibility models (Provincie Antwerpen, 1998; De Vos, 2000; Meersmans et al., 2008) were used to calculate ecosystem service potential indicators (P1–P3) based on the biophysical variables soil type and hydrological characteristics. Combining these service potential indicators (P1–P3) with land use (LU) allowed to calculate actual ecosystem service provision indicators (ES1–ES3) based on expert judgment (Appendix C). Secondly, the developed causal network was converted into a Bayesian network in Netica (Norsys Software Corporation, 1998). This model was then spatially applied in the pilot study area on a 100 m pixel basis by using the attribute values of each pixel (for the variables V1–V3 and LU) as evidence for the input nodes of the model. For each pixel, the ecosystem service potential and provision indicators (P1–P3, ES1–ES3) were calculated through Bayesian inference. Consequently, suboptimal service provision can be attributed either to suboptimal biophysical potential or to suboptimal land use. Next, all separate service provision indicators (ES1–ES3) were combined into a single ecosystem service bundle index (EBI) to allow an integrated evaluation of the currently delivered bundle of ecosystem services. To enable mapping of model outputs, the weighted average value of each target variable was used. Mapping of the model output enabled fine-tuning and visual validation of the operational model. Additionally, model results were validated on the field in the upstream part of the pilot area involving local experts in landscape planning.

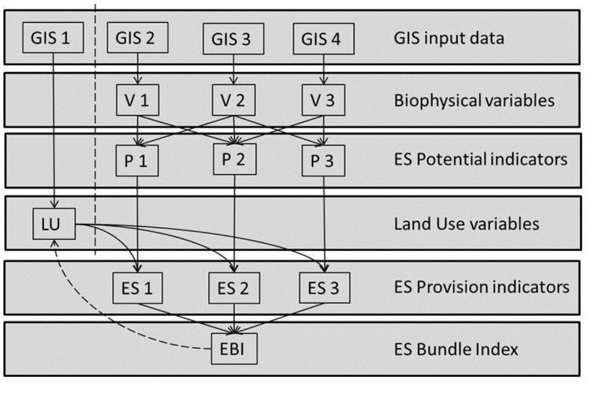

Figure 3.3 – Influence diagram for the provision of multiple ecosystem services. The full lines indicate how the actual ecosystem service bundle index (EBI) is determined based on spatial data of the input variables. The dashed arrow illustrates how the optimal land uses are selected based on the maximal attainable EBI value under certain biophysical conditions.

3.4.2. Operational model

For converting the conceptual model into an operational BBN, network variables and their states were defined and CPTs of non-input variables were populated with knowledge rules (Appendix C). The model was divided into three submodels which were operationalized with data from currently used governmental classification systems (Provincie Antwerpen, 1998; De Vos, 2000) and empirical model results (Meersmans et al., 2008).

Four input variables run the service provision model: soil texture, hydrology, soil profile development and land use. Each of these variables was available in a GIS layer with a pixel resolution of 100 m. Data on soil texture and profile development was obtained from the Belgian national soil classification system (AGIV, 2001). Hydrology data was generated by updating the drainage classification system from the soil map based on the most recent digital elevation model with 25 m resolution (MVG, 2011). Land use was obtained from the classification system as proposed by Van Esch et al. (2011). For each of the modelled services, the 37 original land use classes were reclassified in a limited set of uniform management classes. A list of all variables used in the model and their states can be found in Appendix D.

Figure 3.4 displays the operational BBN to calculate the EBI. Each model variable is represented by a network node that lists all possible states of the variable. The probability that the realized value of a variable belongs to a particular state is indicated both by a belief bar and by a percentage value (P($X = x_i$)). For the quantitative variables, whose states refer to real values ($V(x_i)$), a mean expected value ($\mu = \sum(P(x_i)V(x_i))/100$) and a standard deviation ($s = \sum(V(x_i) - \mu)^2 P(x_i)/100$) is given.

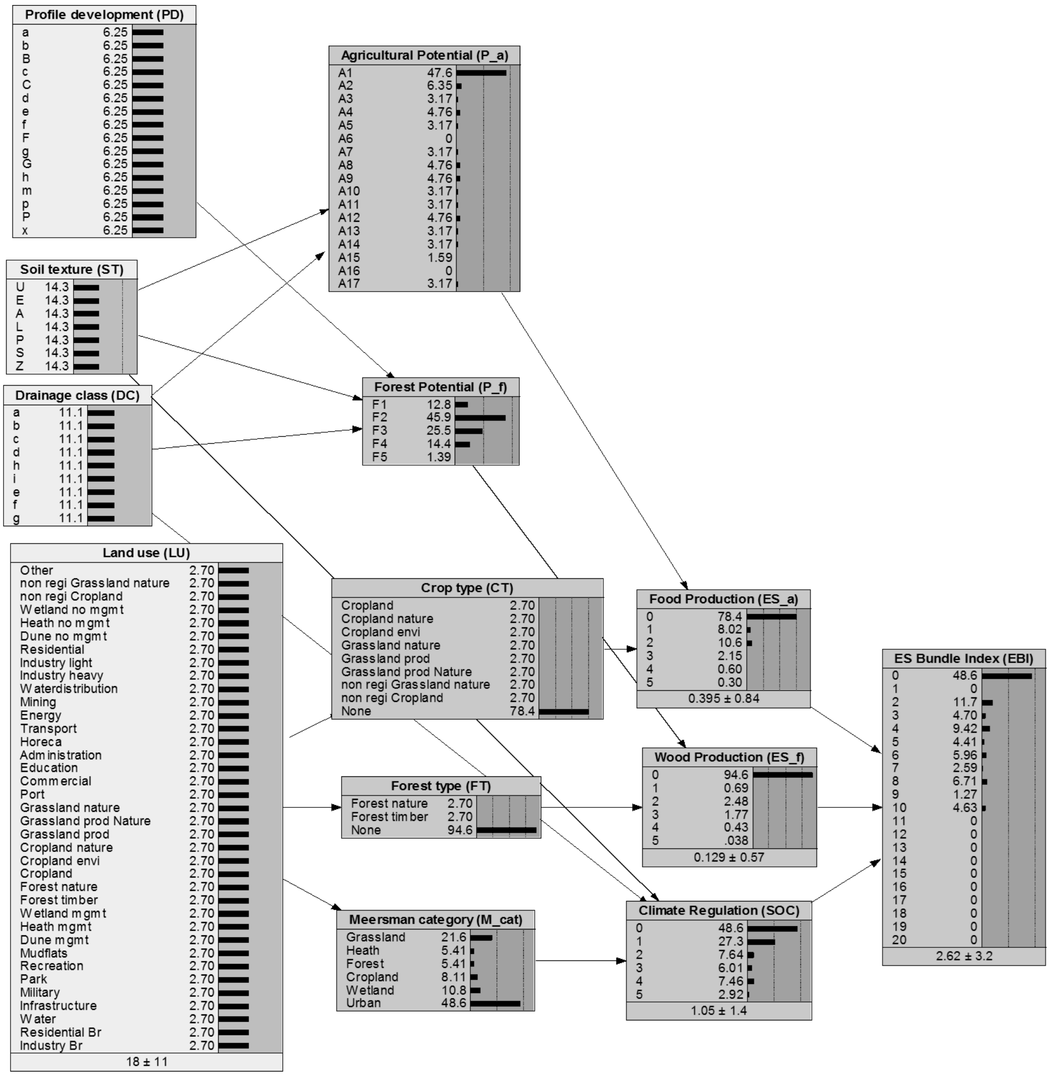

Figure 3.4 - Bayesian network for the provision of three ecosystem services and the EBI. Food production is noted here as "ES_a"; wood production as "ES_f", and climate regulation as "SOC". The numbers and bars refer to probabilities, the numbers in the footer of the nodes refer to mean expected values and standard deviations.

Biophysical agricultural production potential was defined by the soil's texture and hydrological characteristics. To every combination of states of these two variables a level of production potential was assigned based on a governmental feasibility system (Provincie Antwerpen, 1998) ranging from very low (A1) to very high (A17). Similarly, potential wood production was scored from very low to very high (F1–F5) for state combinations of soil texture, profile development and hydrology based on research from De Vos (2000). As only one level of production potential was assigned to every combination of parent states, these relationships are deterministic. Subsequently, actual food and wood production levels were derived from both the potential production levels and the relevant land use classes. These CPTs were populated based on expert judgment and scaled from 0 (no production) to 5 (very high production) (Appendix C).

The submodel that describes climate regulation through soil organic carbon storage is derived from an empirical study conducted by Meersmans et al. (2008). It does not strictly follow the model structure proposed in Figure 3.3 as carbon storage potential and land use were both entangled in Meersmans' analysis. The Meersmans regression formula (Appendix C) quantifies soil carbon storage based on soil texture, soil drainage class and four land use classes. The data were complemented with data from Post et al. (1982) and Adhikari et al. (2009) for wetlands which were not considered as a separate class in Meersmans' research. We also added a sixth land use class 'urban' to this submodel for categorizing soil sealed pixels. As the model is a test version, no distinction was made between paved and non-paved urban surfaces, thus ignoring potential carbon sequestration in parks and gardens. Soil carbon storage in urban land use pixels was considered zero as no additional carbon will be stored under paved area. Soil carbon storage was divided into 6 classes from 0 (no storage) to 5 (very high storage). In first instance, all relationships in the Bayesian network were kept deterministic.

For each pixel, an ecosystem service bundle index EBI was calculated as a weighted sum of the ecosystem service provision indicator scores for each individual service (ES_F for food production, ES_W for wood production and ES_C for climate regulation):

$$EBI = F_1 * ES_F + F_2 * ES_W + F_3 * ES\_C \tag{1}$$

The factors F_x in the equation allow for weighting the relative importance of the three services in the EBI. In this theoretical pilotcase, weighting is driven by a purely hypothetical management strategy aiming for a proportional distribution of the delivered services within the study area, without considering societal demand factors or stakeholder involvement. Therefore, the weighting factor of carbon storage needed to be doubled (F_1= F_2= 1; F_3= 2), increasing the chance that the model predicts a land use type that maximizes carbon storage as the most beneficial land use type for that pixel and thus increasing the extension of land use for carbon storage up to a total surface area more or less equal to the surface area of land use for food production and for wood production. However, the demand for ecosystem services varies locally (and temporally) and so the composition of the bundle of services is a societal decision too. Stakeholder participation is thus strongly recommended in order to make it applicable in real-life contexts.

3.4.3. Model applications

The operational model was first applied to analyze trade-offs in provision of the three services. Therefore, a case file, listing all possible state combinations of the four input variables (37,296 cases), was simulated in Netica. Processing the case file and generating for each case the mean expected value of the three ecosystem service provision indicators (food production, wood production and climate regulation) resulted in an output file containing all co-occurring levels of provision of the three services. Selecting only the combinations actually occurring in the study area allowed visualization of trade-offs between the three considered ecosystem services.

Secondly, for all state combinations of the three input variables for biophysical conditions the maximum achievable level of service provision was deducted together with a set of optimal land uses, yielding 1008 state combinations of the four variables. Based on the GIS input data of biophysical conditions as well as actual land use, the actual ecosystem service bundle index was calculated for each pixel. Also the potential ecosystem service bundle index was calculated for each pixel based solely on the biophysical data and assuming an optimal land use class (Figure 3.3). The difference between actual and optimal EBI can then be calculated. Mapping the difference in EBI for the whole study site highlighted areas with a strong mismatch between actual and potential service bundle provision and thus opportunities for service delivery improvement. Optimal land use options were derived from the earlier developed optimal land use set.

3.5. Results

3.5.1. Trade-offs in ecosystem service delivery

We discern three different levels of trade-offs between ecosystem services. First level trade-offs are generated by the biophysical potential of the ecosystem to deliver the different services. Knowledge about first-level trade-offs is essential to avoid unrealistic optimization scenarios (e.g. wetland on sandy soils, agricultural land in permanently wet areas, etc.). Second-level trade-offs refer to the actual delivery within the study area, capturing biophysical potential trade-offs as well as land use based trade-offs. Third-level trade-offs, which concern the final provision to society, depend on demand, accessibility, ecosystem service flow and generation of benefits. At this stage, the model allows to calculate first and second level trade-offs based on spatially explicit data from the study area.

First-level trade-offs: biophysical potential

The trade-off between provisioning services food and wood production is described here as an example of a first-level trade-off (Figure 3.5). As food and wood production potential benefit from similar similar soil and hydrological processes, they are strongly correlated and demonstrate a first-level trade-off in biophysical suitability. Land use choices for either forestry or agriculture are on a local scale governed by societal processes such as judicial status (e.g. nature reserves), demand (e.g. horse-keeping vs. grassland demand) and traditions rather than biophysical (un)suitability for forestry or agriculture. However, when generalizing this method over larger areas and over more ecosystem services, a larger variety in biophysical characteristics will make trade-offs in biophysical potential a more decisive factor in land use choice.

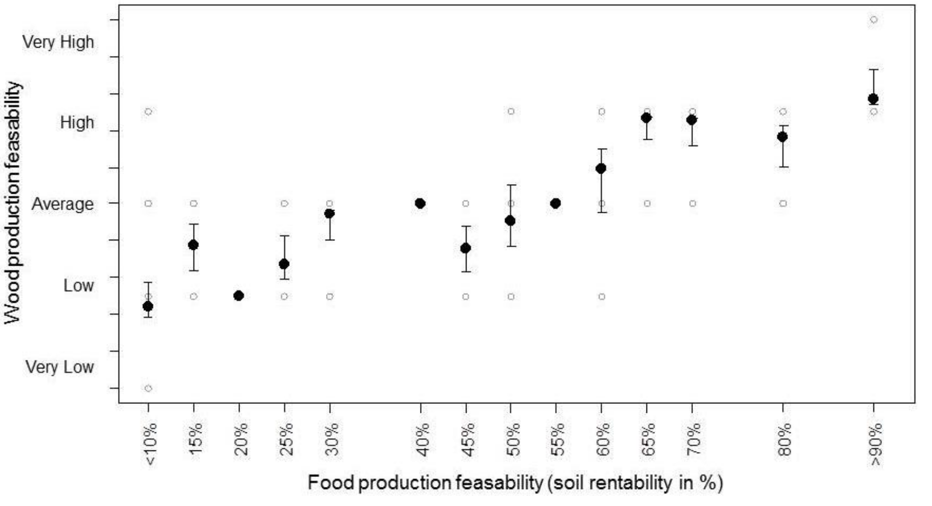

Figure 3.5 - First-level trade-off between food production potential (%) (Provincie Antwerpen, 1998), and wood production potential (five classes) (De Vos, 2000). Grey dots represent feasibility values occurring within the study area (one for each pixel, of which many superposed on each other by model outcomes). Black dots represent mean values, error bars represent asymmetric standard deviations distributed according to relative position of the mean.

Second-level trade-offs: biophysical potential and land use

Only second-level trade-offs between provisioning and regulating services are considered since land use for provisioning services(agriculture vs. forestry) makes them mutually exclusive in current practices within the study area.

- *Food production vs. climate regulation*. As zero food production represents all land uses other than agriculture, climate regulation logically varies from lowest to highest ranges (Figure 3.6,left). At very low to low levels of food production, mean values of climate regulation are highest but with a large variation. At medium, high and very high food production levels, climate regulation drops invariably to very low values.
- W*ood production vs*. *climate regulation*. Zero wood production level refers to all non-forest land uses, which again generates a high variability of climate regulation (Figure 3.6, right). Forests with very low wood production invariably deliver the highest climate regulation. From low toward medium and high wood production levels, mean climate regulation decreases, as well as its variability. Note that mean values as well as maximal values remain relatively high compared to food production.

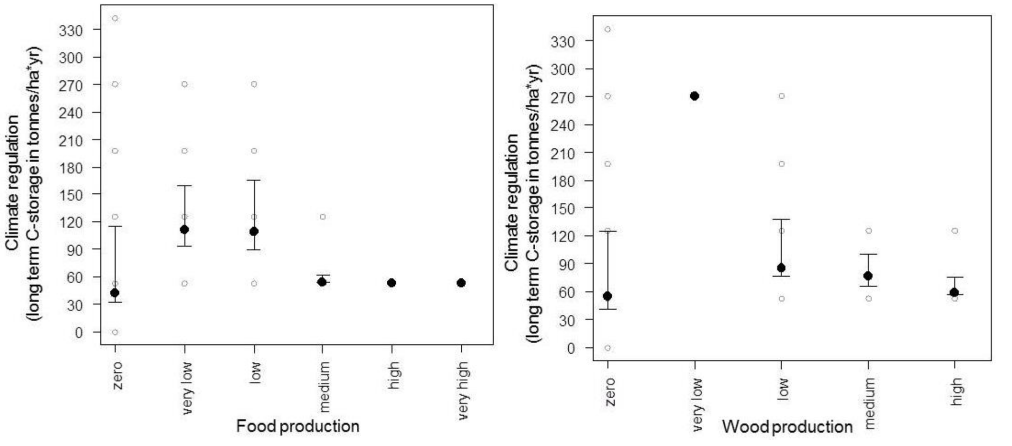

Figure 3.6 – Second level trade-off between Food production (agricultural production) and Climate regulation (long term soil organic carbon storage), and between Wood production and Climate regulation. Grey dots represent feasibility values occurring within the study area (one for each pixel, of which many superposed on each other by model outcomes). Black dots represent mean values, error bars represent asymmetric standard deviations distributed according to relative position of the mean.

3.5.2. The Ecosystem Service Bundle Index

Calculating the difference between actual and optimal EBI allows to identify opportunities for optimizing ecosystem service delivery in a spatially explicit way (Figure 3.7). When the difference between EBI scores is 0, the current land use is optimal. This is the case on 45% of the study area's non-urbanized surface. In areas with positive EBI difference, a shift toward the optimal land use(s) as suggested by the model will improve service delivery. Maximum differences are found in the valleys, although opportunities for improving service delivery also occur on higher grounds.

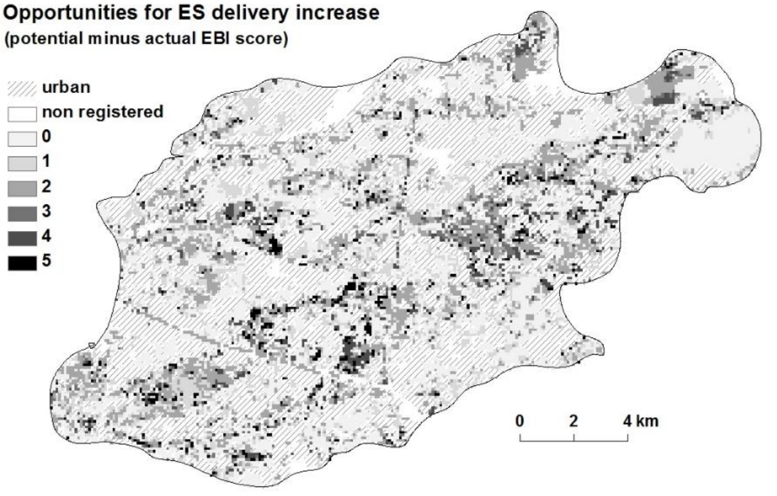

Figure 3.7 – Identification of opportunities for ecosystem service delivery based on the difference between potential and actual ecosystem service bundle delivery, throughout the study area. Note that built-up surfaces (hatched) are not included in the calculations.

3.5.3. Developing optimal scenarios

Every pixel has, attached to its optimal EBI score, one or several optimal land use scenarios to attain the maximum service bundle delivery. Land use planning requires representation of the optimal location of different land use types (Figure 3.8). There are zones where several land use scenarios can generate the maximum score, since only three services were taken into account. Wetlands are logically the most suggested land use in the valleys along the rivers and in local depressions outside of the valleys (Figure 3.8a). On the other hand, intensive cropland is an optimal land use option on well-drained ridges, with clustered blocks toward the southernmost parts where loamy and clayey soils prevail (Figure 3.8b). Production forests are pro-posed in a similar way by the model, but the central and northern parts where sandy soils dominate are more represented (Figure 3.8c) as these are less suitable for cropland. Grasslands, as the optimal combination of food production and carbon storage, are the suggested land use in the wetter and sandy parts of the study area (Figure 3.8d).

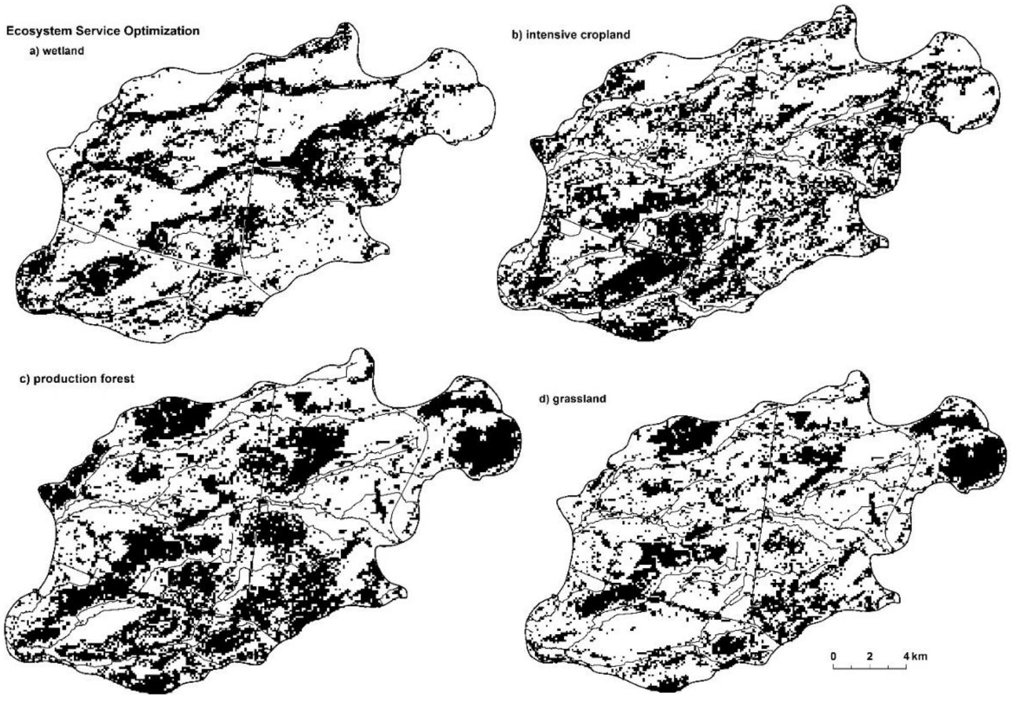

Figure 3.8 – Locations where the model suggested optimal land use to be wetland (a), intensive cropland (b), production forest (c), and grassland (d).

Bearing in mind that optimization was based on only three services and assuming a certain societal demand, average improvements in ecosystem service bundle delivery can be derived for every land use shift suggested in this optimization exercise (Figure 3.9). The extent of improvement depends on both first and second-level trade-offs within each pixel. The average delivery improvements for every land use shift, the number of times it was suggested as well as the uncertainty to apply this in the study area, can offer valuable support in regional policy design. Cropland-to-wetland is by far the most advantageous and most often suggested shift in the Nete basin, despite the fact that in this study only one regulating service was taken into account. The next best land use shift is wetland-to-cropland. This shift is not suggested frequently and results from inconsistencies in the GIS input layers near artificial channels where hydrology is not reflected in the relief as is the case with natural courses. Uncertainties in EBI improvement provide additional arguments to opt for e.g. wetland-to-natural grassland instead of wetland-to-extensive production grassland (same EBI improvement, lower uncertainty).

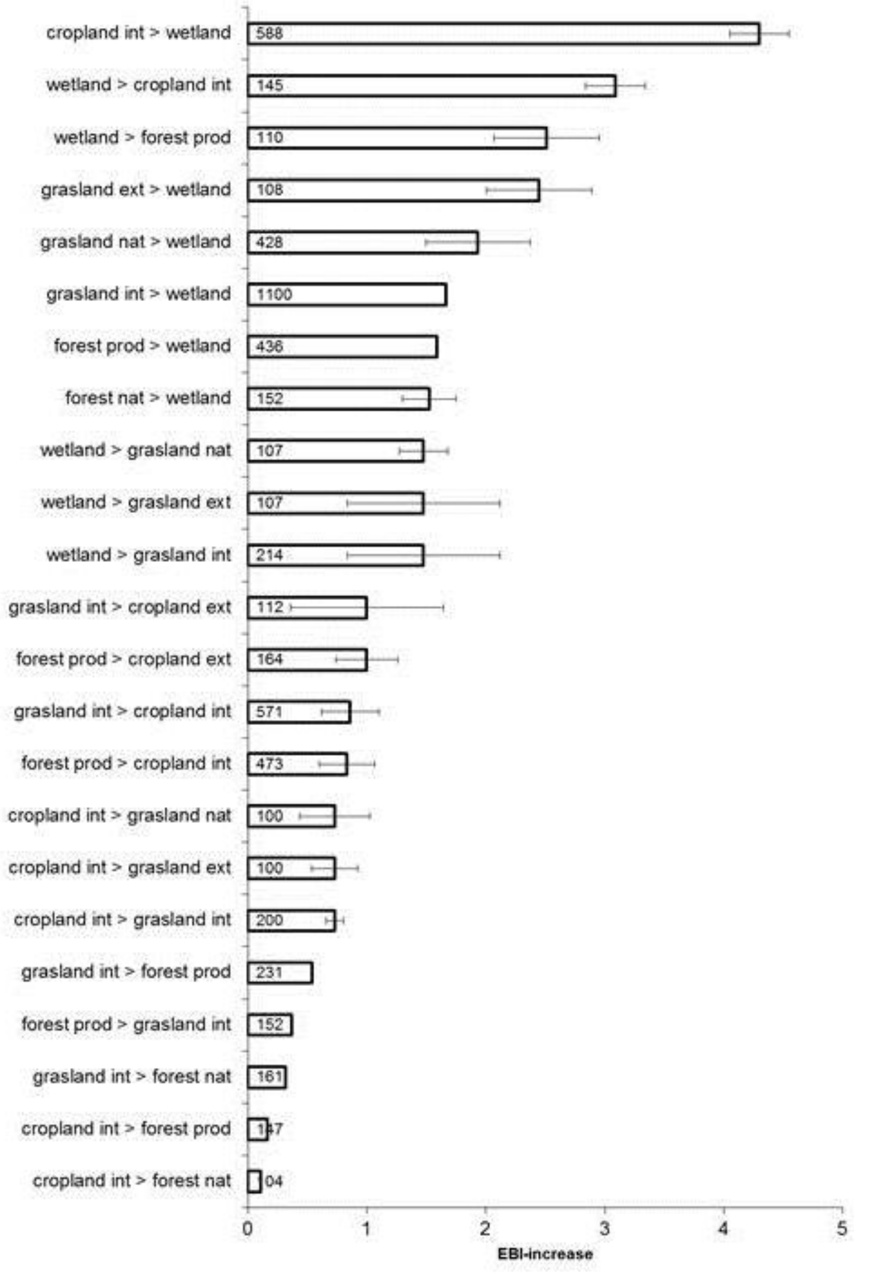

Figure 3.9 – Benefit for all pixels' land use shifts (before > after) proposed by the optimization model in the Nete basin. The benefit is represented as the mean increase in EBI. Bars represent the standard deviation, number represents the number of times the shift is proposed by the optimization model.

3.5.4. Model validation

As mentioned by Landuyt et al. (2013), the use of expert knowledge to define conditional probabilities decreases the possibilities for true (quantitative) model validation. Merging empirical model results and expert knowledge will result in outcomes which differ from both original data sources or other measured proxies or indicators. Validation could only be performed by checking whether predicted changes in ecosystem service bundle delivery actually concur with observed changes between two periods in time. Data to perform this validation are mostly not available. Application of the model posterior to land use changes, in order to take informed decision on these changes, however requires confidence in the model. Pitchforth and Mengersen (2013) propose a suite of alternative validation techniques to establish confidence in expert elicited Bayesian networks. One such technique is face validity, in which experts evaluate whether or not the model results appear to be right. This method is quite easy to apply and it allows for a first assessment of the model's credibility. However, to make the model applicable in real-life context a sound validation should be carried out.

In our case, the land use scenarios with optimal EBI scores were scanned for presence of unrealistic gradients, illogical land use pre-dictions (e.g. wetland on a dry soil) and patchworks of very differing land use types. Face validation was performed during an extended field trip in the study area, during which local experts involved in landscape development planning compared the maps generated by the model with their knowledge of historical land use and development plans. As there exists as yet no standard way of reporting, and face validation was purely qualitative, two map examples are provided to demonstrate the potential of the model.

In Figure 3.10a two different land uses are shown: frame A and B are intensive cropland, resp. with and without a series of parallel drainage ditches; frame C covers forest. Conversion to wetland was deemed most effective in frame A by the local experts. Figure 3.10b shows that in frame A the actual EBI score is very low compared to its optimal EBI, and in both of the other frames the EBI score is close to its optimum. Figure 3.10c shows that converting the land use in frame A from cropland into wetland would increase the EBI score to 10 (see box), and that the optimal land use for service delivery in the two other frames corresponds to the actual land use.

Also, recent changes in land use in several instances concur with the suggested optimization by the model. Figure 3.11 shows an example of a parcel which was covered with cropland in the year 2000 (Figure 3.11a). The potential increase in the EBI score (Figure 3.11b) is predicted 4 by the model if land use is converted into wetland (Figure 3.11c). From Figure 3.11d it can be seen that intensive agricultural use has indeed been abandoned on the parcel, and this was confirmed as a deliberate choice by the local experts in order to enhance ecological potential of the area.

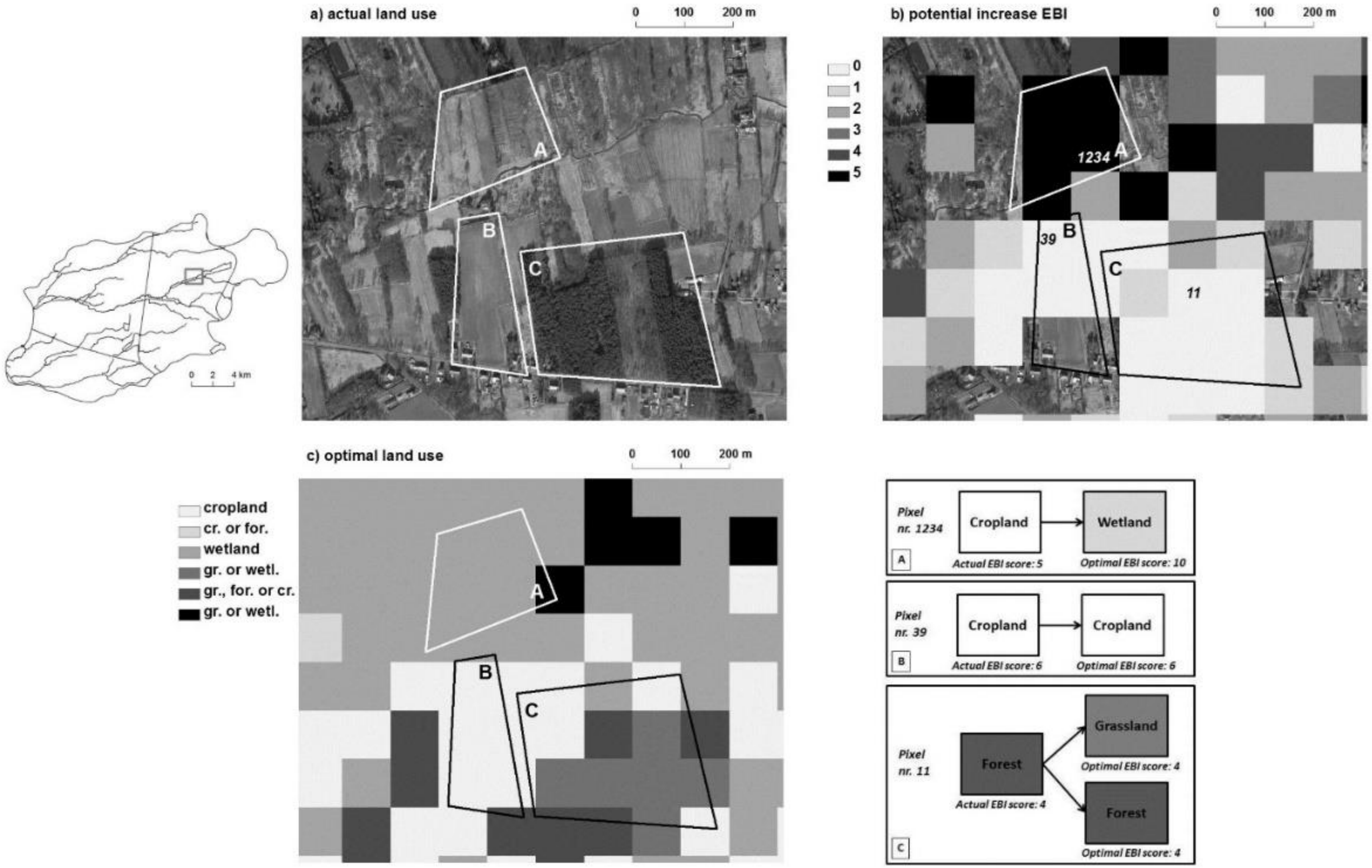

Figure 3.10 – Illustration of a face validity test of the EBI. Cr.: cropland, for.: forest, gr.: grassland. Photograph source: AGIV (2007).

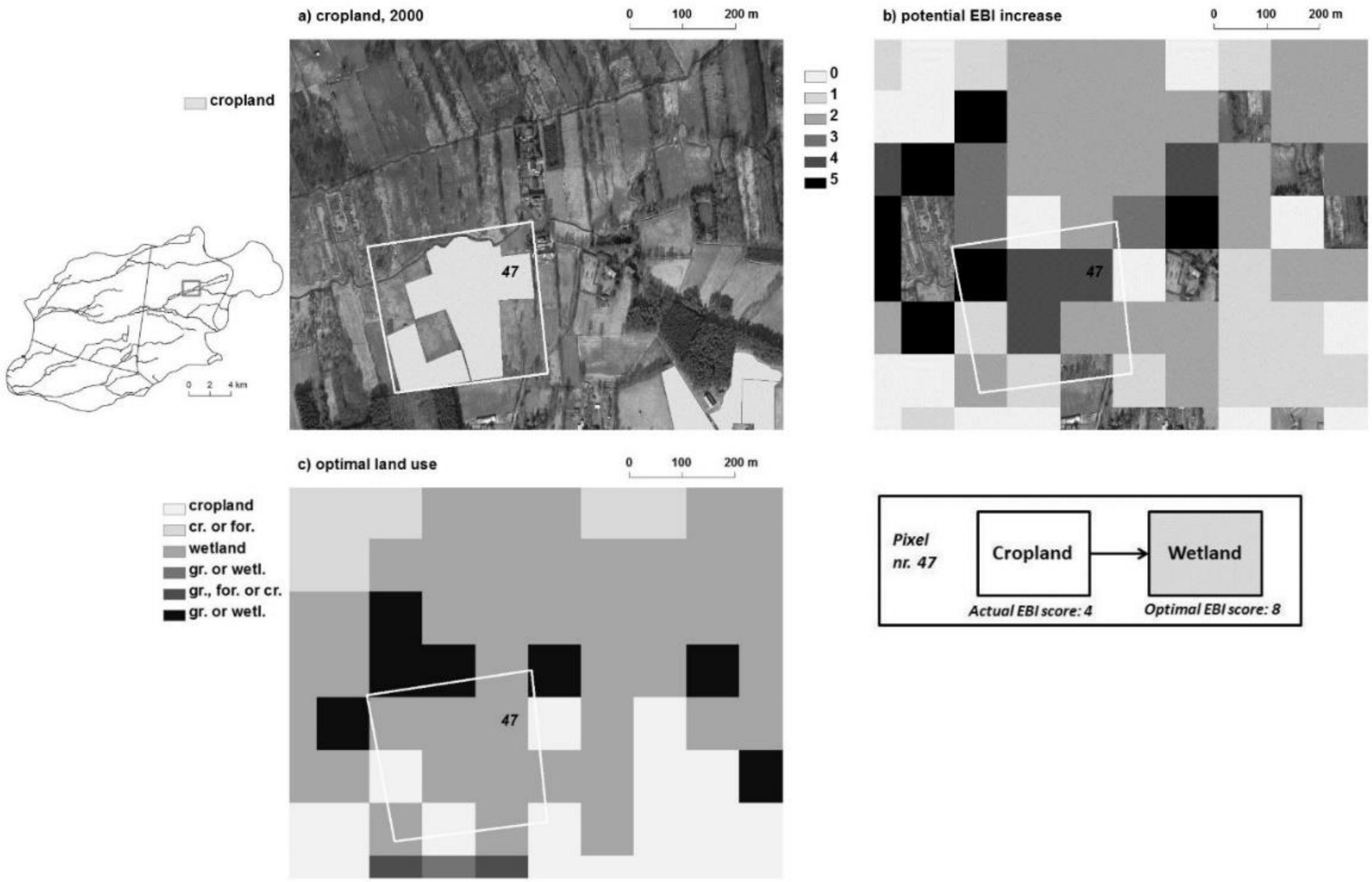

Figure 3.11 – Illustration of field validation in the same restoration area as Figure 3.10. Cr.: cropland, for.: forest, and gr.: grassland. Photograph source: AGIV (2007); cropland maps: ALV (2000, 2011).

3.6. Discussion

The ecosystem service bundle index EBI was developed in response to the eminent need for spatially explicit, transparent and updatable modelling and mapping techniques for multiple services, based on the best available data. EBI as a tool allows to assess ecosystem service delivery in the actual or future scenarios, and to develop spatially explicit optimization scenarios for decision support. EBI allows to assess the capacity of ecosystems to deliver services regardless of the actual land use as well as the effect of different land use scenarios. EBI points out service optimization opportunities as discrepancies between actual land use and the ecosystem's biophysical potential. Since multiple services are comprised in the index, optimization scenarios for a whole bundle of services can be developed by managing controlling factors (biophysical and land use) toward a maximum total ecosystem service delivery scenario. The model offers opportunities to spatially distribute services in a most beneficial way and this on varying spatial scales, and depending on societal or political preferences. The model can be used as a tool in decision supportive approaches such as multi-criteria analysis. The advantage of the model in this respect is the practicability with which an integrated, yet scientific assessment of the effects of land use change on the delivery of multiple ecosystem services can be carried out, making it easier to perform multi-criteria assessments. In comparison with visual assessments techniques such as radar charts, the model has the benefit that the input data as well as the results can be made spatially explicit as it is embedded within a GIS environment, allowing to provide information on high resolution and increasing the usefulness of application in land use assessments.

The EBI was developed to test BBN as a working environment to merge available mechanistic models on hydrology or biogeochemistry, local expert knowledge, etc., to study the effects of land use change on ecosystem system service delivery. Although the EBI was not developed as a one-size-fits-all model, the way in which a Bayesian environment was applied can serve as a basis for the development of a more generic spatially explicit ecosystem service assessment tool, such as InVEST or ARIES, with the additional functionalities of (1) combining all the available knowledge into a single model, (2) to optimize land use toward the delivery of multiple ecosystem services.

As EBI is a prototype, input data for this pilot study was kept simple to allow testing of the basic functioning. Research directions and further potential or required model developments for implementation were revealed in a SWOT analysis (strengths, weaknesses, opportunities and threats) of this pilot study.

3.6.1. Strengths

BBNs allow to capture complexity that is inherent to ecological research and even more to ecosystem services which also involves aspects of economic and social sciences. The current model structure as developed in this chapter takes into account the main biophysical and land use processes of three ecosystem services and results in indicators per ecosystem service which are bundled in a single index (EBI). Bayesian model updating is facilitated by the independence of the relationships which populate the CPTs and is therefore a common practice in Bayesian belief network modeling (Castelletti and Soncini-Sessa, 2007; Marcot et al.,2006). Although the simplest model generating valid results is always preferable (Ockham's razor), complexity can be added by taking more services into account, including spatially explicit judicial land use controls (e.g. Agricultural Development Areas; NATURA 2000), including distance or surface rules when building scenarios and integrating quantification and valuation metrics(economic or non-economic). The combination of quantitative and qualitative data, empirical results and expert judgments in an intuitive structure provides a higher credibility and increases the chances of usage in a political context. Engaging local stakeholders and experts in the model development, application and scenario evaluation process is an interesting and often applied tool in Bayesian modelling for enhancing the validity and acceptance of a model and its outcomes (Smith et al., 2007; Marcot et al., 2006). In fact, offering discussion-support and facilitating stakeholder engagement are major advantages of Bayesian modelling. Moreover, the visualization of model output scenarios through mapping offers opportunities in the context of participatory modelling which Fish (2011) denoted as essential in ecosystem service assessment (Castelletti and Soncini-Sessa, 2007). Further progress and application of EBI will include participative processes:

- In model parts where empirical data or achieved knowledge is lacking or uncertain, probabilities can be assigned and CPTs populated through expert discussions. For example, the relationships between actual agricultural provision, realized land use and provision potential can be refined in several rounds of expert consultation. Conflicting opinions raised between experts could be translated to uncertainties and be included in the model.
- Deliberative consensus techniques should determine the weighting factors used in the EBI Eq. (1). As these factors reflect the relative importance assigned to the services, they strongly influence the optimization results. Therefore, deciding on suitable weights is a political question with a potentially high socio-economic impact and should be based on a broad consensus of local stakeholders. Local experts can also validate the model. Mapping the model outcomes (e.g. for different weighting factors) allows for visual model validation and can be conducted through a stakeholder workshop. Formulated suggestions or raised doubts implemented into the model may strongly enhance its validity and acceptance compared to top-down communicated maps

3.6.2. Weaknesses

Due to limited availability of empirical data, model validation in ecosystem service modelling is often challenging (Kareiva et al.,2011). The problem is that many services in se cannot be measured directly while commonly used statistical methods like K-fold cross validation almost always rely on empirical data. Model validation however is strongly encouraged: the few studies that have put ecosystem service maps to the test immediately pointed out major biases (Villa et al., 2009; Eigenbrod et al., 2010). Alternative validation methods often used in Bayesian modelling, such as sensitivity analysis and visual network structure appraisal, mainly rely on expert judgment. Nevertheless, assembling objective expert opinions to verify accuracy and robustness of these models is often difficult. In this study, we relied on scientific expert evaluations of the network structure and visual assessments of the mapped model output to validate EBI. Validation at the appropriate application level (e.g. local stakeholders and experts of a subcatchment) is therefore strongly advised.

In this pilot study only three services were included to demonstrate the potential of EBI. Application in real-life contexts however requires incorporation of all relevant provisioning, regulating and cultural ecosystem services. A great challenge when adding more services lies in the design of the weighting function (prioritization of services, equation 1). This should be done cautiously using the appropriate scientific tools as this can strongly affect the optimization outcomes. The weighting can entirely be driven by expert judgment, as presented here, or it can be done on a participatory basis involving local stakeholders and policy makers (e.g. Koschke et al.,2012). A more objective weighting method could be based on the ‘spatial accounts’ concept as applied by the Flemish government in spatial planning. Instead of defining surface goals for land use types in a certain study area, quantitative goals can be defined for each ecosystem service. The weighting equation can then be trained so that for each ecosystem service a maximum % of the goal is fulfilled. This however requires repeated generation of the weighting factors and should (partially) be dealt with outside of the Bayesian environment. Although this weighting is very sensitive, it gives an indication of the impact degree of changes in land use. Alternatively, the weighting equation could be eliminated and the index expressed in economic terms.

As feedback mechanisms can be difficult to implement in Bayesian belief networks, another challenge when increasing the number of services lies in implementing spatial and temporal interactions between ecosystem services and conditions.

3.6.3. Opportunities

Many policy makers, decision makers and stakeholder groups have looked for ways to generate awareness for our dependency on natural systems and to stimulate incorporation of ecosystem services in decision making. Qualitative maps and illustrative monetary quantifications have been very convincing and powerful arguments. However, at least in Flanders, there is a very strong demand for concrete and applied planning for ecosystem services in the local political realities. Therefore, the risks of taking decisions with adverse effects must be minimized, remaining uncertainty must be acknowledged, scientific credibility and public acceptance must be guaranteed. EBI is a first step to foresee in such an integrated approach.

The potentials of EBI are demonstrated throughout this chapter, but only a full application in a pilot study, including expansion to a broader range of services, incorporation of judicial land use claims and involvement of stakeholders and decision makers in different stages of EBI development will deliver a true proof of concept. Several of these case-studies at different scales are being planned, and the model extensions are under development.

While there are still important limitations related to the use of Bayesian networks in ecological modelling, recent growing scientific interest will probably lead to model improvements toward the future (Aguilera et al., 2011). Some shortcomings of Bayesian networks in regard to ecological modelling, like the absence of feed-back loops, are for example currently dealt with by the use of time dependent nodes (Bashari et al., 2008) or time sliced models (Cain,2001; Kjaerulff, 1992). Further technical advances will similarly increase the feasibility of Bayesian models to take into account a broader set of ecosystem services and to expand toward spatially explicit applications.

3.6.4. Threats

The same political eminence and the former preference for traditional top-down approaches might cause decision makers to apply the existing ‘as is’ mapping tools on their local case and select maps and quantification results to serve their organization’s stakes. The consequences, on the field as well as for the long-term credibility of the ecosystem service concept, might only surface on the long run.

Although EBI is in itself a fairly simple assessment tool, potential users need to appreciate the complexity of the interacting ecosystem services in the total bundle, and – to a certain extent – also comprehend the combination of biophysical and land use variables, to thoughtfully apply the index for land use planning. The challenging requirement of dialog with experts, stakeholders from different policy domains and local populace might cause reluctance for application. Scientist and experts from their side, mostly being unfamiliar in science/policy dialog, might be critical toward EBI since being objective has been sometimes interpreted as being technically outside of political realities. Both decision makers and experts however increasingly realize that effective implementation of state-of-the-art knowledge-based policy requires this engagement.

3.7. Conclusions

Ecosystem service mapping and quantification is a complex research field, involving challenges concerning data scarcity, complexity, validation, credibility, participatory design and many others. Meanwhile, application of the ecosystem service concept requires simple indicators for decision support. This chapter presents a principle by which complexity and uncertainties can be crystalized in a single index, allowing for practicable but qualitative assessments of the impacts of management scenarios on bundled service delivery. EBI allows for supporting decision makers in providing information on the consequences of their choices. It enables to refer their choice to a benchmark: the optimum, at certain circumstances, in a simple yet complete and underpinned, science based and participatory manner. EBI is a prototype. It shows however that acceptably realistic results are already obtained despite the index being in an early stage of development. Application of this and similar approaches in real-life pilot cases are needed to put EBI to the test.

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4

Processes as indicators of ecosystem services

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4.1. Abstract

Intensively used coastal zones often know a history of hard defense structures to prevent erosion and protect infrastructure against floods. The interruption of sand transport between sea, beach and dunes however causes a domination of late successional stages such as dune shrub. With the decline of young, dynamic vegetation types, a change occurs in the provision of ecosystem services. In spite of the growing awareness on the role of dune dynamics to support human well-being and biodiversity, redynamisation of dunes is rarely implemented in coastal zone management. It has been argued in research documents that this may be caused by a failure to make those benefits tangible and specific. This study aims to underpin the added value of dynamic versus fixed dunes. Five different ecosystem services in a case-study in Belgium were quantified based on (compound) indicators and expressed in monetary units. The value of a natural, dynamic dune system covering the entire gradient of dune succession and dominated by young successional stages was compared with the value of a fixed dune system dominated by late successional stages. The results indicate that a dynamic dune complex may create up to ~50% higher economic benefits, and that the main benefits are on account of recreation and coastal safety maintenance. The results underpin the statement that we can only continue benefitting from the services dunes provide if we accept their mobile nature, but that redynamisation requires a site-specific feasibility analysis.

4.2. Introduction

In recent years, several studies demonstrated the contribution of coastal dunes to the delivery of ecosystem services such as protection against floods, recreation, salinization prevention and drinking water production (Doody et al. 2005; Everard et al. 2010; Arens et al. 2013; Lithgow et al. 2013). Coastal dunes in Europe are also known to hold large biodiversity values (Martínez et al. 2004). In north Belgium for example, they harbour 40 to 60% of the total number of species in an area covering less than 1 % of the region’s total surface area.

In spite of the important benefits they provide for human well-being and the high biodiversity values, coastal dunes are among the most damaged ecosystems. The main causes of degradation are urban development, agriculture, waste disposal, mining and military activities (Lithgow et al. 2013). Around the world, the rate of change associated with construction in coastal zones has been occurring several times faster than changes occurring inland (Martínez et al. 2004). In Belgium, urbanization has led to devastation of nearly 50 % of the coastal dunes. Besides direct destruction of dunes, dunes also suffer from degradation resulting from the placement of shore protection structures to protect these developments against erosion and flooding (Nordstrom 2000; Pilkey & Cooper 2014). Hard engineering structures such as dykes and groynes block sand transport and may in some cases even aggravate erosion (Pilarczyk 1998; Boers et al. 2009; Vaidya and Kudale 2015).

Fragmentation by roads, houses, railways, land use changes, … obstructs the flow of sand and prevents dunes from rejuvenating. Consequently, dynamic dune systems covering the entire range of dune habitats and with an important portion of bare sand and young vegetation types evolve towards a fixed ecosystem dominated by late successional stages. The loss of pioneer stages does not only affect the ecosystem services typically associated with these habitats (Boerema et al. 2016), but also decreases biodiversity. Research demonstrates that habitats with the highest biodiversity values in coastal dunes are associated with early successional stages that depend on the supply of fresh sand (Howe et al. 2010; Provoost et al. 2011).

As stated by Favennec (2002): “To continue to benefit from the ecosystem services dunes provide, we must accept the mobile nature of dunes and their fluctuations in sand budget.”. Not only does conventional flood defence hamper sand dynamics, its maintenance may also become unsustainable in light of sea level rise. Nature-based solutions are a long-term sustainable alternative with multiple additional benefits (Temmerman et al. 2013). To date, the economic benefits of a dynamic dune system remain undocumented, which may explain why decision-makers in most European countries are reluctant to allow natural forces to shape coastal defence (Nordstrom et al. 2015).

This study provides evidence for the socio-economic benefits of dynamic dunes for the case of Flanders, Belgium. To this purpose it builds further on the methodology and accounting framework to quantify and monetise the ecosystem services from natural areas in Flanders (Staes et al. 2017), in order to estimate more in detail the ecosystem services of coastal dunes along the Westcoast in Flanders. In a second step, we describe, quantify and monetise the differences between dynamic and fixed dunes, which thus far have mainly been expressed in descriptive analyses (Everard et al. 2010; Hanley et al. 2014; Feagin et al. 2015). The question addressed in this chapter is if a dynamic dune system with constant sand movement provides more and/or more valuable ecosystem services than a dune system fixed by vegetation. In other words: can remobilization of dunes, by removing hard structures that hamper sand transport, result in significant socio-economic profits?

This chapter does not address issues related to the concept or available methods and data for mapping, quantifying and monetising ecosystem services in general (as e.g. discussed in Liekens et al. 2013; Landuyt et al. 2016). This chapter discusses the limitations and uncertainties in methods and data for the selected ecosystem services and the case study, and indicates the range of uncertainties in quantification and monetisation. This is only one part of the overall uncertainties related to assessing ecosystem services and the socio-economic importance of natural areas like coastal dunes.

4.3. Methodology

4.3.1. Study area

The study area (Figure 4.1) is located in the northwest of Belgium and covers the nature reserve 'Westhoek' (~340 ha). The area used to be a highly mobile dune field dominated by dynamic vegetation such as marram grass (*Ammophilia arenaria*), both in the frontal zone (up to 50-200m from the shoreline) and more inland (>1000m from the shoreline). The frontal dunes became largely fixed and dominated by shrub after the construction of a dune revetment blocking the transport of sand from beach to dune at the end of the 1970s. The large mobile dune in the central part of the reserve also got increasingly fixed, most probably due to increased climatic variability and – as a result – increased precipitation in certain years (Provoost et al. 2014). In more recent years, the government spent great efforts to maintain the diversity of habitats and prevent the area to develop into an entirely fixed dune complex dominated by shrub and eventually woodland. This included the creation of 2 artificial breaches (Figure 4.1).

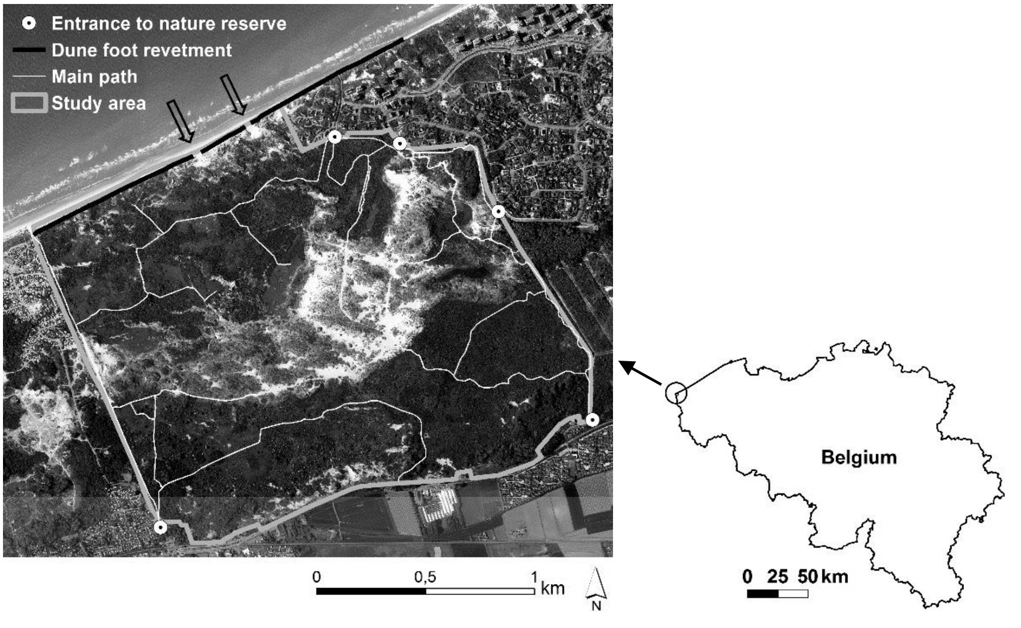

Figure 4.1 – Location and aerial photograph of the study area with indication of the breaches (arrows) (AGIV 2012)

4.3.2. Ecosystem services analysis: quantification and economic valuation

This study builds on the methodological framework of the ECOPLAN project to assess, quantify and monetise the ecosystem goods and services in Flanders (www.ecosysteemdiensten.be). This accounting framework has been used for assessment at the national level of all NATURA 2000 sites (Staes et al. 2017), in the context of the Flanders Regional Ecosystem Assessment (Jacobs et al. 2016) and to evaluate impacts of policies at the regional scale or at the project level (Broekx et al. 2013). Based on a qualitative assessment of the importance of ecosystem services provided by coastal sand dunes (Everard et al. 2010), interviews with stakeholders and visitors (De Nocker et al. 2015) and local site specificities, five ecosystem services were selected: water provision, coastal safety maintenance, water quality regulation, climate regulation and recreation. Ecosystem services related to the capture of air pollutants by vegetation or impacts from nearby natural areas on real estate values or public health are relevant for this area but were not included because of the limited scope for better assessment within this study. Other ecosystem services which are considered in similar studies on dune ecosystem services (e.g. Everard et al. 2010) have not been included, such as wood production, military use and educational use. These were either absent in the study area or had low economic value.

For all ecosystem services except coastal safety maintenance, quantification and monetisation are performed using today's configuration of habitats (see Appendix E for habitat description) and socio-economic use of the nature reserve. Because of the presence of an artificial sea defence, which partly replaces the natural protection against floods, it was not possible to assess coastal safety maintenance based on today's configuration of habitats. Instead, the situation from before construction of the dune foot revetment was used (year 1953).

For each of the selected ecosystem services a compound indicator was developed that expresses how much of an ecosystem service is being delivered per year (Table 4.2). The total value of the ecosystem services in € y^{-1} is the product of these indicators with its monetary value. Using the ArcGIS-tool 'Zonal Statistics' (ESRI 2016) a mean value per habitat type was calculated for each ecosystem service (€ ha^{-1} y^{-1}). To take into account uncertainty on the values, a low and high estimate of the value was provided when data was available.

Water provisioning

Coastal dunes are typically associated with a freshwater lens. The coarse texture of the dune sand allows quick replenishment of the reservoir while the infiltrating rain water is naturally purified in its course through the sand. The shallow location of the reservoir makes dunes attractive for abstraction of drinking water, as pumping and treatment costs are limited. In the case of the study site, water abstraction infrastructure is located at 200m distance from the border of the study area. Part of the extracted water comes from the dunes within the study area. Recently, the extraction was reduced so the reserve is no longer affected by desiccation. The applied methodology for calculating water provisioning is described in Staes et al. (2017) and Van der Biest et al. (2015). This GIS-tool first estimates the amount of naturally infiltrated water based on precipitation, soil texture, groundwater depth and vegetation type. Secondly, cones of depression were modelled which take into account the yearly volume of water abstracted.

Groundwater as a source for producing drinking water is very valuable in the coastal region, because there is a limited capacity to produce drinking water in that region and demand is high, especially in the summer season due to tourism. There are no real market prices for the capture of groundwater as a source for producing groundwater, although licenses to capture groundwater are limited, and a specific natural resource tax is applied to groundwater abstraction (0.075 € m^{-3}). This Flemish groundwater tax can be seen as an indicator to compensate for the environmental and resource costs, as defined in the Water Framework Directive, and is used as the lower bound for the monetisation in the overall ECOPLAN framework. However, the marginal value of groundwater is higher in that region, which is illustrated by the high prices local drinking water companies pay to import water from other regions or the high costs of specific technologies used to produce water locally. These additional costs are estimated at 0.2 € m^{-3}, which reflects the high estimate of the monetary value (Broekx et al. 2013). The drinking water companies and its clients are the beneficiaries of this ecosystem service.

Water quality regulation

One of the important criteria for good groundwater quality, both for the purpose of clean drinking water provision as for biodiversity support, is a sufficiently low concentration of nitrogen (N). In the study area, N is mainly available through atmospheric deposition caused by industry, traffic and agriculture (yearly average of 11 kg N ha^{-1}, VLM 2011). Nitrogen in dune ecosystems can either be retained in the ecosystem by plants and organic matter, or lost by leaching, nitrification and subsequent denitrification, and grazing (Olff et al. 1993). N retention is strongly influenced by the presence of calcium in dune sand and uptake by vegetation. Calcareous soils (young sand deposits), are characterised by higher nitrification rates in comparison with soils with a decalcified top layer (ten Harkel et al. 1998). With low degrees of denitrification, this causes nitrate leaching to groundwater (ten Harkel et al. 1998; Pinay et al. 2007). Nitrification rates are smaller in more developed soils with a decalcified top layer and lower pH. Incorporation in plants is also higher in well-developed soils, which directly take up ammonium from atmospheric deposition and where nitrate is partly lost through mineralisation and nitrification of dead organic material. Ten Harkel et al. (1998) showed that in foredunes in the Netherlands about 70% of the atmospheric deposition (ammonium) leaves the soil as nitrate, while at non-grazed, dry innerdunes (grasslands which are decalcified down to 40-50 cm depth) only 13% leaches to groundwater. In dune slacks of older successional stages (as found in the study area), where groundwater flow reaches the surface, N removal by denitrification is estimated to account for 5 % of the atmospheric deposition (Adema and Grootjans 2003). Dune shrub with *Hippophae rhamnoides* lives in symbiosis with N-fixing bacteria and nearly triples the amount of leaching to groundwater compared to atmospheric deposition (Stuyfzand 1984). In the absence of literature values for dune shrub with *S. repens*, we used the denitrification value for old dune slacks (5%), as this vegetation type often evolves from dune slacks. Average N-leaching from forests within the study area is estimated 29% (Staes et al. 2017).

The monetary value of N retention is based on the benefits to society from health problems (intestinal cancer) associated with N intake through drinking water (Van Grinsven et al. 2010). The benefits of the ecosystem service are for the health care insurance (avoided expenses) and the patients and their families (avoided private health care costs and suffering). For Belgium, this value is estimated at a range of 0.6 to 2.4 € kg N^{-1}, accounting for uncertainties in the exposure assessment (% of population using drink water from the controlled tap water network).

Coastal safety maintenance

Protection against floods comprises two aspects: (1) the mass of sand deposited in the past that now forms a physical barrier against waves and water; and (2) the maintenance or improvement of this mass of sand by the supply of fresh sand, and the capacity of the system to keep up with sea level rise. The first is usually estimated by quantifying the damage costs and number of casualties caused by flooding (TEEB 2010; Koks et al. 2014). The second can be valued using the replacement cost for artificial dune foot nourishment. As this study focusses on the dynamic processes of erosion and sedimentation in dunes and their contribution to human well-being, it was decided to use the replacement cost for dune foot nourishments to value the benefit of coastal safety maintenance.

The volume of sand accumulated per year in shifting dunes along the shoreline is used as indicator for the maintenance of coastal safety. Several studies have shown that Ammophila species require a certain amount of sand burial each year in order to remain vigorous (De Rooij-Van Der Goes et al. 1995; Keijsers et al. 2015). Without supply of fresh sand, soil starts to develop and marram grass degenerates because of the occurrence of nematodes. The presence of marram grass can thus be used as indicator of active sand transport. According to Aggenbach and Jalink (1999), marram grass thrives under deposition rates of more than 10 cm per year. Vegetation remains dynamic at lower rates of deposition (4 cm y^{-1}), but turns into moss at 2 cm y^{-1}. This is in line with the findings of Martin (1959), who states that annual sand burial rates of more than 7 cm are needed to keep Ammophila vegetation vigorous. In the situation from before construction of the dune foot revetment (1953), a continuous strip of Ammophila is found from the beach up to ~200m inland (frontal dunes). It can be expected that, given the large distance from the beach, a certain amount of the deposited sand originates from wind erosion within the frontal dunes. The estimate of sand deposition (4 to 10 cm y^{-1}) should be corrected for this. Based on a comparison with average sedimentation rates in coastal dunes in neighboring countries (5 to 10 m^3 m^{-1} y^{-1} in The Netherlands, Arens et al. 2013; 2 m^3 m^{-1} y^{-1} in France, Carter 1980), it is assumed that half of the sediment in the shoreline is transported from the beach and half originates from erosion within the dunes. The minimum and maximum estimates of coastal safety maintenance are thus a sedimentation rate of 2 and 5 cm y^{-1}, resulting in an additional sand volume of 4 to 10 m^3 m^{-1} y^{-1}, or 200 to 500 m^3 ha^{-1} y^{-1}, for a frontal dune zone of 200 m wide. Within the study area, marram grass is also found at a distance of up to 1 km from the shoreline. These inland shifting dunes however do not form a continuous complex with those from the shoreline but are separated by a strip of fixed dunes parallel to the coast. The occurrence of marram grass here can be attributed solely to eolian and management related (trampling, grazing) forces continuously reworking the present sand, rather than deposition of fresh sand from the beach. Accumulation of fresh sand, and thus coastal safety maintenance, is only taken into account for the embryonic and shifting dunes in the frontal dune zone. To estimate the benefit, the surface area covered with embryonic and shifting dunes along the shoreline should be multiplied with the yearly sedimentation rate.

The cost to replace natural supply of sand by artificial nourishments is used as indicator for the economic value of sand accumulation. According to Deltafact (2012) 1 m^3 of dune foot nourishment costs 16 €. The economic value for coastal safety maintenance by dunes (Table 4.2) corresponds relatively well with the costs to maintain an existing dyke (60 to 150 € m^{-1} y^{-1}) and increase its height as adaptation to sea level rise by 2100 (5000 € m^{-1}) (MDK 2016). When spreading these expenses over the period 2017-2100, this would lead to a cost of 0.15 to 0.26 million € y^{-1} for the entire stretch of the reserve (1260 m).

Climate regulation

Climate regulation through soil organic carbon sequestration is quantified using the results of an empirical study on local field measurements in northern Belgium (Ottoy et al. 2015). In this study, total amount of carbon stored in the upper first meter of the soil is estimated based on statistical characterization of soil profile characteristics (texture and groundwater depth) and land use type. Yearly additional sequestration is then calculated by dividing the amount of stored carbon – which represents equilibrium soil concentrations – by 100, assuming that soils reach their equilibrium SOC concentration without further accumulation after a period of 100 years (Broekx et al. 2013). The obtained values (Table 4.2) compare relatively well with measurements of soil organic carbon in coastal dunes in the United Kingdom (Beaumont et al. 2014), where mean sequestration rates in the upper 15 cm over a period of 160 years were estimated at 582 ± 262 kg C ha^{-1} y^{-1} in dry dune grasslands and 730 ± 262 kg C ha^{-1} y^{-1} in wet dune slacks.

The benefits from carbon storage are valued based on the literature on the social costs of carbon and the guidelines by mainly European economic administrations on how to account for carbon emissions or capture in economic analysis. The benefits relate to the avoided damages from climate change, which are however hard to estimate and with great uncertainty. Therefore, the marginal costs of meeting the greenhouse gas emissions reduction targets - as part of policy plans to limit global warming to 2°C temperature increase relative to the pre-industrial level of 1780 - is used as a proxy of the value policy makers and society give to carbon storage. Using data from literature and recommendations from economic agencies, this can be estimated at 220 € ton C^{-1} (Aertsens et al. 2013).

Recreation

Although it is well accepted that the study area is important for recreation and tourism, there are no area specific data to assess the total number of visits per year to the site. Therefore, the number of visits has been estimated using a detailed model analysis that accounts for attractiveness of landscapes, infrastructure for recreation, the proximity of population, data and preferences of visits to natural areas in Flanders and data on tourism and their activities (De Nocker et al. 2017). It is estimated that the study area attracts around 300000 to 500000 visits a year.

To assess the total economic value of the area as a whole, and accounting for the limited studies on economic valuation of recreation in Flanders, we used the data from scientific literature on the estimation of the welfare benefit a visitor attaches to its visits (Staes et al. 2017). To this purpose, we build on the meta-analysis of relevant literature on site-specific per-visit values, taking into account spatial variation and differences per habitat type made in the context of the National ecosystem assessment for the UK (Sen et al. 2014). This amounts to 4.5 € per visit on average, with a range of 3-9 € per visit, accounting for the large uncertainties in assessing this value and the benefit transfer to the study area. This amounts to 0.9 – 4.5 € y^{-1}).

This meta-analysis confirms that the values for coastal nature are in line with the average values, but does not allow to distinguish between the detailed habitat types we use in our study. Therefore we used additional information on preferences from visitors to attribute this value to different habitat types. First, we used the number of pictures taken from each habitat type and uploaded to the websites Flickr and Panoramio (Figueroa-Alfaro and Tang 2016) as an indicator of the relative importance of different habitat types for these visitors. The contribution of each habitat in attracting visitors to the reserve is assumed to be reflected in the relative number of pictures taken in the different habitat types compared to the total number of pictures (239). Each of the 239 uploaded pictures was assigned a habitat type based on visual interpretation of the image and, when available, description of the photo. An additional correction factor was applied for the non-linear relationship between surface area of attractive landscapes and number of visits (an increase in surface area of highly attractive habitats is expected to result in a relatively smaller increase in number of visits). In the absence of quantitative data, studies were used that assess the relationships between natural surface area (independent of its attractiveness) and number of visits (Colson 2009; Siikamäki 2011). An increase in natural area of 10% resulted in an increase in numbers of visits of 4%. This correction factor was applied to the difference in monetary value between the habitat in the fixed and in the dynamic scenario.

4.3.3. Scenarios

The benefits of dynamic dunes are estimated by comparing the economic value of a dune system consisting of both dynamic and stabilized dune types (dynamic scenario) with the value of a largely stabilized dune landscape (fixed scenario).

A dynamic dune system is characterized by an important amount of bare sand which is subject to eolian sand transport. All the different stages in dune development are present, creating opportunities for high species diversity. Embryonic and shifting dunes dominate the frontal dunes. Vegetation here is adapted to constant burial and acts as sediment trap, thus building up new dunes (Feagin et al. 2015). Further inland, dune slacks are found that testify of recent blowout activity (Arens et al. 2013). A precondition for a dynamic dune system is the continuous transport of sand by wind and/or hydrodynamic forces, on the interface between beach and dune as well as more inland. No artificial structures should impede sand transport from sea to coast (e.g. by groynes) and between beach and dunes (dykes, revetments etc.), or act as traps for sand migrating further inland (roads and houses). For the study area, this scenario corresponds best with the situation before construction of the dune revetment. The habitat composition for this scenario (Figure 4.2 left, Table 4.1) is based on aerial photographs of 1953 (INBO, unpublished data).

In the fixed scenario (Figure 4.2 right, Table 4.1), which is best represented by the present situation, sand transport to, from and within the dunes is hampered by concrete structures. Without supply of fresh sand or regular disturbance (wind, herbivores, trampling, …), sand becomes colonized by vegetation and succession leads to fixation of the dunes by shrub and eventually woodland. Dune slacks become less profound and their water table lowers due to increased evapotranspiration, eventually leading to domination by shrub. Encroachment by dense vegetation impedes windblown formation of new dune slacks. Shifting dunes are found only in a narrow strip along the shoreline where strong eolian forces are able to blow some of the beach sand over the dyke into the dunes. The habitat composition for this scenario (Figure 4.2 right, Table 4.1) is based on satellite images of 2016 (INBO, unpublished data).

Based on the aerial photographs of 1953 it was not always possible to distinguish dune shrub with *H. rhamnoides* from dune shrub with *Salix repens*. For the ecosystem service calculations, the surface ratio between *S. repens* and *H. rhamnoides* in the present situation was extrapolated to allocate both types of shrubs where uncertainty exists in the photographs of 1953.

A comparison is made between the ecosystem services in both scenarios for the entire study area, for the dunes in the frontal zone (shoreline) and for the inner dunes. The frontal dune zone is defined as the dune area closest to the sea, which in a natural situation with active sand supply is dominated by embryonic and shifting dunes and may extend up to 200 m from the beach. It is delineated based on the extent of embryonic and shifting dunes in the dynamic situation of 1953 (Figure 4.2). In the fixed scenario, large parts of the embryonic and shifting dunes within the frontal zone have become fixed with moss, grass or shrub.

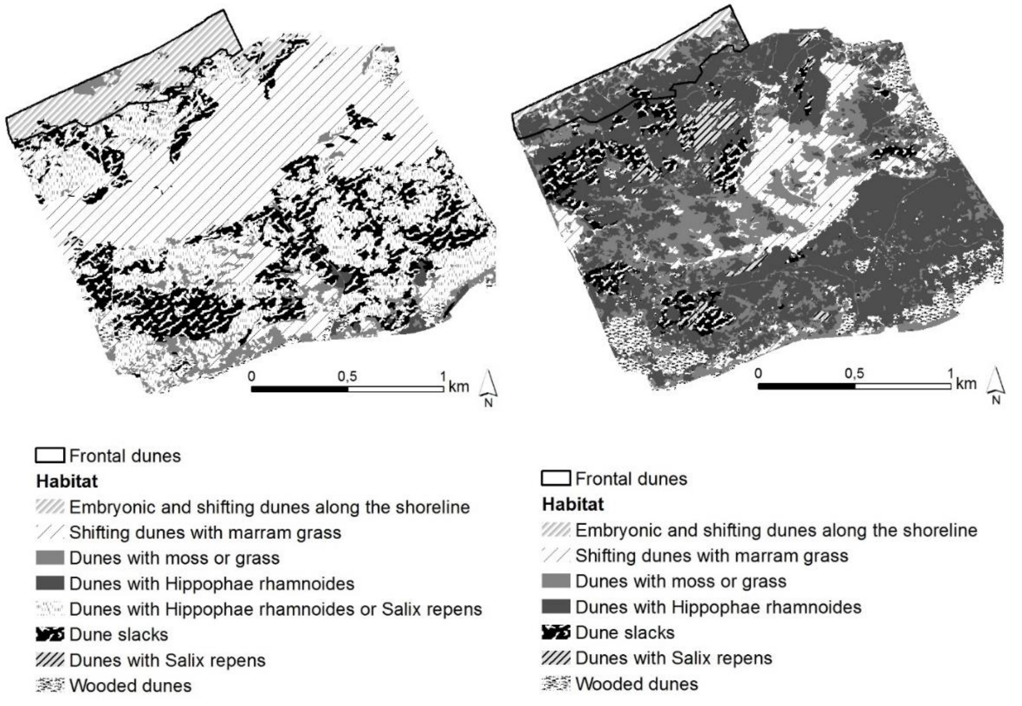

Figure 4.2 - Left: dynamic scenario (situation ~1953). Right: fixed scenario (situation ~2016).

Table 4.1 - Surface area (ha) per habitat type in the dynamic and in the fixed scenario.

Habitat	**Dynamic**		**Fixed**	
	ha	**%**	**ha**	**%**
Embryonic and shifting dunes along the shoreline	24,3	7	7,3	2
Shifting dunes with marram grass	126,1	38	39,4	12
Dunes with moss or grass	19,1	6	70,8	21
Dunes with *Hippophae rhamnoides*	76,9	23	162,7	48
Dunes with *Salix repens*	8,6	3	10,5	3
Dune slacks	75,8	23	24,6	7
Wooded dunes	5,2	2	21,0	6
TOTAL	*336,0*	*100*	*336,3*	*100*

4.4. Results

4.4.1. Ecosystem services of different dune habitats

In general, there are large differences in ecosystem service supply and associated values among the various habitats (Figure 4.3, Table 4.2), ranging from 1342 – 5514 € ha^{-1} y^{-1} (dunes with moss or grass) to 10649 – 52097 € ha^{-1} y^{-1} (shifting dunes with marram grass). This is the result of the high value for recreation and to a minor extent coastal safety maintenance, the latter being a service restricted to the frontal dunes. The habitats with the highest value for recreation are habitats associated with wet soils (dune slacks and dunes with *S. repens*) and habitats with shifting sand (shifting dunes with marram grass and embryonic and shifting dunes).

In comparison with coastal safety maintenance and recreation, average values for the three other ecosystem services are relatively small. The sum of the lowest estimates for recreation and coastal safety maintenance (1011 – 3033 € ha^{-1} y^{-1} for dunes with moss or grass) is still higher than the sum of the maximum estimate of the three other ecosystem services in the same habitat (307 – 454 € ha^{-1} y^{-1}).

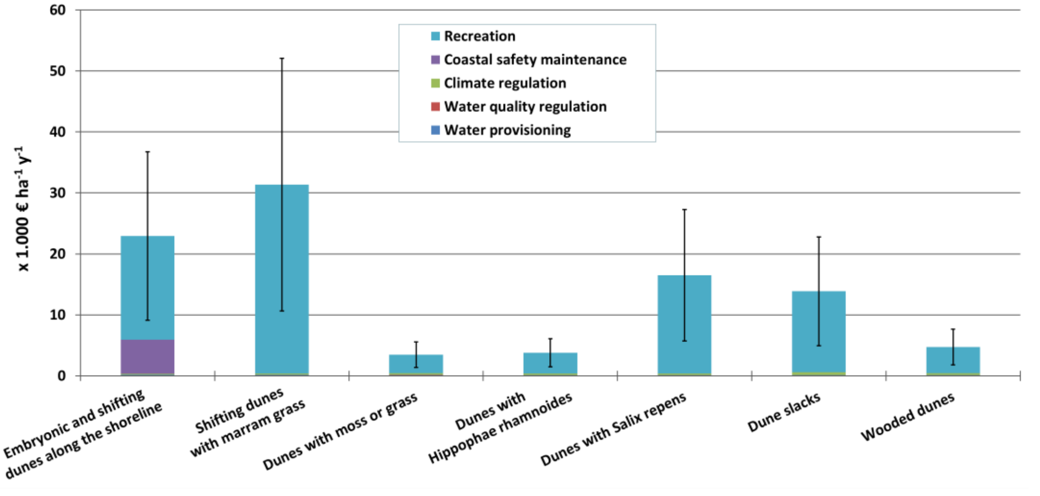

Figure 4.3 - Sum of the average economic value (x 1000 € ha^{-1} y^{-1}) of each dune habitat for 5 different ecosystem services. Error bars represent the sum of the minimum and the sum of the maximum estimates of the different ecosystem services per habitat type. Values for recreation do not include the correction factor on the increase in surface area of a particular habitat

Table 4.2 – Estimated ecosystem service delivery per habitat type. Negative values for water quality regulation reflect an enrichment of groundwater with nitrate. Values for recreation do not include the correction factor on the increase in surface area of a particular habitat.

		Embryonic and shifting dunes along the shoreline	**Shifting dunes with marram grass**	**Dunes with moss or grass**	**Dunes with *Hippophae rhamnoides***	**Dunes with *Salix repens***	**Dune slacks**	**Wooded dunes**
Water provisioning	$m^3\ ha^{-1}\ y^{-1}$	1281	1171	1008	830	784	705	749
	€ m^{-3}	0,075 (min) - 0,2 (max)	0,075 (min) - 0,2 (max)	0,075 (min) - 0,2 (max)	0,075 (min) - 0,2 (max)	0,075 (min) - 0,2 (max)	0,075 (min) - 0,2 (max)	0,075 (min) - 0,2 (max)
	€ $ha^{-1}\ y^{-1}$ [min]	96	88	78	62	59	53	56
	€ $ha^{-1}\ y^{-1}$ [max]	256	234	208	166	157	141	150
	€ $ha^{-1}\ y^{-1}$ [average]	176	161	143	114	108	97	103
Water quality regulation	kg N $ha^{-1}\ y^{-1}$	3,0	3,0	8,6	-29,7	0,5	0,5	7,0
	€ $(kgN)^{-1}$	0,6 (min) - 2,4 (max)	0,6 (min) - 2,4 (max)	0,6 (min) - 2,4 (max)	0,6 (min) - 2,4 (max)	0,6 (min) - 2,4 (max)	0,6 (min) - 2,4 (max)	0,6 (min) - 2,4 (max)
	€ $ha^{-1}\ y^{-1}$ [min]	2,0	2,0	5,7	-79,2	0,3	0,3	4,7
	€ $ha^{-1}\ y^{-1}$ [max]	7,9	7,9	23,0	-19,8	1,3	1,3	18,7
	€ $ha^{-1}\ y^{-1}$ [average]	5,0	5,0	14,4	-49,5	0,8	0,8	11,7
Climate regulation	ton C $ha^{-1}\ y^{-1}$	0,7	1,1	1,2	1,2	1,3	2,1	1,4
	€ $(tonC)^{-1}$	220	220	220	220	220	220	220
	€ $ha^{-1}\ y^{-1}$	154	233	223	260	278	465	313

		Embryonic and shifting dunes along the shoreline	**Shifting dunes with marram grass**	**Dunes with moss or grass**	**Dunes with *Hippophae rhamnoides***	**Dunes with *Salix repens***	**Dune slacks**	**Wooded dunes**
Coastal safety maintenance	$m^3\ ha^{-1}\ y^{-1}$	200 (min) - 500 (max)	0	0	0	0	0	0
	€ m^{-3}	16	16	16	16	16	16	16
	€ $ha^{-1}\ y^{-1}$ [min]	3200	0	0	0	0	0	0
	€ $ha^{-1}\ y^{-1}$ [max]	8000	0	0	0	0	0	0
	€ $ha^{-1}\ y^{-1}$ [average]	5600	0	0	0	0	0	0
Recreation (without correction on factor area)	# visits $ha^{-1}\ y^{-1}$	1492 (min) - 2487 (max)	1828 (min) - 3046 (max)	522 (min) - 870 (max)	424 (min) - 707 (max)	924 (min) - 1540 (max)	1801 (min) - 3002 (max)	493 (min) - 822 (max)
	€ (# visits)$^{-1}$	3 (min) - 9 (max)	3 (min) - 9 (max)	3 (min) - 9 (max)	3 (min) - 9 (max)	3 (min) - 9 (max)	3 (min) - 9 (max)	3 (min) - 9 (max)
	€ $ha^{-1}\ y^{-1}$ [min]	5664	10325	1011	1134	5367	4433	1436
	€ $ha^{-1}\ y^{-1}$ [max]	28320	51627	5055	5672	26833	22167	7178
	€ $ha^{-1}\ y^{-1}$ [average]	16992	30976	3033	3403	16100	13300	4307
SUM	€ $ha^{-1}\ y^{-1}$ [min]	9117	10649	1342	1427	5704	4952	1812
	€ $ha^{-1}\ y^{-1}$ [max]	36734	52097	5514	6068	27268	22773	7648
	€ $ha^{-1}\ y^{-1}$ [average]	22925	31373	3428	3748	16486	13862	4730

4.4.2. Ecosystem services of dynamic versus fixed dunes

The sum of the selected ecosystem services of the dynamic scenario (1.58 – 7.27 million € y-1 for the entire study area) is 54% higher than the sum of the fixed scenario (1.03 – 4.70 million € y-1 for the entire study area), (Figure 4.4). However, uncertainties on recreation and coastal safety maintenance, the two main ecosystem services, are large. The larger benefits for the dynamic scenario can be attributed to the greater extent of embryonic and shifting dunes in the front zone which are important for both coastal safety and recreation, and the presence of wet dune habitats and dunes with marram grass more inland where recreation is relatively high.

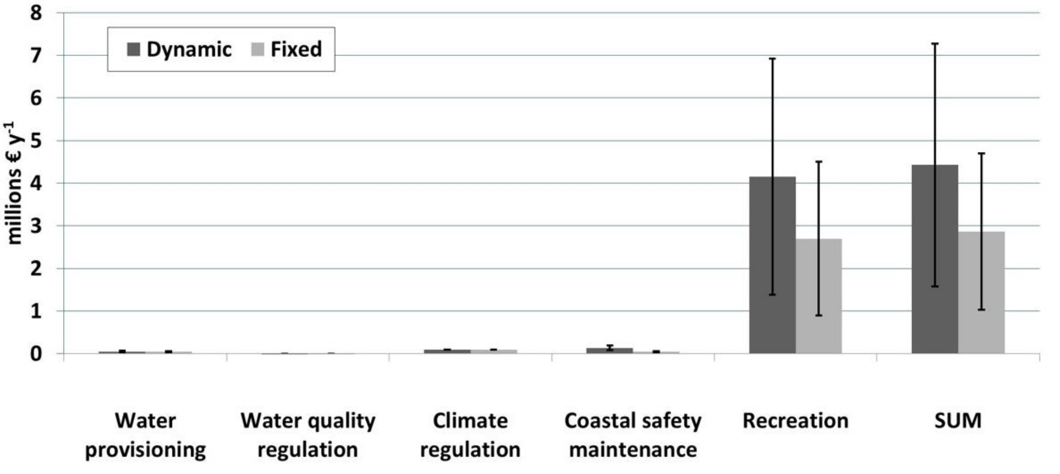

Figure 4.4 - Sum of the average economic value (x 1 million € y^{-1}) of the entire study area for 5 ecosystem services. Error bars represent the sum of the minimum and the sum of the maximum estimates. Values for recreation include the correction factor on the increase in surface area of a particular habitat.

The differences amongst the ecosystem services are even more pronounced when considering only the frontal dunes (Figure 4.5). Economic benefits in the dynamic scenario (0.18 – 0.71 million € y^{-1}) are double those in the fixed scenario (0.09 – 0.36 million € y^{-1}). While the habitats of the frontal dunes are important for coastal safety maintenance, they are negligible in terms of water provisioning, water quality regulation and climate regulation. Recreation and climate regulation become more important in the inner dunes (= the entire study area without the frontal zone, Figure 4.6) compared to the frontal dunes (Figure 4.5). Although climate regulation is rather small in most dune habitats (Figure 4.3) and seems negligible in comparison with recreation, benefits reach a total value of 0.10 million € y^{-1} in the dynamic and 0.09 million € y^{-1} in the fixed scenario. This is explained by the large extent of dune slacks (dynamic scenario) and the development of humus rich soils in woodland (fixed scenario).

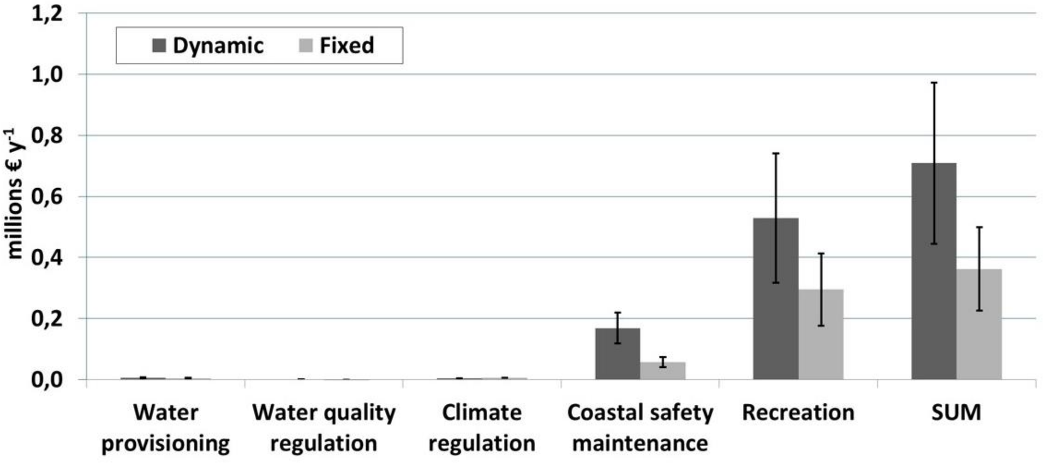

Figure 4.5 - Sum of the average economic value (x 1 million € y^{-1}) for 5 ecosystem services of the frontal dunes. Error bars represent the sum of the minimum and the sum of the maximum estimates. Values for recreation include the correction factor on the increase in surface area of a particular habitat.

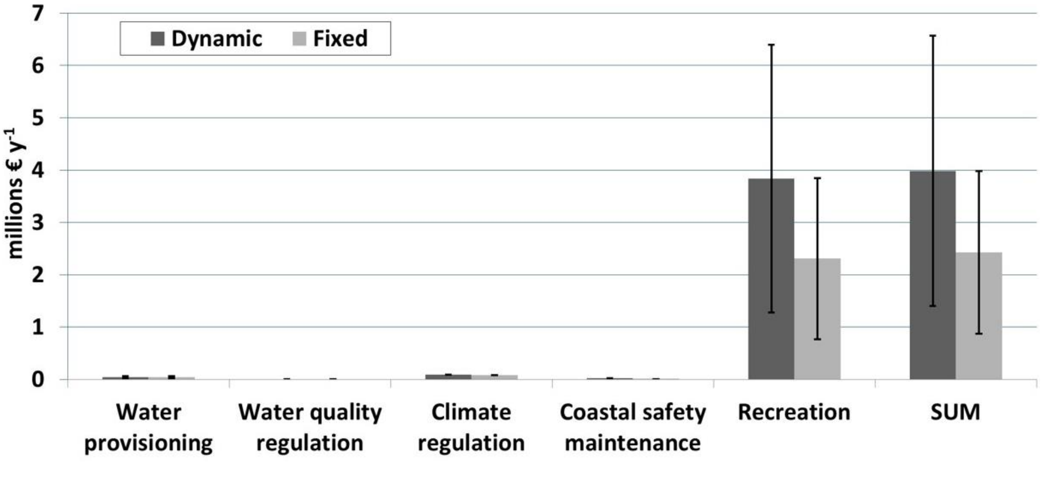

Figure 4.6 - Sum of the average economic value (x 1 million € y^{-1}) for 5 ecosystem services of the inner dunes. Error bars represent the sum of the minimum and the sum of the maximum estimates. Values for recreation include the correction factor on the increase in surface area of a particular habitat.

4.5. Discussion

4.5.1. Large benefits, few services

The data illustrates that recreation and coastal safety are two very important ecosystem services generated by dynamic dunes, providing important benefits for both inhabitants of the coastal zones and the entire region. Even if we account for the uncertainties and shortcomings for quantification and valuation of the different services studied, these services are more important compared to the other ecosystem services taken into account in this study.

The great difference between the value for coastal safety maintenance of marram grass dunes versus vegetation of stabilized dunes, is explained by the capacity of dynamic vegetation to withstand sand burial up to a meter per year. When sand is deposited, shoots rapidly grow through it and the new sediment is held with the roots. Vegetation of fixed dunes is not adapted to large amounts of sand burial and will die off. Woody vegetation is also more prone to erosion as stems are less flexible and not able to lie flat during high winds as grasses can (Feagin et al. 2015).

Water quality regulation and climate regulation are typically associated with well-developed soils (accumulation of organic matter), thus low in young deposits along the shoreline and increasingly important as dunes grow older. Even in fully developed soils of wooded dunes they however remain relatively small in comparison with inland habitats. This is related to the coarse texture of dune sand which – in comparison with soils containing a fraction of clay – generally has lower capacity to retain organic matter and nutrients, and higher decomposition rates because of the low resistance for air penetration (Wardle 1992). The average carbon sequestration rate for northern Belgium is 1409 kg C ha^{-1} y^{-1} in the upper 1 m of the soil (Ottoy et al. 2015), while dune grasslands and shrubs vary between 1000 and 1250 kg C ha^{-1} y^{-1}. The wetter habitats of dune slacks form an exception to this (2113 kg C ha^{-1} y^{-1}) as they have a much higher biomass production and prevent breakdown of organic matter due to permanent wet soils.

The low value for water provisioning is explained by the gradual decrease of the water extraction rate over the last decennium and the low market price of water. The decrease in water extraction was needed to prevent salt water intrusion, but is also part of a policy to reduce the ecological impact of drinking water production (Vandenbohede et al. 2009).

The high value for recreation reflects the proximity to a densely populated and rich region, with a short coastline with highly developed touristic infrastructure and limited area of dunes. It confirms that the study area is an important area for recreation and tourism in Flanders, and the estimated number of visits is in line with these of other important areas for recreation with a similar size (Staes et al. 2017). The higher recreational value that was obtained for habitats associated with dynamic dunes (shifting dunes with marram grass and dune slacks) corresponds well with the study of De

Nocker et al. (2015) in which 400 visitors of the nature reserve where asked to score the attractiveness of dune habitats based on different sets of photographs. Dune slacks and dunes with marram grass and bare sand were found to be most attractive because of the variety of species and colors and the presence of water (dune slacks) and because they are characteristic for the region (shifting dunes). Lowest scores were attributed to shrub, which visitors found rather monotonous. The relative value of forest is less than the value found by De Nocker et al. (2015). The woodland in the study area is mostly species-poor plantation of mainly populous species, while the woodland in the study of De Nocker et al. (2015) is species-rich oak forest. This may result in an underestimation of the value of a dune forest which evolves from natural succession. Uncertainty also exists on the relationship between surface area of attractive habitat and number of visits and the correction factor that was applied. In spite of this uncertainty, the results clearly demonstrate the importance of safeguarding habitats of young successional stages and preventing excessive shrub encroachment to maintain its high recreational value.

As explained above, the five ecosystem services capture the most important ones (based on total monetary value) but not all of them. Some of these provide a minor economic value in the protected nature reserve like wood production (coppice derived from nature management) and meat production (extensive grazing as nature management). Important regulatory services that are not included are the capture of air pollutants by vegetation and related impacts on public health, and impacts from nearby natural areas on real estate values or public health. These services are important for the area as a whole but not enough information was available to be able to make a clear distinction on the contribution of specific vegetation or habitat types.

The protected status also excludes ecosystem services such as hunting or military use, which are common practices in many coastal dunes in Europe (Everard et al. 2010) and may increase the economic value of certain dune habitats. The value for water production is also relatively low compared to other dune regions. An increase in the rate of water extraction would however reduce the value of other ecosystem services and especially of the conservation value. Dune slacks, for example, would desiccate and turn into drier vegetation types which are found less attractive by visitors (De Nocker et al. 2015). Recreation is an important service from the dunes, but it is also a potential threat, both in terms of loss of dune areas to build tourist infrastructure as in terms of managing visits within the protected area.

An additional underestimation of the value of dunes results from not taking into account the negative effects of climate change on the other ecosystem services besides coastal protection. As sea level rises, the frontal dunes will gradually erode, thus reducing the total surface area of dunes, leading to a reduction of the space available for ecosystem services (e.g. reduction of the volume of the groundwater reserve, less space for recreation and carbon sequestration, …). This effect will be less pronounced in dynamic dunes which are able to grow with sea level rise and compensate for the erosion by migrating inland, given that dune migration is not inhibited by the presence of infrastructure, agriculture, etc.

4.5.2. Dynamic dunes generate more benefits

The results indicate that a dynamic dune complex with a large proportion of young successional stages is better to deliver the two single most important ecosystem services for the study area than a fixed dune system. The high value attributed to the characteristic habitats of dynamic dunes is in line with our overall understanding of the relative importance of uniqueness in attracting tourists (Lyon et al. 2011; De Valck et al. 2016). As dynamic dunes improve diversity of less familiar habitats, they are likely to attract more visitors.

The difference between dynamic and fixed dunes is most distinctive for the shoreline dunes, where coastal safety maintenance plays an important role. When focusing only on the frontal zone, redynamisation may increase the economic benefits up to nearly two times the value compared to the fixed scenario. To underpin this conclusion, a comparison is made with the investment costs for restoration of dynamic dunes. The Life Dunes project (Zevenberg and Zijlstra 2012), which aimed to restore 4700 ha of dunes along the Dutch coast, invested a total of 4.7 million € for measures to rejuvenate dunes, restore dune dynamics, combat grass encroachment and restore biodiversity. In case similar measures would be taken to restore natural dynamics in the study area (340 ha), it can be expected that this would require an investment of 0.38 million € (8% of 4.7 million €). Our assessment of the potential benefits of a dynamic dune system (Figure 4.4) suggests a very good benefit/cost ratio for this type of investment, even if we account for all uncertainties in the assessment. However, it is important to note that the benefits are not strictly for those who bear the investment costs. The total benefits calculated with the ecosystem services assessment are to be seen as benefits to society in general (direct or indirect financial benefits by e.g. creating health benefits or avoiding damage costs). Depending on the service, benefits can be assigned to specific stakeholders (e.g. recreation). Depending on the aim of the ecosystem services assessment, disaggregation of the costs and benefits for different stakeholders can be required to avoid hidden social inequality (Krutilla 2005). However, public investments such as Life projects have a broader aim to create societal benefits. In this case, calculating an aggregated overall benefit is informative to communicate about the overall importance of such projects.

Dynamic dunes along erosive coasts gradually lose the frontal dunes as a result of progressive erosion. Parts of the mobilized sand will be blown into the inner dunes, turning them into dynamic systems and allowing the dunes to migrate transgressively inland (Arens et al. 2010). If no space is available at the inner margin of the dunes (e.g. due to the presence of houses), additional measures such as beach nourishments are needed to guarantee coastal safety and allow for redynamisation. In order to complete the balance of benefits versus costs, these expenses should also be taken into account. Nourishments, however, in certain cases are also needed to reduce the impact of waves on dykes and prevent dyke instability during extreme storms (Verwaest et al. 2009).

4.5.3. Dynamic dunes generate benefits for biodiversity

Despite the higher economic benefits of dynamic dunes versus fixed dunes, this study does not advocate to convert all dunes to highly dynamic systems dominated by marram grass and dune slacks. One of the characteristics of dune landscapes is the extremely high diversity of habitats and species occurring in a small area (Brunbjerg et al. 2015). Sand deposition and erosion regularly disturb the equilibrium between abiotic and biotic conditions which is being reached as succession progresses. This increases the sharpness of multiple ecological gradients, such as topography, groundwater depth, lime content, pH, salinity and slope aspect (Pye et al. 2007b; Everard et al. 2010). Decreasing disturbance by sand dynamics leads to less pronounced gradients, reduction of habitat heterogeneity and consequent loss of (dune-specific) species (Bonte and Hoffmann 2005; Everard et al. 2010; Brunbjerg et al. 2014). This research does not specifically take into account these effects of (gamma-)diversity on ecosystem service delivery. Although literature exists that links structural complexity to landscape aesthetics and to number of visits (Harrison et al. 2014; De Nocker et al. 2015), it is a challenge to express in monetary terms its exact contribution in attracting people. A drawback of applying an economic valuation on ecosystems is that it does not fully account for the intrinsic value of biodiversity and the impact of species diversity on ecosystem services (TEEB 2010). The results of this study nevertheless are useful to underpin the societal importance of remobilizing dunes along the shoreline as described in earlier analyses (Pye et al. 2007b; Everard et al. 2010; Arens et al. 2013), and advocate to re-expose front dunes to natural hydrodynamic and eolian forces wherever possible (Pye et al. 2007a). In this case, this drawback is even irrelevant, since both a biodiversity and an ecosystem service approach benefit from an increase in dynamics (Howe et al. 2010; Provoost et al. 2014).

4.5.4. Benefits are site-specific

The values of the ecosystem services in this study are a function of the supply and the societal demand in the present day situation. The benefits for drinking water provision for example are low in all habitats because of the reduced water extraction. A similar exercise in another study area may generate other results and differences between habitats may appear stronger.

Although the results show that redynamisation of dunes may generate important economic benefits, this conclusion cannot be generalized and transferred to other study sites. This research was conducted in an environment where the dunes are wide and high enough to prevent flooding during an extreme storm and given a sea level rise of + 60 cm (Van der Biest et al. 2009). The method for calculating coastal safety maintenance is also based on the assumption that a net transport of sand from beach to dunes takes place. Where the shoreline is naturally subject to long-term erosion and relatively narrow, additional measures (e.g. nourishments) would be needed to compensate the negative sand budget (Arens et al. 2010) and prevent flooding. It is recommended to perform a detailed and site-specific evaluation of the effects of removing structures to remobilize dunes on flood risks and other ecosystem services (Arens et al. 2013; Lithgow et al. 2013; Nordstrom 2014; Ruckelshaus et al. 2016), and to compare with true benefits. Alternatives should be sought which allow remobilization of dunes so that, on the long term and with rising sea level, they continue to form a natural protection against floods. One of the main challenges in highly urbanized coastal zones is the lack of space to let dunes migrate inland, or to allow for managed retreat (Nordstrom et al. 2015). Recent advances in nature-based engineering seem promising (Stive et al. 2013; Arens et al. 2013; Temmerman et al. 2015), but are not always viable (Narayan et al. 2016). Redynamisation of dunes requires space for dune to migrate. The most urgent needs to protect against floods are generally along settlements where buildings are constructed near the shoreline and no space is available between sea and buildings. Nature-based adaptation not only requires engineering solutions but also increasing flexibility and adaptation by decision-makers and coastal users. Adaptation measures might include the abandoning of existing activities or structures that are low-profitable or that lie in areas with important natural relics, or seaward expansion of the coastline.

4.6. Conclusions

Dunes create substantially more ecosystem services when sand transport between sea, beach and dunes is not hampered by artificial hard structures such as dikes and groins. Especially coastal safety maintenance and recreation depend on the constant supply and movement of sand and decline when dune vegetation evolves into shrub. The higher value of young dune vegetation compared to older successional stages is related to the capacity of plants of early stages to withstand sand burial and accumulate sand, and, for recreation, to the visitor appraisal of habitat types characteristic of dynamic dunes (shifting dunes and wet dune habitats). Climate regulation by carbon sequestration becomes more important as dunes get stabilized when sand transport diminishes, resulting from soil development under dense vegetation cover. The ecosystem service nevertheless remains relatively low in comparison with inland habitats due to the coarse texture of dune sand. This research underpins earlier descriptive studies on the ecosystem services and their societal benefits of dynamic dunes with quantitative information. Such information may help to convince decision-makers to restore natural dune dynamics wherever possible.

4.7. References

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5

A process-based approach to align biodiversity and ecosystem services in spatial planning

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Focus on processes to plan for biodiversity conservation and ecosystem services. *In preparation*.

5.1. Abstract

Although the consideration of socio-economic demands along with biodiversity conservation is now high on the environmental policy agenda, it is not yet standard practice. This is argued to be related, amongst others, to an insufficient consideration of multiple values from the earliest stages in planning processes on. Another cause is the lack of awareness among stakeholders and practitioners of the underpinning role of ecosystem functioning and biodiversity to support human well-being. Meanwhile, there is mounting critique on the absolute focus of biodiversity conservation on static properties such as species and habitats. The establishment of more ecologically sensible objectives that include ecosystem processes besides species and habitats is put forward as a more effective way of environmental conservation. Methodological approaches increasingly consider ecosystem processes. However, the processes that are included mostly relate to aspects of biodiversity such as connectivity and productivity, and rarely do they include the biotic and abiotic mechanisms that underlie biodiversity. We here report on the development of a method that integrates three principles which we identify as key to advance the integration of ecosystem services with biodiversity conservation in planning practice: (1) include the variety of ecosystem processes that underlie biodiversity and production of ecosystem services, (2) link the ecosystem processes to biodiversity and to socio-economic benefits to identify the common ground between conservation and socio-economic demands and (3) incorporate multiple benefits and involve stakeholders early in planning processes. The main aim of this chapter is to demonstrate how including processes opens opportunities to align biodiversity and ecosystem services in early planning phases and to provide inspiration to advance current approaches for planning for biodiversity and ecosystem services.

5.2. Introduction

Given the fast growth of the world population, safeguarding the necessary space to protect biodiversity and ensuring natural processes is a major challenge worldwide for spatial planning both on land and at sea. Over the past decades, different concepts have been established that aim to find compatibilities between nature conservation and socio-economic development. The ecosystem approach (CBD 2004), marine spatial planning and ecosystem-based management (McLeod et al. 2005) all focus on combining biodiversity conservation and sustainable and equitable use rather than on isolated, sectoral objectives such as individual species or economic benefits. In recent decades, the notion of ecosystem services, which connects aspects of ecosystem functioning to human well-being and underlines the dependency of humans on ecosystems, gained a lot of attention. Highlights are the publications of the Millennium Ecosystem Assessment (MEA) in 2005 and The Economics of Ecosystems and Biodiversity (TEEB) in 2010, and the foundation of the Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES) in 2012.

Although the integration of biodiversity with ecosystem services is now high on the research and policy agenda (Douvere et al. 2007; Agardy et al. 2011), it is still not common in practice. A wide range of decision-support tools exists that allow to spatially allocate resources to realize environmental objectives while taking into account societal demands (e.g. Marxan with Zones, Zonation). However, there is a lack of considering multiple values from the earliest stages of planning processes on (Bennett et al. 2017; Guerrero et al. 2017). It is also argued that, in spite of the merits of publications such as the MEA and TEEB, there is still often insufficient information (Ortiz-Lozano et al. 2017) and awareness (Guerry et al. 2015) among stakeholders and in spatial planning practice of the underpinning role of ecosystem functioning and biodiversity to support human well-being. There is thus a need for approaches that support the development of strategic objectives based on an integration of stakeholders' preferences with knowledge on ecosystem functioning.

Biodiversity conservation has long focused on the preservation of individual species (assemblages) and habitats (Jepson 2016). Ecosystems, however, evolve through biophysical interactions and complex ecological processes taking place on spatial and temporal scales beyond the boundaries of a single habitat. It is increasingly recognized that conservation efforts are more successful if also ecological processes are considered (Klein et al. 2009; Bennett A.F. et al. 2009; Pettorelli et al. 2018) and international standards are now being adapted to include them (Watson et al. 2016). However, existing approaches generally take only a limited number of processes into account, and these are most often related to biodiversity aspects such as connectivity and succession (Tulloch et al. 2016). Very few papers are found that also consider abiotic processes (e.g. sand transport) and interactions between biotic and abiotic components (see e.g. Edwards et al. 2010) that, together with biotic mechanisms, shape an environment supporting biodiversity. The inclusion of the variety of natural processes, both biotic and abiotic, is also stated by Ockendon et al. (2018) as an essential progress towards improved landscape restoration. Especially when linking to ecosystem services the role of including the variety of ecosystem processes becomes even more obvious as they are the driving mechanisms for these benefits (Haines-Young and Potschin 2010). Research in management of ecosystem services shows that decision-making based solely on structural properties such as land use and habitat can result in strongly adverse effects (Van der Biest et al. 2015) and calls for a consideration of ecosystem processes (Kremen et al. 2005; Nicholson et al. 2009; Rieb et al. 2017). Processes thus constitute a key factor in managing landscapes for both biodiversity and ecosystem services optimization.

We here report on the development of a method that integrates three principles which we identify as key to advance the integration of ecosystem services with biodiversity conservation in planning practice: (1) include the variety of ecosystem processes that underlie biodiversity and production of ecosystem services, (2) link the ecosystem processes to biodiversity and to socio-economic benefits to identify the common ground between conservation and socio-economic demands and (3) incorporate multiple benefits and involve stakeholders early in planning processes. The method is not aimed at supporting spatial allocation of areas to enhance ecosystem processes such as in structured conservation planning, but rather to guide in the creation of a shared vision in early stages of planning processes and to identify objectives for environmental management. We illustrate its use in light of the development of a strategic vision for the Belgian coastal ecosystem which is an intensively used area with high pressures on remaining important biodiversity values.

5.3. Methodology

5.3.1. Generic methodology

Central in the approach is the focus on ecosystem functioning as the motor of a healthy ecosystem. A well-functioning ecosystem can be defined as a system which has the ability to maintain its structure and processes over time in the face of external stress (CBD 2004). Ecosystems are characterized by structural properties and shaped by underlying processes that allow them to adapt to changes. Ecosystem processes are here defined as changes in the stocks or in the fluxes of products and energy resulting from interactions among organisms (incl. humans) and with their abiotic environment, and from abiotic driving forces. Ecosystems consist of different habitats, which the Convention of Biodiversity defines as "essential to the concept of biodiversity conservation, where the aim is to conserve natural habitats supporting the preservation of the ecological processes which underpin ecosystem function". Ecosystem services likewise result from structural characteristics and underlying ecological processes that form these structures (Haines-Young & Potschin 2010). As processes are the drivers of both biodiversity and ecosystem services (Nicholson et al. 2009), they enable to integrate objectives for biodiversity and for ecosystem services. This is the key rationale of the proposed methodology which is described in a stepwise procedure (Figure 5.1).

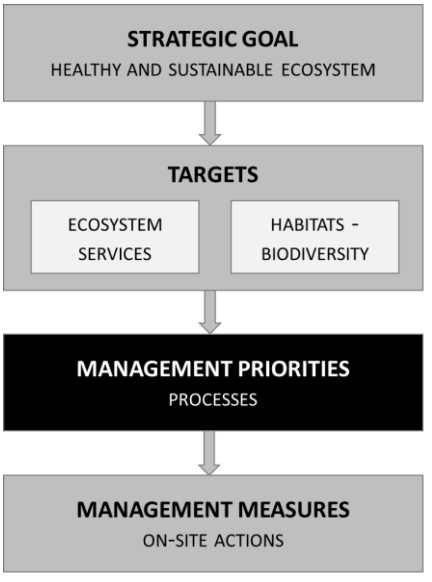

Figure 5.1 – Schematized overview of the rationale of the proposed methodology.

Step 1: Set term and identify external drivers of change

The first step consists of setting the time scale by which the strategic goal and objectives should be achieved and identifying the external drivers of change. External drivers of change refer to processes taking place on large temporal and spatial scales beyond the boundaries of the ecosystem under consideration, and which are difficult to control by governance only on the local scale and within the established term. Examples are demographic growth and climate change. Both the targeted time frame and the drivers of change will influence future socio-economic demands (Step 2) and the capacity of the ecosystem to provide certain ecosystem services and to develop habitats and maintain biodiversity goals.

Step 2: Identify habitat and ecosystem services targets

In a second step, the habitats and relevant ecosystem services that the ecosystem should provide on the defined term are identified. Habitats can be identified based on biodiversity targets of conservation frameworks (e.g. local conservation objectives; NATURA2000 in Europe) for which they provide opportunities. These include habitats occurring naturally in the ecosystem and habitats that are expected to occur in the future, for example because of active management or environmental changes. The scale on which habitats are defined should be such that variable effects of processes between habitats (see Step 3) can be distinguished. If a process has different effects depending on the habitat under consideration, it is recommended to consider these as separate habitats.

Relevant ecosystem services are selected based on the capacity of the particular ecosystem and its habitats to provide certain ecosystem services and based on socio-economic demands. As the aim of the method is to develop a strategic vision for the future, it is important not only to consider today's capacity and demand for ecosystem services, but also future potentials and needs which may alter under the external drivers of change identified in Step 1. An ecosystem service is considered to be relevant if its economic or social value is (expected to become) high, or if it is specific to the ecosystem (e.g. fisheries production in marine ecosystems). The involvement of stakeholders in the identification of relevant ecosystem services is needed (Reyers et al. 2009; Keune et al. 2015).

Step 3: Prioritize ecosystem services and habitats

Next, the ecosystem services are given a weight for their anticipated demand in the ecosystem within the defined time frame and taking into account the external drivers of change (Step 1). A variety of methods exists to assess socio-economic priorities, but stakeholder involvement is strongly recommended (Keune et al. 2015). A key criterion for effective stakeholder engagement is to have a balanced representation of all relevant stakeholders (Durham et al. 2014). This is especially important when involving them in stages of the planning process with a potentially strong impact on decision-making, as is the case in this methodology.

Depending on local conditions and on the goal of application of the method, habitats can be considered equally important or they can also be attributed a weight. Certain habitats may be less desirable because of its dominance within the area, or because they hold low biodiversity values. A weight can for example be attributed based on the biological value of the habitat (number of (rare) species, particular species, etc.), its desired surface area, etc..

The weighing of ecosystem services and habitats will be used in a later step to identify the strength of trade-offs, conflicts and synergies among ecosystem services and habitats, and eventually to prioritize ecosystem processes.

Step 4: Describe ecosystem processes

For each habitat and ecosystem service, the processes are identified that contribute to their development, maintenance or delivery. Natural and anthropogenic processes are taken into account as they both affect ecosystem functioning. Natural processes are essential for the development and the functioning of a sustainable ecosystem and the production of the selected ecosystem services. Anthropogenic processes also have an impact on ecosystem functioning (positive or negative), but they are, in contrast to natural processes, not essential for the development and maintenance of a sustainable ecosystem. In view of the development of a strategic vision, only those processes should be included that have a signification contribution to or impact on the identified habitats and ecosystem services, and that do not fall under external drivers of change (Step 1). This step should be based on the best available knowledge on ecosystem functioning and on how processes and feedbacks result in the development of habitats and ecosystem services, including processes taking place on longer time and spatial scales. This requires the involvement of experts from different disciplines, both natural (biodiversity, water quality, hydrodynamics, habitat types, ...), as socio-economic sciences. A score is assigned that expresses the magnitude and direction of the impact of a process on the occurrence and the quality of a habitat or the provision of an ecosystem service, called the impact score. This is preferably based on quantitative information such as derived from models or measurements, or expert judgment when no quantitative data is available. Contrasting effects of processes (positive and negative effects on a habitat or ecosystem services) should be avoided as much as possible, to prevent loss of information when combining them in a single score. This can be done in several manners: 1) Divide into narrower defined habitats when parts of react differently to disturbance or provide different ecosystem services (e.g. tidal areas into vegetated tidal marshes and non-vegetated tidal flats). 2) Specify parts of ecosystem services that have different effects on different habitats (e.g. divide fishing production into benthic and pelagic fisheries). 3) Specify processes to more detail when the general process is important for different and/or conflicting reasons in habitats (e.g. divide emissions into methane production and nitrous oxide production). Alternatively, positive and negative effects can be weighed against each other resulting in a single overall score that takes differences into account. Uncertain processes regarding effect sizes are either merely identified, but not included, the range of the expected effects can be provided or the weight of the expected effect can be corrected based on its probability.

Step 5: Recommendations for ecosystem processes

The impact score of each process on both habitats and ecosystem services (Step 4) is multiplied with the priority score for the habitat or ecosystem service (Step 3). The sum of these multiplications is calculated per process for the habitats and for the ecosystem service, resulting in a (weighted) sum for habitats and a (weighted) sum for ecosystem services for each process.

Both sums can be plotted relative to each other in an XY-diagram (Figure 5.2), allowing to identify conflicts, synergies and trade-offs between biodiversity conservation and societal demands. Processes in the upper right corner create important benefits for multiple ecosystem services and habitats. In the lower left corner are processes with strong negative impacts on ecosystem services and on habitats. The graph thus shows a trend of multi-functionality from the bottom left to the top right (arrow). Trade-offs can result from (1) opposite effects of a process on multiple ecosystem services or habitats, (2) a strong impact on one or on a few ecosystem services or habitats or (3) a moderate impact on one or on a few ecosystem services or habitats with high demand (high priority score).

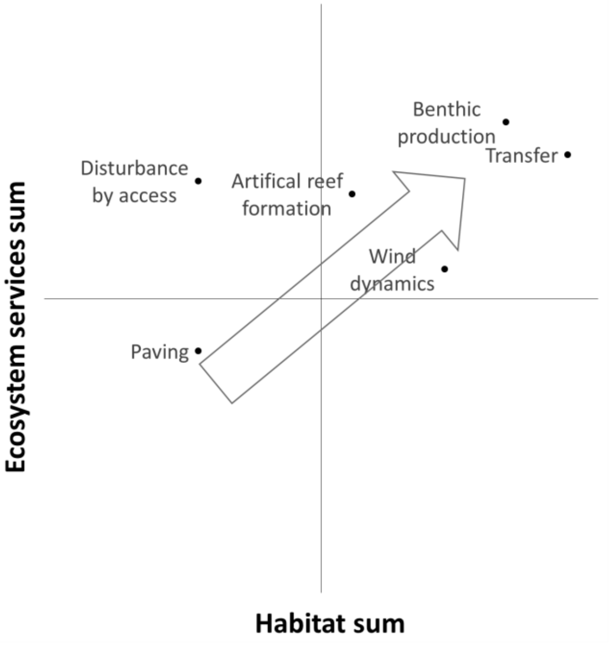

Figure 5.2 – Hypothetical example of Step 5 of the proposed method. Arrow: degree of multi-functionality.

Table 5.1 – Hypothetical example of Step 5 of the proposed method. Calculation of (weighted) sum of habitats and ecosystem services per process.

	Habitats				Ecosystem services			
	Pelagic	**Tidal marshes**	**Dune foot**	**(Weighted) sum**	**Fisheries**	**Climate regulation**	**Recreation**	**(Weighted) sum**
Priority score	**10**	**10**	**10**		**7**	**4**	**9**	
Ecological processes								
Benthic production	+1	+2	0	**+30**	+2	+1	+1	**+27**
Transfer	+2	+2	0	**+40**	+2	+2	0	**+22**
Wind dynamics	0	0	+2	**+20**	0	0	+0.5	**+4.5**
Anthropogenic processes								
Artificial reef formation	+0.5	0	0	**+5**	+1	0	+1	**+16**
Disturbance by acces	0	0	-2	**-20**	0	0	+2	**+18**
Paving	0	0	-2	**-20**	0	-2	0	**-8**

5.3.2. Illustration of the methodology for the Belgian coast

To illustrate the functionalities of the methodology, we here elaborate upon the development of a strategic plan for the Belgian coastal ecosystem in light of which the methodology is developed. The terrestrial limit corresponds with the transition from polder to dunes, and the marine limit coincides with the boundary of the Belgian Continental Shelf. The land part (80 km²) is dominated by dunes under a protected status as well as degraded dunes used as pasture or private gardens and comprises two areas with tidal flats and marshes. The marine zone (3600 km²) is part of the Southern North Sea and the seafloor is mainly made up of soft sediments with a series of parallel sand banks hosting a high benthic diversity as a result of the highly variable topography and sediment composition (Vanden Eede et al. 2014). Densely urbanized areas are left out from the study as management of open space is the main purpose of the application in the case-study. The relatively small size and high population density create intensively used land- and the seascapes and jeopardize remaining biodiversity values. Several developments are taking place which will further increase spatial claims or change the ecosystem such as blue growth initiatives (e.g. aquaculture, marine biotechnology, ...), harbor developments and the Flemish Bays masterplan (Projectgroep Vlaamse Baaien 2012), urging the creation of a shared vision for a sustainable ecosystem among all sectors.

Step 1: Set term and identify external drivers of change

As protection against floods is a major challenge along the Belgian coastal zone and continuous sea level rise urges long-term solutions, it was opted to set the time scale at 2100, which corresponds to the long-term climate change scenario of the Intergovernmental Panel on Climate Change (IPCC 2014).

Following external drivers of change were identified: (1) effects of climate change related to more winter rainfall, warmer summers, ocean acidification due to increased CO_2-uptake (Van der Aa et al. 2015) and sea level rise; and (2) demographic growth (FPB-FOD 2015). Although an increase in population size in the coastal zone is expected, the spatial demand for housing is considered not to increase because of restrictions related to building in dune areas and a tendency to urban infill in Flanders.

Step 2: Identify habitat and ecosystem services targets

The identification of habitats was largely based on the NATURA2000 habitat types and the European habitat classification EUNIS which distinguishes in more detail marine habitats. Twelve habitats were identified (Table 5.2) of which distribution and total surface area were derived from monitoring data and existing cartographic information (Van der Biest et al. 2017b). Artificial marine structures (jetties, ship wrecks, groynes, wind turbine foundations, ...) were included because of their ubiquity, potential ecological values (Perkol-Finkel et al. 2012), distinct ecological functions and ecosystem services they may facilitate (Wetzel et al. 2014). A large differentiation was applied to dune ecosystems in which processes related to sand dynamics and soil development strongly influence species assemblages (Brunbjerg et al. 2015) and ecosystem services (Van der Biest et al. 2017a).

Relevant ecosystem services were identified using the Common International Classification of Ecosystem Services CICES v4.3 (EEA 2016) as reference framework. Additionally, marine-specific ecosystem services that were not included in CICES were selected from the marine typology of ecosystem services of Böhnke-Henrichs et al. (2013). An initial selection of the most relevant ecosystem services was made based on the expected demand now and by 2100. From this list, the ecosystem services whose consumption does not threaten ecosystem functioning and sustainability were not retained (e.g. several cultural ecosystem services such as spiritual value and health benefits), since the overall aim is a strategic vision for a healthy and sustainable ecosystem. This resulted in a list of 8 ecosystem services, of which 4 provisioning ecosystem services (agricultural production, fisheries production, aquaculture production, drinking water provisioning), 3 regulating ecosystem services (flood protection, climate regulation, water quality regulation) and 1 cultural ecosystem services (recreation). This preliminary list was proposed to a multidisciplinary group of experts (described in Appendix F Table F.2) who added 2 provisioning ecosystem services (renewable energy production and sediment extraction).

Table 5.2 – Habitats identified in the development of a strategic plan for the Belgian coastal ecosystem, with indication of their approximated total surface area in the Belgian coastal zone (km²) and cartographic source.

Habitat type	**Description**	**Surface area (km²)**
Pelagic	The water column of the Belgian part of the North Sea	-
Gravel beds	Accumulation of loose grind and pebbles at the edge of a sand bank	max. 526.2
Submerged sandbanks and foreshore	Permanently submerged sandbanks at variable depths	524.8
Tidal flats and marshes	Habitats of fine sediment in the tidal zone above low tide and below spring tide, ranging from bare flats to densely vegetated on the least frequently flooded parts	1.3
(Artificial) reefs	Biogenic reefs formed by dense concentrations of the sand mason worm *Lanice concilega*, or fouling communities on permanently submerged artificial hard substrata	141.4
Estuary	Downstream part of a river that discharges in the sea and is subject to tidal forces and characterized by a salt gradient, including tidal flats and marshes and sand banks with varying salt gradient	0.4
Lower beach and emerged sand banks	Sand banks above low tide and below high tide, including beaches	1.7
Upper beach and dune foot	Part of the beach above high tide where vegetation starts to develop + embryonic dunes	1.2
White dunes	Young, dynamic dunes dominated by dune building species such as marram grass	3.1

Habitat type	**Description**	**Surface area (km²)**
Grey dunes – herbaceous	Dunes fixed by moss or grass, with reduced sand dynamics and increasing soil development	5.8
Grey dunes – shrub	Older dunes fixed by shrub and woodland, with important soil development	9.1
Dune slacks	Depressions in the dune landscape which are temporarily or permanently flooded by fresh water	0.9

Step 3: Prioritize ecosystem services and habitats

A group of stakeholders (described in Appendix F Table F.3) was invited to individually give a score of 1 (not important) to 10 (extremely important) to each ecosystem service, reflecting what they believe are the socio-economic benefits the coastal ecosystem will need to provide by 2100. Respondents were asked to take into account the effects of climate change and population growth (Step 1). The final score per ecosystem service is calculated as the average of all respondents and resulted in following scores: flood protection [10], recreation [9], water quality regulation [8], renewable energy from wind [8], fisheries production [7], sediment extraction [6], drinking water provisioning [5], aquaculture production [3], climate regulation [3] and agricultural production [1]. With the exception of tourism and renewable energy from wind, all provisioning ecosystem services received a low to moderate score. The low score for agricultural production can be attributed to underrepresentation of the sector in the stakeholder group, but also to the limited agricultural activities and potential within the study area related to the low efficiency of dune soils for most crops and the nearby presence of highly fertile ground in the polders. Except for climate regulation, regulating ecosystem services received high scores which can be explained by their supporting role in the provision of other ecosystem services with high socio-economic demands. Water quality regulation for example reduces risks of algal blooms with important negative effects on tourisms which received a high weight. Flood protection and recreation were found to be the most important ecosystem services of the coastal zone.

Since the overall strategic goal is a healthy ecosystem, the different habitats are considered equally important (weight [10]).

Step 4: Describe ecosystem processes

The Belgian coastal zone is one of the most studied and monitored coastal areas in the world. This step is based on an extensive study of the available knowledge on ecological functioning (see Van der Biest et al. 2017b), and expert judgment when data was lacking. The impacts of processes (see Appendix F Table F.1 for a description of processes) on habitats and ecosystem services were synthesized in an impact matrix (Table 5.3). To each process (rows), a score with a numeric value (Table 5.4) is assigned that expresses its contribution to or impact on a habitat or on an ecosystem service (columns). This score is either derived from quantitative data, or using expert judgment. The matrix was first completed by the project partners and afterwards reviewed and adjusted by the group of experts (Appendix F Table F.2).

Processes with multiple and contrasting effects were given a score +/- with a numeric value of 0 (positive and negative effects are expected to be equally large), or the effects were weighed against each other resulting in a single overall score which takes the differences into account. Uncertain relationships were included by attributing the lowest possible score for the anticipated direction of the influence (± 0.5).

Table 5.3 – Impact matrix of the impacts of processes on habitats and ecosystem services (see Appendix F Table F.1 for a description) in the Belgian coastal zone. Dark blue: marine processes, brown: terrestrial processes, light blue: processes taking place at sea and on land.

		HABITATS													ECOSYSTEM SERVICES										
		Pelagic	Gravel beds	Tidal flats and marshes	(Artifical) reefs	Submerged sandbanks and foreshore	Estuary	Lower beach and emerged sand banks	Upper beach and dune foot	White dunes	Grey dunes - herbaceous	Grey dunes - shrub	Dune slacks	HABITAT SUM	Agricultural production	Fisheries production	Aquaculture production	Sediment extraction	Drinking water provisioning	Flood protection	Climate regulation	Water quality regulation	Wind energy	Recreation and tourism	ECOSYSTEM SERVICES SUM
	Priority score	-	-	-	-	-	-	-	-	-	-	-	-		***1***	***7***	***3***	***6***	***5***	***10***	***3***	***8***	***8***	***9***	
ECOLOGICAL PROCESSES	Hydrodynamics (HD)	++	+	+	+	+	++	+	-	-	0	0	0	**7**	0	++	++	0	0	--	0	+	++	0	**24**
	Morphodynamics (MD)	0	-	++	+	++	-/+	++	0	0	0	0	0	**6**	0	-/+	-/0	+	0	+	0	0	0	0	**14.5**
	Ecological engineering (EE)	+	++	0	++	++	+	0	0	0	0	0	0	**10**	0	++	+	-/0	0	+	+	++	0	0	**43**
	Benthic production (BeP)	+	++	++	++	++	++	++	0	0	0	0	0	**13**	0	++	0	0	0	0	+	+	0	+	**34**
	Pelagic production (PeP)	++	++	+	+	+	++	+	0	0	0	0	0	**10**	0	++	++	0	0	0	+	+	0	-/+	**31**
	Transfer (T)	++	++	++	++	++	++	++	0	0	0	0	0	**14**	0	++	++	0	0	0	++	++	0	+	**51**
	Primary dune formation (DUNE)	0	0	0	0	0	0	0	++	0/+	0	0	+	**3.5**	0	0	0	0	+	++	0	0	0	++	**43**
	Large-scale wind dynamics (LW)	0	0	-/0	0	0	0	0	0	++	+	-	+	**2.5**	-	0	0	0	-	++	0	+	0	+	**31**
	Small-scale wind dynamics (SW)	0	0	0	0	0	0	0	0	+	++	0	+	**4**	0	0	0	0	0	+	0	+	0	0	**18**
	Infiltration (IF)	0	0	0	0	0	0	0	0	0	0	++	++	**4**	++	0	0	0	++	0	0	+	0	0	**20**
	Evapotranspiration (ET)	0	0	0	0	0	0	0	0	0/+	0	0	--	**-1.5**	-	0	0	0	--	0	-	0	0	0	**-14**
	Soil development (SOIL)	0	0	0	0	0	0	0	0	-	+	++	+	**3**	+	0	0	0	-/+	-	0/+	+	0	0	**0.5**
	Vegetation development (VEG)	0	0	++	0	0	0	0	++	++	++	++	++	**12**	0	0	0	0	0	++	+	0	0	+	**12**
	Primary production (land) (PP)	0	0	+	0	0	0	0	+	+	+	++	++	**8**	0	0	0	0	0	0	++	++	0	0	**2**
	Gas emissions (GHG)	0	0	0	0	0	0	0	0	0	0	0	0	**0**	0	0	0	0	0	0	--	+	0	0	**2**
	Denitrification (DEN)	+	0	+	0	+	+	+	+	0	+	+	+	**9**	--	0	0	0	0	0	-	++	0	0	**11**
	Population dynamics (POP)	++	++	++	++	++	++	++	++	+	++	+	++	**22**	-/+	++	-/0	0	0	0/+	0	0	0	+	**26.5**
ANTHROPOGENIC PROCESSES	Sediment extraction (SED)	-	--	0	--	-	0	0	0	0	0	0	0	**-6**	0	-	0	++	0	0	0	0	-	0	**-3**
	Sediment dumping (DUM)	-	--	-	--	-/0	-	0	0	0	0	0	0	**-7.5**	0	-	-	0	0	0	0	0	0	0	**-10**
	Bottom disturbing fishing (BeF)	0	-	0	--	--	0	0	0	0	0	0	0	**-5**	0	+	0	0	0	0	0	-	0	+	**8**
	Pelagic fishing (PeF)	-	0	0	0	0	0	0	0	0	0	0	0	**-1**	0	+	0	0	0	0	0	0	0	+	**16**
	Artificial reef formation (ARF)	-/+	0	0	++	-	-	0	0	0	0	0	0	**-1**	0	+	++	-	0	+	0	++	0	+	**42**
	Artifical infiltration (AIF)	0	0	0	0	0	0	0	0	0	0	0	+	**1**	+	0	0	0	++	0	0	+	0	0/+	**23.5**
	Drainage (DRA)	0	0	0	0	0	0	0	0	0	-/0	-/0	--	**-3**	++	0	0	0	-	0	--	--	0	0/+	**-20.5**
	Water extraction (EXTR)	0	0	0	0	0	0	0	0	0	-	-	--	**-4**	-	0	0	0	+	0/+	--	--	0	0	**-13**
	Manuring (MAN)	-	0	-	0	0	--	-	+	--	--	-	--	**-11**	++	0	0	0	0	-	-	--	0	0	**-27**
	Grazing (GRZ)	0	0	-	0	0	0	0	0	0	+	-	+	**0**	++	0	0	0	--	0	--	-	0	0	**-22**
	Cropping (CRP)	0	0	+	0	0	0	0	0	0	--	--	--	**-5**	++	0	0	0	--	0	-	0	0	0	**-11**
	Disturbance by access (TR)	0	0	0	0	0	0	0	--	+	0	-	-	**-3**	0	0	0	0	0	-	0	0	0	++	**8**
	Surface hardening (PAV)	0	0	0	0	0	0	--	--	--	--	--	--	**-12**	--	0	0	0	--	--	--	--	0	-/+	**-54**
	Sand nourishing (NOUR)	-	--	0	--	+	-	-	+	+	0	0	0	**-4**	0	-	0	+	+	++	0	0	0	++	**42**
	Nature management (NAT)	+	++	++	+	+	+	+	++	+	++	+	++	**17**	0	++	0	0	+	+	-	+	0	++	**52**
	Biological invasions (INV)	--	--	--	--	--	--	0	0	0	-	--	-	**-16**	--	--	--	0	-	-	0/+	0/+	0	-/+	**-31.5**
	Noise and visual disturbance (DIS)	--	--	--	--	--	--	--	--	--	--	--	--	**-24**	0	0	0	0	0	0	0	0	0	-/+	**0**

Table 5.4 – Score and numeric value per type of impact of processes on habitat and ecosystem services.

Score	Type of impact	Numeric value
--	important negative impact	-2
-	moderate negative impact	-1
-/0	relationship is uncertain, rather negative impact is expected	-0.5
0	no relationship	0
+/-	positive and negative effects are expected to be equally large	0
0/+	relationship is uncertain, rather positive impact is expected	0.5
+	moderate positive impact	1
++	important positive impact	2

Step 5: Recommendations for ecosystem processes

This step was performed as described in the Generic Methodology section.

5.4. Results

The impact matrix (Table 5.3) shows in detail which trade-offs, conflicts and synergies among habitats and ecosystem services arise when influencing certain processes. Although effects of anthropogenic processes in the Belgian coastal ecosystem are mostly rather negative for the habitats, some also provide benefits for habitat development (e.g. sand nourishing creates opportunities for the development of young, embryonic dunes) and for multiple ecosystem services (artificial reef formation provides benefits for water quality regulation through the filtering capacity of fouling communities). Ecological processes have positive effects on the majority of habitats and ecosystem services, although some have important negative impacts especially on ecosystem services (e.g. effects of evapotranspiration on agricultural production, climate regulation and drinking water provisioning). Drinking water provisioning, flood regulation, climate regulation and water quality regulation are ecosystem services that are most affected by anthropogenic pressures. This is in line with conclusions from other studies that regulating ecosystem services present highest trade-offs with provisioning services (Bennett E.M. et al. 2009; Howe et al. 2014), to which most of the anthropogenic processes are linked. Most uncertainties are related to effects of population dynamics and biological invasions on ecosystem services, and contributions of processes to recreation.

Figure 5.3 also shows that anthropogenic processes cause more conflicts than ecological processes, with the expectation of nature management which is meant to improve habitats. Biotic processes (e.g. EE, T, POP, BeP, PeP) are generally ranked higher than abiotic processes (HD, MD, SD, LW, SW, IF), underpinning the contribution of organisms to ecosystem services provision (Luck et al.

2003). Surface hardening (PAV), biological invasions (INV) and manuring (MAN) respond least to long-term socio-economic needs in the coastal zone and should be reduced to a minimum.

Bearing in mind the synergies, conflicts and trade-offs (Table 5.3, Figure 5.3) a series of management priorities was formulated to optimize ecosystem services and biodiversity in the Belgian coastal system (Figure 5.4). The backbones of the vision to which all goals can be linked are space for dynamic processes and connectivity. One of the key drivers of diversity at sea and in the dunes is the transport of sediment and the continuous reshaping of the environment (HD, MD, LW, SW). Sand transport is also crucial to create new dunes (DUNE) and maintain a resilient shoreline which is able to adapt to external stress such as sea level rise. Biotic processes (BeP, PeP, POP, VEG, PP) and fluxes (T) are crucial to create and sustain diversity, and to develop the self-regulating capacity of ecosystems (EE) which is the driver of ecosystem services such as nutrient cycling (SOIL, DEN), climate regulation (GHG) and coastal safety (EE, DUNE), but also control of biological invasions (INV). Management should primarily focus on sustaining and enhancing these biotic and abiotic dynamic processes to safeguard ecosystem services and habitats. Connectivity on the other hand may facilitate biological invasions and require active management (NAT). Moderate consumption of ecosystem services may be compatible with qualitative habitat development (SED, DUM, BeF, PeF, AIF, DRA, EXTR, GRZ, CRP, TR). Urbanisation (PAV) poses a threat to a healthy ecosystem because less space is available, sedimentary and biotic processes get interrupted, habitats fragmented and disturbed by noise (DIS). Based on our approach, we recommend drastic changes in coastal zone management in light of climate change, i.e. to develop strategies to advance or retreat rather than to maintain the current line.

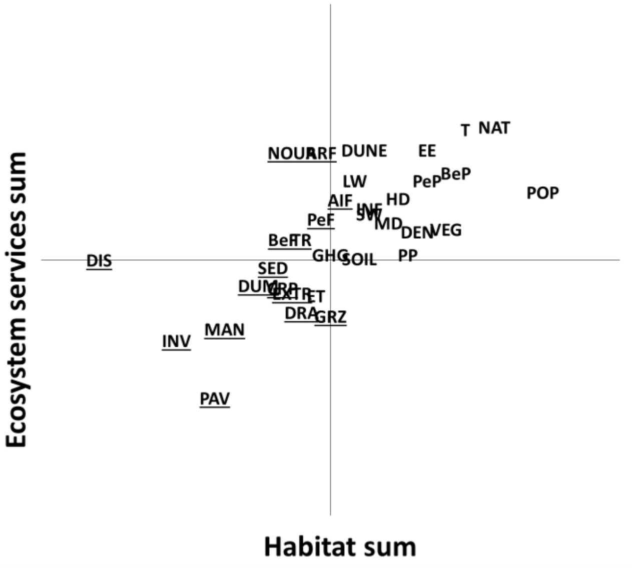

Figure 5.3 – Plots of the processes in the Belgian coastal zone based on their contribution to habitat development and ecosystem services and weighted for societal demands. Underlined: anthropogenic processes, not underlined: ecological processes. Abbreviations in Table 5.3.

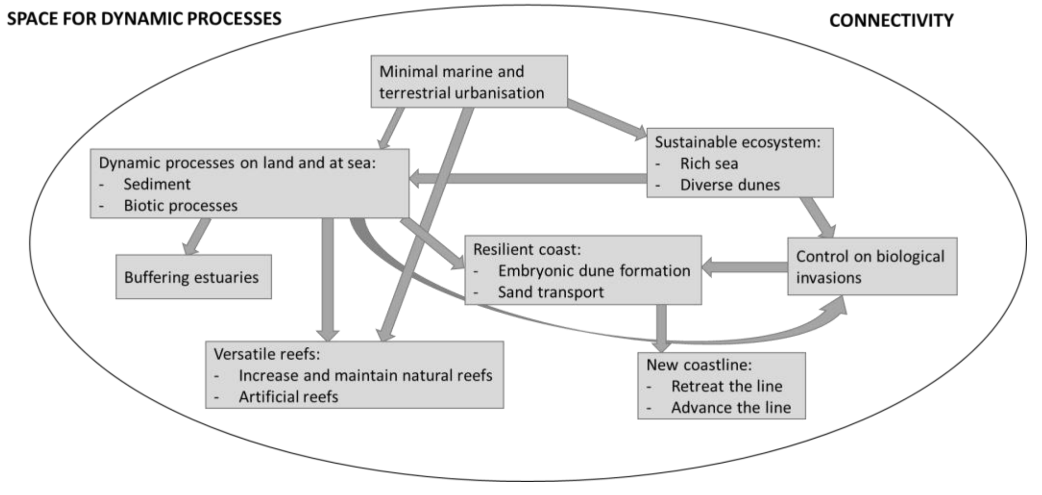

Figure 5.4 – Scheme of the strategic vision for the Belgian coastal zone indicating how the management goals are connected to each other. Backbones of the vision are space for dynamic processes and connectivity.

5.5. Discussion

5.5.1. Creating a shared vision

The presented approach is developed for the purpose of defining long-term objectives to achieve a healthy and socio-economically sustainable ecosystem, which has a rather conceptual nature and takes place in an early and strategic stage of spatial planning and policy. Considering effects on biodiversity and on ecosystem services from the beginning of the planning process stimulates and assists in the search for alternative, ecosystem-based solutions to avoid or mitigate potential conflicts and/or optimize ecosystem benefits. It also opens opportunities for early stakeholder involvement, avoiding local opposition to spatial plans and protected areas resulting from top-down imposed regulations (Beunen et al. 2013).

The approach provides guidance in the development of a shared vision in environmental planning by identifying the ecosystem processes that create conflicts and synergies between biodiversity conservation and ecosystem services. Processes in the outer corners have most and/or strongest impacts on habitats and/or ecosystem services. Recommendations for these processes are generally most straightforward. Processes in the far upper right corner have mainly positive impacts both on habitats and on ecosystem services. Management efforts should primarily focus on restoring or sustaining these processes in view of a healthy ecosystem. In the far lower left corner are processes that should be minimized as much as possible to avoid changes in ecosystem functioning that lead to losses of biodiversity and ecosystem services. In some cases, active management of these processes may be required to restore ecosystem functions.

The most important trade-offs occur in the upper left and in the lower right corner. Depending on the benefits created by a process, negative impacts on other values are tolerable or not. For these processes a choice needs to be made whether to accept trade-offs, avoid these processes or minimize negative impacts by restricting or adapting the process. In the case study, many anthropogenic processes are located in the upper left corner, illustrating the potential multi-functionality of certain anthropogenic interventions and – depending on the location along the habitat axis – benefits for some habitats. Adaptation to reduce trade-offs is possible by 1) decreasing the consumption of the ecosystem services so that the process reduces in intensity, frequency or geographical extent or (2) search for alternative forms, location or timing to produce the ecosystem services. For example, recreation also involves human trampling of embryonic dunes for example (Figure 3, TR), which hampers succession of these dunes. Access to the most critical areas can be restricted by creating (temporary) no-go zones, while access can be allowed in other parts. Very few processes will have only positive effects on habitats and negatively affect ecosystem services (lower right corner), which may be explained by the diversity of ecosystem services that is taken into account, underpinning the need to account for the variety of ecosystem services including provisioning, regulating and cultural services. The presented methodology thus supports both the paradigm of conserving nature for its intrinsic value as for people's benefits (Mace 2014).

As the axes represent the sum of effects, important impacts may be masked due to an accumulation of multiple smaller effects or effects on less important ecosystem services. Processes occurring on the negative side of the habitat or the ecosystem services axis may still have benefits for habitats or ecosystem services resp., depending on how far they are located along the axis. It is therefore important when defining objectives to also inform on the exact ecosystem services and habitats that are affected. Factors that can play a role in the decision are e.g. the local conservation status of the impacted habitat(s) or the economic or social value of the affected ecosystem service(s).

The method prioritizes processes by ordering them according to a degree of multi-functionality, based on the sum of the multiple effects a process has on ecosystem services and habitats. The debate on choosing over priorities thus shifts from a focus on conflicts between sectors to a common goal of multi-functionality. This facilitates communication among stakeholders as (1) sectors are not explicitly targeted in the discussion, (2) potential benefits are also taken into account and may compensate minor negative effects and (3) socio-economic benefits of biodiversity conservation are made explicit. This multi-functionality can be illustrated for the case of pelagic fishing (Figure 5.3, PeF). PeF, located in the upper left corner, falls in the category of processes that should be restricted, adapted or for which negative impacts can be accepted. PeF indeed can disturb the pelagic food web and may change ecosystem functioning in case of overfishing. PeF, however, has important benefits for multiple ecosystem services of high importance in the coastal zone (fisheries production and recreation, in particular recreational angling and fish markets as tourist attraction). Restricting the intensity of pelagic fishing (e.g. through quota) may thus be compatible with biodiversity and ecosystem services targets.

Thinking ahead of the impact of future socio-economic pressures helps to prevent damage to the ecosystem resulting from gradually increasing intensity or frequency of anthropogenic processes. Those processes with strong or multiple adverse effects are identified, allowing to search for alternative, ecosystem-based solutions with less impact on the environment. For example, the construction of a hard-engineered, concrete dike (PAV) to prevent erosion of the dune foot negatively affects several habitats (Figure 5.3, negative score for 'habitat SUM') and ecosystem services (Figure 5.3, negative score for 'ES SUM'), either directly because of habitat destruction or indirectly because of the interruption of sand transport between beach and dunes and degradation of young dunes with marram grass which rely on the continuous supply of fresh sand. This affects important ecosystem services such as recreation and landscape attractiveness as the typical dune landscape disappears, and flood protection since dunes are not able to grow with sea level rise without supply of sand (Van der Biest et al. 2017a). An alternative form to prevent dune foot erosion is through artificial sand nourishment (NOUR), which has a much less negative impact on habitats (Figure 5.3, slightly negative score for 'habitat SUM') and creates opportunities for ecosystem services such as recreation (Figure 5.3, high positive score for 'ES SUM').

5.5.2. Biotic and abiotic processes

The usage of expert elicitation opens the opportunity to take into account ecosystem processes which, due to complexity, are either replaced by static biodiversity metrics (species, habitats) or restricted to a few processes mostly related to aspects of biodiversity rather than to the underlying functional mechanisms that drive biodiversity. An approach such as presented here allows to select the key processes as the basis for good ecosystem functioning and that need to be considered in spatial allocation of land use, protected areas and other management measures. For these key processes, that constitute the objectives of management and planning, more detailed and quantitative methodologies are than preferably used in case these are available.

Focussing on processes enables embedding biodiversity and ecosystem services in large scale planning as processes do not stop at the boundaries of protected areas. By considering processes as key to manage the ecosystem it becomes possible to take into account interactions between habitats resulting from dynamic processes on variable spatial scales. The methodology allows to assess the impact of a process on multiple habitats within an ecosystem. Reef formation through ecological engineering for example is typically associated with benthic habitats, but an increased species diversity and biomass production associated with reef formation positively affects the pelagic habitat through increased bentho-pelagic coupling. As illustrated for the case-study, a process-based approach is particularly suitable for boundary ecosystems such as land-ocean zones where transboundary processes and connectivity are determining, and for large-scale and long-term planning where processes take place on different scales. The benefits of large-scale wind dynamics for flood protection for example may be realized only on the long term as dynamic dunes in contrast to stabilized dunes are capable of migrating inland along with sea level rise.

By including anthropogenic besides natural processes, it becomes possible to account for feedbacks between ecosystem services. This is identified as one of the main shortcomings in the most commonly applied approaches for assessing ecosystem services (e.g. InVEST, ARIES), which can assess ecosystem services simultaneously, but independent of each other (Rieb et al. 2017). It also allows to include interactions between ecological and socio-economic factors, which is identified as another main challenge for ecosystem service research (Bennett 2017). However, interactions among habitats and ecosystem services and feedbacks of habitats and ecosystem services on processes can be incorporated only to a limited extent, i.e. when a change in habitat or ecosystem services is linked to a change in a process affecting another habitat or ecosystem service (e.g. an increase in surface area of dune shrub goes along with reduced large-scale wind-dynamics, negatively affecting remaining dynamic white dunes). Another restriction is that non-linear effects of processes and interactions between processes that reduce or increase the impact of a process on a habitat or ecosystem services are not included in the approach as presented here. However, there is potential for users to further elaborate this. The main aim of this chapter is to demonstrate how including processes opens opportunities to align biodiversity and ecosystem services in early planning phases and to provide inspiration to advance current approaches for planning for biodiversity and ecosystem services.

The application of the methodology on the case-study shows the merits of focussing on processes to deal with recently identified shortcomings related to the integration of biodiversity conservation with ecosystem services in spatial planning. Despite its shortcomings, the methodology supports the process of aligning biodiversity conservation with socio-economic demands, provides an approach to identify the key processes management should focus on and provides opportunity for early stakeholder involvement supporting the creation of a shared vision. This process-based rationale is further elaborated upon to integrate ecosystem services in impact assessments (see Chapter 6).

5.6. References

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6

A process-based approach to integrate ecosystem services into impact assessments

Van der Biest K., Schellekens T., Bonte D., Ysebaert T., Vanagt T., Maris T. and Meire P. Integrating ecosystem services into impact assessments: a process-based approach. *In preparation.*

6.1. Abstract

Policy makers increasingly acknowledge the importance of considering ecosystem services and biodiversity in impact assessment to reduce ecosystem degradation and halt the ongoing loss of biodiversity. Recent research demonstrates how an inclusion of ecosystem services can add value to an impact assessment, i.e. by shifting the focus from avoiding negative impacts to creating opportunities, by linking effects on ecological functioning to benefits for society and by providing a multi-disciplinary framework that allows to consider cross-sectoral, cumulative effects. However, with these advantages also new challenges arise which obstruct implementation in practice. The most commonly used ecosystem services models have a single-service approach, where each ecosystem service is being assessed separately without considering interactions among ecosystem services. This restricts their capacity to account for cross-sectoral effects. Integrating ecosystem services into impact assessments also increases the time- and resource investments to perform an impact assessment as they cover a wide variety of disciplines and they generally need detailed and spatially-explicit information. This chapter presents an approach that tackles these challenges and aims to facilitate the integration of ecosystem services into impact assessments. The approach focusses on ecosystem processes as the nexus of ecosystem services and biodiversity and the basis to evaluate effects of a project. Using the case-study of the Belgian coastal ecosystem, we illustrate how it restricts the potentially large amount of data needed to quantify ecosystem services, how it allows to account for cumulative impacts beyond the spatial and temporal scale of the individual project and how it facilitates the development of ecosystem-based alternatives.

6.2. Introduction

Policy makers increasingly acknowledge the importance of integrating biodiversity with ecosystem services in spatial planning and impact assessment to reduce ecosystem degradation and avoid further biodiversity losses (Mandle et al. 2016; Mascarenhas et al. 2015; Staes et al. 2017). Notable examples are the EU Biodiversity Strategy 2011-2020 which requires member states to map and quantify ecosystem services, and the foundation of the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) in 2012 which aims to improve the science-policy interface in response to requests from decision makers (IPBES website 2017). Including ecosystem services has the potential to improve traditional impact assessments and deal with some of its shortcomings (Geneletti 2011; Honrado et al. 2013). Where impact assessments traditionally focus on avoiding negative impacts on the environment, an ecosystem services perspective allows accounting for potential benefits a local anthropogenic intervention – hereafter referred to as a project – may create, besides potential risks (Baker et al. 2013; Goodstadt et al. 2010). The added value for human well-being of avoiding environmental risks is also often not clear (Baker et al. 2013; Bowd et al. 2015;

Hattam et al. 2017; Honrado et al. 2013; Mandle et al. 2016). Ecosystem services are directly expressed in terms of gains for human well-being, making the benefits for stakeholders more tangible. Finally, impact assessments typically evaluate on a sectoral basis (e.g. soils, climate, biodiversity, health, …) and do not always account for cumulative, cross- sectoral effects (Baker et al. 2013; Hattam et al. 2017). The ecosystem services paradigm is based on an integration of all aspects of ecosystem functioning with socio-economic factors and is thus cross-sectoral by definition. Its holistic character also supports the inclusion of effects that might be overlooked due to unawareness, and the consideration of trade-offs and synergies among different sectors.

However, the implementation of ecosystem services in impact assessments faces some difficulties which may impede its success and usefulness in practice (Baker et al. 2013). Ecosystem services assessments are comprehensive, covering a variety of disciplines, and therefore resource- and knowledge-intensive. In spite of the efforts spent on knowledge integration in recent years, through the development of hands-on instruments such as InVEST and ARIES, application in practice remains challenging (Baker et al. 2013; Broekx et al. 2013; Mandle et al. 2016; Partidário and Gomes 2013; Van der Biest et al. 2014, 2015). The extensive amount of data that is needed to quantify ecosystem services has been identified as an obstacle to this so-called 'implementation gap' (Cook and Spray 2012). Furthermore, although the ecosystem services framework does provide an opportunity to consider cross-sectoral effects, additional steps need to be taken to make use of this potential. The most common ecosystem services toolboxes (such as InVEST and ARIES) are single-service approaches: each ecosystem service is being assessed separately without considering feedback mechanisms between ecosystem services (Rieb et al. 2017). Cumulative, cross-sectoral effects however can only be identified if such interactions are taken into account. A standardized framework to select amongst the ecosystem services to be considered in an impact assessment is also lacking (Hooper et al. 2014; Staes et al. 2017). Ecosystem services with poor information are often neglected (Grêt-Regamey et al. 2017), thus threatening unbiased decision-making (Seppelt et al. 2012). Lastly, although an integration of different sectors has been argued to facilitate the design of alternatives (Geneletti 2014), flexible tools that support the iterative process of alternative design and refinement based on expected environmental impacts remain scarce (Bigard et al. 2017; Partidário and Gomes 2013). These arguments may support the notion that an integration of ecosystem services in impact assessments is more a burden than a help (Baker et al. 2013).

Focussing on the underlying ecosystem processes that produce ecosystem services and biodiversity has increasingly been acknowledged as key to predicting changes in ecosystem services (Nicholson et al. 2009; Schneiders and Müller 2017; Zalewsky et al. 2013) and for more effective biodiversity conservation (Klein et al. 2009; Watson et al. 2016). Ignoring the causal relationships between ecosystem processes and ecosystem services may lead to losses of what the ecosystem services concept actually aims to conserve, i.e. biodiversity (Wong et al. 2015). Considering ecosystem processes as the nexus of ecosystem services and biodiversity and the target of ecosystem-based management allows aligning ecosystem services with biodiversity conservation in spatial planning

(Chapter 5). Impact assessments likewise should evaluate how a project affects processes, and how this ultimately results in changes in ecosystem services and biodiversity values, beyond the scale of the individual project and on longer time-scales, rather than on assessing merely the current state of the ecosystem. We propose a methodology that builds upon this rationale and discuss how it can help to overcome some of the challenges related to the inclusion of ecosystem services in impact assessments. We first elaborate on the methodological principles in a generic way and illustrate its functionalities by applying the method on a project in the Belgian coastal zone.

6.3. Methods

6.3.1. Generic methodology

The methodology focusses on the key role of ecosystem processes as underlying mechanisms for ecosystem services and biodiversity (Nicholson et al. 2009; Watson et al. 2016) – cfr. the ecosystem approach as defined by the Convection on Biological Diversity (CBD) – and as objectives of ecosystem-based management. Ecosystem processes are here defined as “changes in the stocks or in the fluxes of products and energy resulting from interactions among organisms (incl. humans) and with their abiotic environment, and from abiotic driving forces” (Chapter 5). Ecosystem processes are directly related to the supply and consumption of ecosystem services and biodiversity assets (Figure 6.1). These include ecological (e.g. natural infiltration) as well as anthropogenic processes (e.g. artificial infiltration of pre-treated sewage water in dunes) as they both influence ecosystem services and biodiversity. Alterations in the rate of a process resulting from changes in the parameters of a process within the ecosystem under study are caused by: (1) direct human interventions related to the project (e.g. habitat destruction); (2) variations in the ecosystem’s properties related to the project (e.g. increased inflow of pollutants) or to external stresses. External stresses refer to changes taking place on larger spatial scales than the ecosystem under study (e.g. effects related to climate change) or drivers from outside the ecosystem that affect processes within the ecosystem (e.g. nutrient supply through ocean currents). While external stresses affect the supply of ecosystem services through changes in processes (e.g. reduction of flood protection due sea level rise), they can also alter the demand for ecosystem services in a more direct way (e.g. increased need for cooling due to higher summer temperatures). The demand for ecosystem services, biodiversity conservation or external stresses are driving forces for a project. The relationships between biodiversity and ecosystem services are very diverse and can be negative and positive (Ricketts et al. 2016), although evidence supports a positive effect of biodiversity on ecosystem services in general (Hooper et al. 2011; Harrison et al. 2014) and a negative effect of overconsumption of ecosystem services on biodiversity (Bennett et al. 2015).

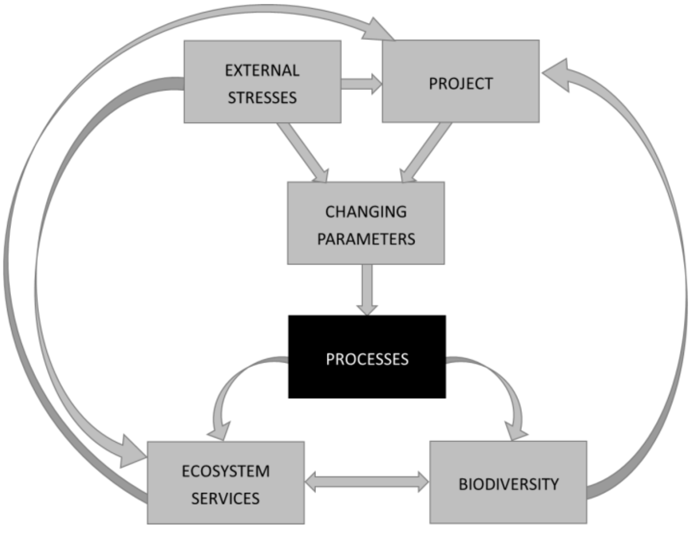

Figure 6.1 – Integration of ecosystem services and biodiversity in impact assessments based on an assessment of underlying processes

The methodology is described in a stepwise procedure. Steps 1 – 5 are the preparatory steps to elaborate the framework, step 6 integrates its application into impact assessments. Steps 1 – 5 should be performed at the scale at which the ecosystem is affected by a project. This requires an interdisciplinary perspective, involving a multidisciplinary team with experts from different fields and stakeholders (Nesshöver et al. 2017). The method is formalised using coupled matrices, one for the impacts of processes on habitats and ecosystem services and one for the effects of changing parameters on processes (Figure 6.2). The relationships between changing parameters, processes, habitats and ecosystem services in Step 2 and Step 3 should be backed-up with scientific evidence and assumptions documented. This allows users and practitioners to understand and consider well the consequences of their choices (Guerry et al. 2015), and avoid hesitation to use the tool caused by a lack of understanding and trust in the instrument (i.e. 'black box' effect).

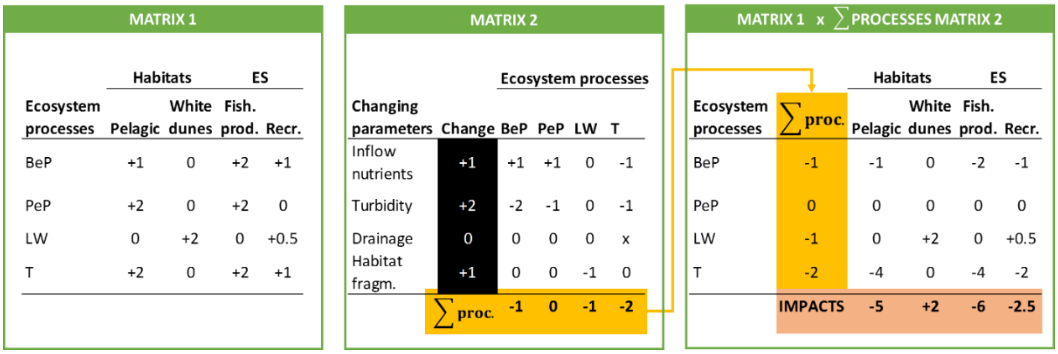

Figure 6.2 – Scheme of the methodological approach using coupled matrices. Matrix 1 = effects of processes on habitats and ecosystem services. Matrix 2 = effect of changing parameters on ecosystem processes. For each process in Matrix 1, the value of its impact on a habitat or on an ecosystem service is multiplied with the sum per process as derived from Matrix 2. Black cells: to be filled in to perform an impact assessment (Step 6). BeP = benthic production, PeP = pelagic production, LW = large-scale wind dynamics, T = transfer.

Step 1: Identification of habitats and ecosystem services

In a first step, the habitats and relevant ecosystem services of the ecosystem under study are identified (Matrix 1 in Figure 6.2). Habitats are used as umbrellas for biodiversity values or species and can for instance be identified based on internationally recognized systems (e.g. NATURA2000, EUNIS, ...) or local frameworks. Any habitat that occurs naturally in the ecosystem or that is expected to occur in the future, for example because of active management, can be included. The scale on which they are defined should be such that contrasting effects of processes between habitats can be distinguished as much as possible (see Step 2). This means that when the effects of a process are variable within a single habitat, it should be considered to further subdivide this habitat into separate habitats (e.g. distinguish between vegetated and non-vegetated tidal flats). The total number of habitats however needs to be balanced so that the instrument remains manageable.

The involvement of stakeholders in the identification of relevant ecosystem services is essential (Paavola and Hubacek 2013; Reyers et al. 2009). To be able to account for long-term effects it is necessary to consider not only current use of the ecosystem but also to bear in mind future socio-economic needs (Honrado et al. 2013; Landsberg et al. 2014), which may alter under climate change, demographic growth etc. This requires an explicit choice to be made on the time frame of the analysis and may depend amongst others on policy goals at different levels and climate change projections.

Step 2: Identification and functional description of ecosystem processes

For each of the selected habitats, the processes are identified that contribute to their development or maintenance or to biodiversity within a habitat (Matrix 1 in Figure 6.2). For ecosystem services, the processes are identified that contribute to their production and to their consumption and that have an impact on habitats. This is based on the best available knowledge on the ecosystem functioning and the socio-economic dependencies on the ecosystem. A score is attributed that expresses the magnitude and the direction of the impact of a process on each habitat and ecosystem service (the impact score). This score may vary depending on the magnitude of the process (e.g. positive effect of moderate increase of a process on an ecosystem service but negative effect of strong increase), allowing to account for linear as well as non-linear effects. This step can be done based on quantitative information such as derived from predictive models, or based on expert judgment in case quantitative information is lacking. The effects of biodiversity on ecosystem services can be taken into account by including the driving mechanisms of biodiversity within a habitat as ecological processes and attributing a score for their effect on ecosystem services. Effects of ecosystem services consumption on biodiversity can be taken into account by linking anthropogenic processes to the habitats that cover biodiversity values.

Contrasting effects of processes (positive and negative effects on a habitat or ecosystem service) can be avoided by subdividing into narrower defined habitats as described in Step 1, but also by subdividing ecosystem services (e.g. benthic and pelagic fishing) or processes (e.g. methane production and nitrous oxide production as greenhouse gas emissions). It is recommended to avoid contrasting effects as much as possible in order to prevent loss of information when combining them in a single score. In case further subdividing is not possible or desirable, the effects can be weighed against each other resulting in a single overall score. Uncertain processes regarding effect sizes can be included by providing a range of the expected effects or by calculating an expected value based on the probability distribution of the range of effects. The latter is typical for Bayesian belief modelling, a technique which is increasingly used in ecosystem services assessments (Grêt-Regamey et al. 2013; Van der Biest et al. 2014; Landuyt et al. 2015). Especially when the process has a strong impact on the final outcome, as determined from a sensitivity analysis, it is important to take the range on effects into account.

Step 3: Identification and description of drivers of change

Changing parameters are those parameters that may cause a variation in the rate of processes and eventually lead to the modification of biodiversity and ecosystem services (positively or negatively). These changing parameters (Matrix 2 in Figure 6.2) can be related to direct human interventions (e.g. dredging), indirect effects resulting from changes related to the project (e.g. changes in flood frequency) or external stressors (e.g. temperature increase due to climate change). Changes related to the project can result from degradation of existing habitat or from the creation of new habitat.

As scientifically derived quantifications are usually not available, the effects of changing parameters on processes can be qualitatively described based on scientific evidence. A score is attributed for the size and the direction of the impact of a changing parameter on a process, and this score may vary depending on the intensity of the changing parameter. Similarly as in Step 2, linear and non-linear effects can thus be defined. Changing parameters for which the effects on processes are uncertain can be merely highlighted or, if possible, attributed a range of the expected effects or an expected value can be calculated based on the probability distribution of the expected effects. This is especially important when the changing parameter has a strong impact on the final outcome.

Step 4: Impact on ecosystem processes

A project usually causes multiple changes in the environment and a single process can thus be affected by more than one parameter. The different effects are taken into account by summing per process the expected changes resulting from the different parameters. The unit change of a parameter should be proportionate to its effect on a process and may be different from the unit change in another parameter (e.g. a moderate increase in a process may be caused by a moderate increase in one parameter or by a strong increase in another parameter). Because processes interact with habitats and ecosystem services in different ways, they cannot be compared quantitatively. It is therefore recommended to reclassify the final sum per process to a range in the same order of magnitude as the range used for the scores.

From this step onward, the effects related to the modification of existing habitat and those resulting from creation of new habitat are treated separately. A separate sum per process is made for both, so that effects on habitats and ecosystem services (Step 5) can be analysed individually.

Step 5: Impact on habitats and ecosystem services

The sum of the changes per process (Step 4) is multiplied with the impact score of processes on habitats and on ecosystem services (Step 2) (coupling of matrix 1 and 2 in Figure 6.2). Finally, the effects of the different processes on each habitat and ecosystem service are accumulated, resulting in a final outcome of how habitats and ecosystem services are affected by the project.

Step 6: Application in an impact assessment

The method allows to assess one habitat transformation at a time, consisting of a modification or degradation of the existing habitat and, if this is the case, creation of a new habitat. The user can choose the spatial resolution of the assessment, and this should depend on the size of the habitat patch that is affected, the scale of the dominant processes shaping the habitat and/or on the scale of the project. Besides the selection of the appropriate scale, the user needs to provide information on:

- the type of the existing, affected habitat
- the expected changes in parameters as identified in Step 4 (black column in Figure 6.2) resulting from a modification in the existing habitat
- the type of the potential new habitat

- the expected changes in parameters as identified in Step 4 (black column in Figure 6.2) resulting from the potential creation of new habitat

Changes in parameters should be expressed relative to the current state of the ecosystem. An increase or decrease should thus only be identified as strong if the change is large in comparison with the scale of that changing parameter in the ecosystem today.

6.3.2. Illustration of the methodology for the Belgian coast

We here develop a case of the Belgian coastal ecosystem to illustrate the functionalities of the method. The terrestrial border of the targeted region is delineated by the transition from polder to dunes (up to ~2 km landward from the shore), and the marine limit coincides with the boundary of the Belgian Continental Shelf (~70 km seaward from the shore). The land part (80 km² along a stretch of ~70 km coastline) is covered with dunes, both under a protected status as well as dunes used as pasture or private gardens. The area also comprises two zones with tidal flats and marshes of which one is part of an estuary. The marine zone (3600 km²) is part of the Southern North Sea and is made up of soft sediments with a series of parallel sand banks hosting a high benthic diversity as a result of the highly variable topography and sediment composition (Degraer et al. 2008; Vanden Eede et al. 2014; Verfaillie et al. 2010).

Step 1: Identification of habitats and ecosystem services

The procedure for identifying the habitats and ecosystem services is described in detail in Step 1 in Chapter 5 which elaborates upon the development of a strategic vision for the Belgian coastal ecosystem. The experts involved in this step are described in Appendix F Table F.2.

Habitats were largely based on the NATURA2000 habitat types and additionally the European habitat classification EUNIS which distinguishes in more detail marine habitats (Table 6.1). Artificial reefs (jetties, wind mill foundations, etc.) were included because of their ubiquity, potential ecological values (Perkol-Finkel et al. 2012), distinct ecological functions and ecosystem services they may facilitate (Wetzel et al. 2014).

The Common International Classification of Ecosystem Services CICES v4.3 (EEA 2016) and the marine typology of ecosystem services of Böhnke-Henrichs et al. (2013) were used as reference frameworks to select relevant ecosystem services. This resulted in a list of 8 ecosystem services, of which 4 provisioning ecosystem services (agricultural production, fisheries production, aquaculture production, drinking water provisioning), 3 regulating ecosystem services (flood protection, climate regulation, water quality regulation) and 1 cultural ecosystem services (recreation). This initial selection was presented to a group of experts (Appendix F Table F.2) who added two ecosystem services (energy production from renewable sources and sediment extraction).

Step 2: Identification and functional description of ecosystem processes

This step is based on and described in detail in Step 3 in Chapter 5. Processes were identified based on an in-depth literature study and involving experts from different disciplines (Appendix F Table F.2), both natural (biodiversity, water quality, hydrodynamics, etc.) as socio-economic sciences (Van der Biest et al. 2017c). In total, 32 processes were distinguished and related to habitats and ecosystem services (Table 5.3, Appendix F Table F.1).

A score system ranging from -2 to +2 was used to describe the impact of a process on the quality and quantity of a habitat or on an ecosystem service (Figure 6.2). Multiple and contrasting effects were given a score 0 (positive and adverse effects are expected to be equally large), or the effects were weighed against each other resulting in a single overall score. Uncertain impacts were treated in a deterministic way by attributing the lowest possible score of +0.5 or -0.5. Although it is possible to take account of non-linear effects of processes on habitats and ecosystem services (e.g. positive effect of restricted catch on fisheries production; negative effect of overfishing), they were not included in the case-study.

Step 3: Identification and description of drivers of change

To have a comprehensive list of the potential changing parameters, the principle of the evaluation framework of the Scheldt estuary (Maris et al. 2014) was applied onto the Belgian coastal zone. This method unravels ecosystem functioning based on a tiered approach, and is originally developed to evaluate the current state of an estuarine ecosystem in an integrated way based on an established monitoring program. The highest level consists of the major components that describe ecosystem functioning (Figure 6.3). Each component is split up in several indicators for which parameters are identified that allow evaluating the state of each indicator. Finally, for each evaluation parameter explanatory variables (i.e. changing parameters) are identified that explain why the state of an indicator does not meet certain criteria (Appendix G). For example, a change in the inflow of nutrients (changing parameter) may explain why pelagic production (process) rates vary. Or the access of pedestrians to the dune foot (changing parameter) may explain why embryonic dune vegetation cannot establish and primary dunes are not building up (process). This step was based on an in-depth literature study and involving experts from different disciplines, as described in Step 2.

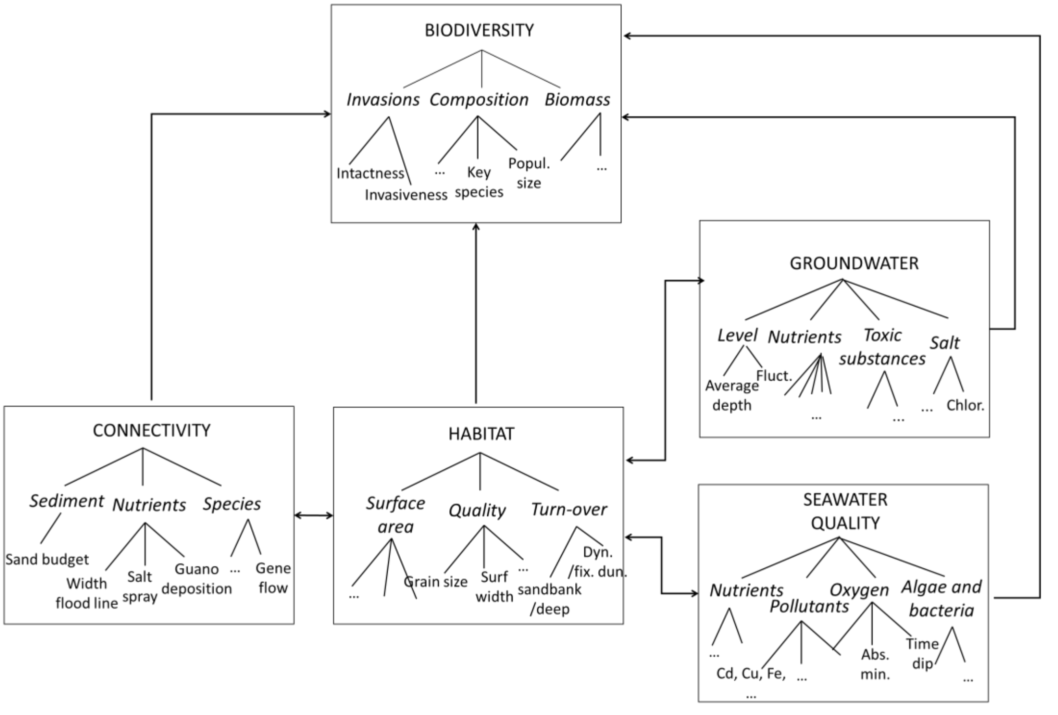

Figure 6.3 – Evaluation framework of the Scheldt estuary (Maris et al. 2014) applied onto the Belgian coastal zone. Major ecosystem components in capital letters, indicators in italic, evaluation parameters in small font.

A change in a parameter can be a moderate or strong increase (score 1 or 2), or a moderate or strong decrease (-1 or -2). This change can be positively correlated to a process (1 for a moderate and 2 for a strong positive effect), or negatively (-1 for a moderate and -2 for a strong negative effect). Non-linear effects were taken into account by allowing the sign and the size of the impact to vary with the sign and the size of change. Uncertain effects were treated in a deterministic way. Knowledge gaps were highlighted using the symbol “x”.

Step 4: Impact on ecosystem processes

The effects of changing parameters on each process were summed separately for the impacted and for the potential newly created habitat (Appendix G Tables G.1 and G.2). Knowledge gaps (“x”) were not included in this sum.

Step 5: Impact on habitats and ecosystem services

The sum of the effects of changing parameters on a process was multiplied with the impact score of a process on each habitat and ecosystem service (Figure 6.2). Finally, a sum was made for the effects of the different processes on each habitat and ecosystem service (Appendix G Table G.3). These values were then plotted in a spider diagram to visualize how a project development will affect habitats and ecosystem services. This step was done separately for the existing habitat and for the newly created habitat.

Step 6: Application in an impact assessment

The method was applied to an explorative study and design of a barrier island and marina in the lee of the jetty of Zeebrugge-port (Figure 6.4). The study is part of the Flemish government's Masterplan 'Flemish Bays' for protection of the coast against floods and creation of calm water for navigation (Kusteilanden.be 2017). The impacts of changing parameters on processes (qualitative scores) were estimated based on an extensive literature review (Van der Biest et al. 2017b).

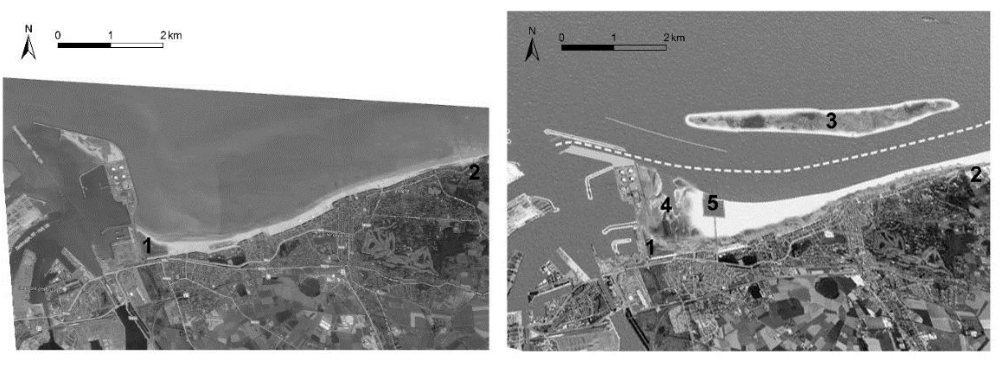

Figure 6.4 – Satellite image of the area today (left, Google Maps) and scenario of artificial barrier island with creation of a marina and tidal flats (kusteilanden.be). 1, 2 = existing tidal flats and marshes; 3 = artificial island; 4 = new tidal flats and marshes; 5 = marina.

6.4. Results

The construction of the island will have effects at the site of the island itself (on-site effects related to destruction of submerged sandbanks and foreshore habitat and creation of new dune habitat) and off-site along the shoreline due to changing hydrological conditions in the lee of the island and increased shipping traffic. Also the marina will have on-site and off-site effects. Because of its illustrative purpose, we here present the results of the assessment of off-site effects of the island and of on-site effects of the marina (conversion of lower beach habitat to marina consisting of pelagic habitat and artificial reefs habitat).

Construction of an island – off-site effects

As one of the objectives of the project is to allow inland ships to navigate safely from the port of Zeebrugge to the east in the lee of the barrier island, it is expected that the intensification of shipping traffic will increase the inflow of pollutants (+1); noise disturbance (+1), disturbance of the soil (+1) and turbidity (+1) due to passage of ships and resuspension of bottom sediments in an area dominated by fine sediments (Van den Eynde and Fettweis 2006). Due to calmer hydrological conditions in the lee of the island, the existing small tidal flats and marsh area will increase drastically (+2). The newly created tidal flats and marshes will be open for the public which will moderately increase the intensity of disturbance from access (+1). Because of the presence of a tidal flat area ~7 km east of the development (Figure 6.4), an increase in habitat area of tidal flats and marshes will improve connectivity (fragmentation new habitat -2) on a larger spatial scale. Habitat area of lower beach will decrease (-2) but this is not assumed to affect connectivity because of its ubiquitousness in that region. Tidal amplitude (-1) and hydrodynamics (-2) will decrease as a result of the island that buffers part of the tidal energy and wave energy. Additionally, a moderate sea level rise is assumed (+1).

The project has a strong positive effect on tidal flats and marshes (Figure 6.5), and a small positive effect on the quality of the pelagic, which is related to the nutrient buffering capacity of tidal marshes (Alldred and Baines 2016; Piehler and Smyth 2011). Both lower and upper beaches experience a negative impact which is explained by a reduction of lower beach area and therefore reduced supply of sand to upper beach and dune foot. The ecosystem service flood protection benefits from the construction of the island because of reduced wave and tidal energy. The improvement of water quality regulation is explained by the nutrient cycling and self-cleaning capacity of tidal flats and marshes (Van Damme et al. 2005), although the relative importance of this function may be negligible compared to the volume of water that floods the area. The effect on fisheries production is related to the importance of tidal flats and marshes as nursery grounds and the impact of connectivity with other marine habitats on population dynamics (Seitz et al. 2014). Aquaculture production could potentially benefit because of multiple small positive effects such as increased pelagic production (more food availability due to higher benthic production in tidal flats compared to lower beach habitat), enhanced transfer and reduced risk of biological invasions in less fragmented habitats (Didham et al. 2007). However, no aquaculture exploitation is currently present in the area. For recreation and tourism the effects of disappearance of beach area are counteracted by the increased value of the tidal area for nature recreation such as bird watching, again related to connectivity.

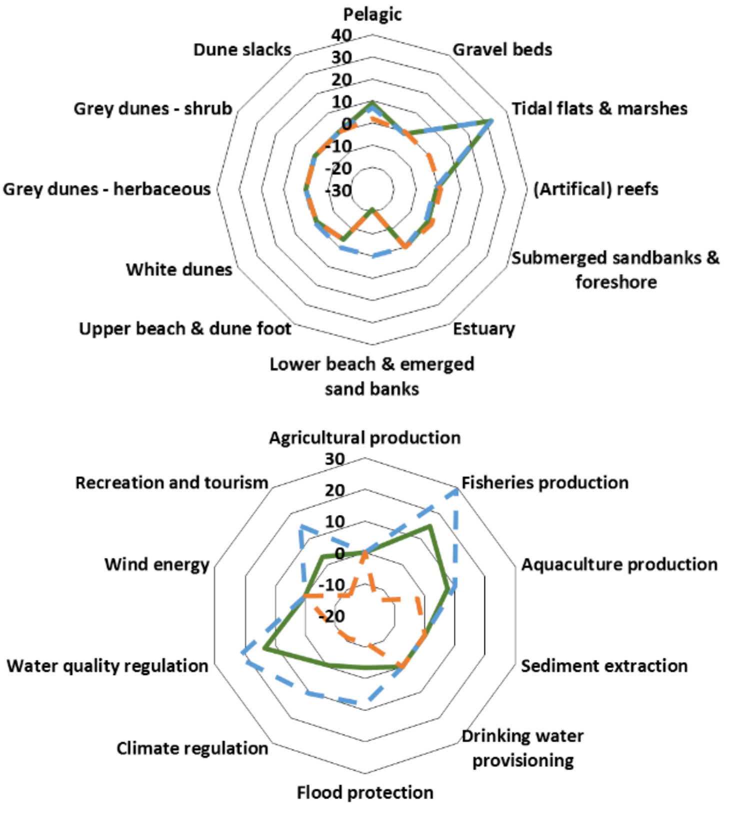

Figure 6.5 – Impact of the development of an artificial island resulting from changing hydrological conditions on existing habitat (orange), on newly created habitat (blue) and the combined effect of both (green). Left: effects on habitats, right: effects on ecosystem services

Construction of a marina

Changing parameters specific to the creation of a marina are an increase in the surface area of artificial reefs (jetties, floating platforms, ...) (+2) and a relatively higher inflow of pollutants from yachts in a confined area (+2). The passage of yachts and small boats will affect turbidity (+1) and soil disturbance (+1). Hydrodynamics are expected to be affected moderately (-1) because of the already calmer conditions in the lee of the harbor jetty and the island. Changes related to the degradation of beach area and sea level rise are similar as in the above described part of the project.

The increased and concentrated inflow of pollutants may potentially affect pelagic production in a negative way, which on its turn can affect habitats that strongly depend on pelagic production such as gravel beds. The effects on this habitat that is located outside the marina, however, are very small because of dilution. Although an increase in artificial substrata related to the construction of the marina (jetties etc.) is expected to strongly increase potential for development of reefs, this is not apparent from Figure 6.6. This is explained by the cumulative effects of morphodynamics with increasing boat traffic in a fine sediment environment. Reef-forming organism on the artificial structures are able to withstand a certain degree of sediment burial but will not survive under continuous sediment deposits (Borja et al. 2010). Increased turbidity may also negatively affect primary production on which many reef-forming organisms feed, depending on boat traffic intensity and draft of boats in relation to the water depth. Largest negative impacts on ecosystem services are for fisheries production, due to lower benthic production and transfer related to less beach area and tidal banks and, to a minor extent, pollutants. This effect however is compensated by an increased potential for fisheries production resulting from the newly created tidal flats and marshes. Recreation and tourism are less adversely affected by an accumulation of effects such as a lower benthic production and transfer resulting in less fish for recreational angling and a risk of temporary algal blooms related to a disturbed food web, and loss of beach area. A lower benthic production and a disturbed food web also affect climate regulation to a minor extent, but this may be compensated by the important carbon accumulating capacity of tidal marshes (Sutton-Grier and Howard 2018). The conversion from beach to marina has positive effects on water quality regulation related to the nutrient filtering capacity of bivalves growing on artificial reefs (Piehler and Smyth 2011). This effect may however be limited given the large volume of water in the marina.

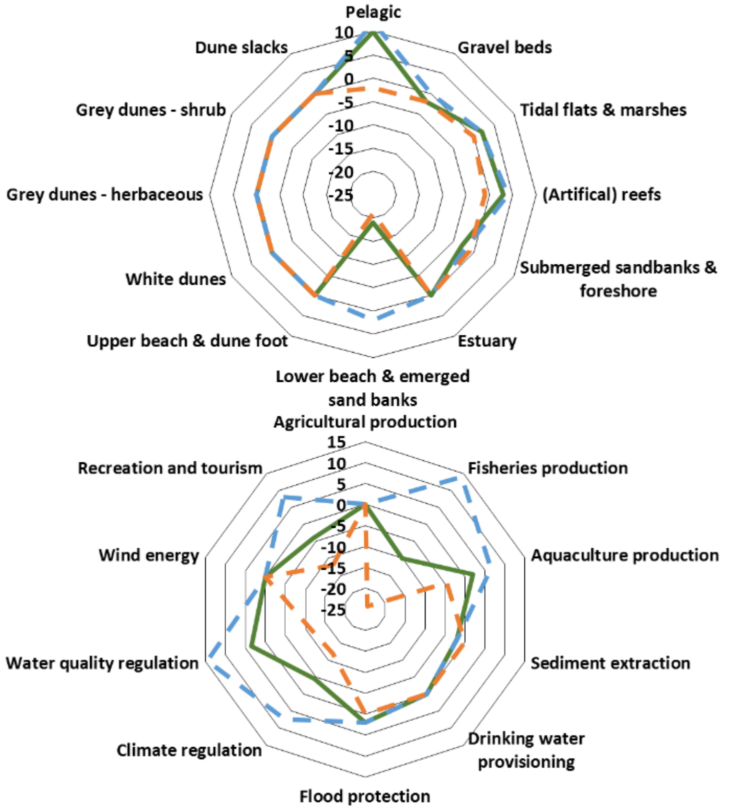

Figure 6.6 – Impact of the development of a marina resulting from the degradation of existing habitat of lower beach (orange), from the newly created habitat pelagic and artificial reefs (blue) and the combined effect of both (green). Left: effects on habitats, right: effects on ecosystem services.

6.5. Discussion

This chapter presents an approach that aims to support the integration of ecosystem services into impact assessments and provide inspiration for the development of alternative tools. The methodology takes an innovative step by evaluating changes in ecosystem processes, from which impacts on ecosystem services and habitats are assessed. We here discuss its merits, others than those related to the inclusion of ecosystem services in impact assessments as mentioned in earlier studies, and constraints in relation to current challenges in including ecosystem services in impact assessments.

6.5.1. Improved coverage of cross-sectoral and cumulative effects

Despite recent progress, cumulative effects remain poorly considered in impact assessments (Tallis et al. 2015). The inclusion of ecosystem services in impact assessments has been identified as an opportunity to better address cumulative effects because of the integrative framing of effects (Baker et al. 2013). However, this opportunity can only be benefitted from if the underlying ecosystem processes that produce ecosystem services and that drive interactions among ecosystem services are taken into account. Impact assessments have indeed been criticised for not sufficiently considering interactions between habitats and species and the role of ecosystem processes (Karlson et al. 2014; Bigard et al. 2017). Key of this methodology is to identify the multiple processes that underlie ecosystem services delivery and habitat development and assess how these processes alter under due to a project. Instead of evaluating individual aspects of the ecosystem (e.g. soil, groundwater), this methodology evaluates changes in a process by taking into account all the parameters that play a role in the process. For example, the effect of pollutants on benthic production may be limited due to dilution, but in combination with increased soil disturbance and turbidity this may affect ecosystem services such as water quality regulation and climate regulation (Figure 6.6). A limitation of the presented methodology, however, is that effects of interactions among processes and among changing parameters are not covered.

To be able to fully account for cumulative effects it is also necessary to take ecosystem processes into consideration when defining the study area, as cumulative effects often occur on larger spatial and time scales. The inappropriate delineation of the study area has notably been mentioned as one reason why cumulative effects remain poorly integrated in impact assessments (Baker et al. 2013; Karlson et al. 2014; Tallis et al. 2015). In the case-study, not only lower beach habitat is negatively impacted by the island (Figure 6.5) due to conversion to tidal flats and marshes but also upper beach and dune foot habitat which is located more inland. This impact is accounted for trough the influence of loss of lower beach (changing parameter) on the process of wind-driven sand supply, of which the upper beach habitat is dependent. On the longer term, this will eventually result in permanent changes in vegetation and species composition in the dune foot and may affect ecosystem services supply such as beach and dune recreation (Van der Biest et al. 2017a). The methodology allows to explicitly include large-scale or long-term processes as the mechanisms behind cumulative effects, such as habitat fragmentation. In the case-study, the creation of a large tidal flats and marshes area will have a greater positive effect on ecosystem services such as fisheries production due to the presence of another tidal flats area nearby (reduction of habitat fragmentation) than only the effect of an increase of habitat area.

6.5.2. A standardized framework to reduce resource-intensity

The integration of ecosystem services in impact assessments is hampered by the sometimes large data requirements to accurately assess ecosystem services (Baker et al. 2013) and the lack of a standardized methodology to identify relevant ecosystem services (Seppelt et al. 2012). The presented methodology offers a way to reduce time investments by allowing to make a selection of only the key ecosystem services that will be affected by a project and that require a more in-depth (data-driven) evaluation. Once the framework is elaborated for a certain ecosystem, the information an impact assessments practitioner has to provide is restricted to the expected changes in habitat and in the changing parameters that affect processes. This low-resource aspect is thus only valid for its application (Step 6). The elaboration of the tool at the ecosystem level (Steps 1 – 5), which is done prior to the application in an impact assessment, requires profound knowledge of the functioning of the ecosystem and its relationships to ecosystem services. The methodology would be most efficient in achieving its goal of facilitating the integration of ecosystem services in impact assessments if the preparatory steps (1-5) are developed in a standardized way and institutionally organized on the regional or national scale. Investments of impact assessments practitioners would then be restricted to the application of the methodology (Step 6) and a more detailed assessment of only the key ecosystem services that will be affected. The number of key ecosystem services that require further analysis could at least be reduced to 7 (construction of the marina) and 6 (reduced hydrological conditions in the lee of the island) out of 10, assuming that every affected ecosystem services would be studied more in detail.

An additional advantage of a standardized framework is that it allows to identify potential effects unbiasedly, without making an a priori selection of impacts that are believed to be (ir)relevant or for which data is scarce, potentially resulting in their overexploitation (Beaumont et al. 2007). The usage of a matrix-approach in this respect supports the identification of knowledge gaps. All changing parameters are positioned in relation to each process and likewise all processes to every habitat and ecosystem services, thus avoiding potential effects being overlooked and a false sense of certainty in the decision making process. A sensitivity analysis may help the decision maker to embrace this uncertainty by informing how strong effects with uncertainty would influence the final results.

6.5.3. Flexibility to support the development of alternatives

Once the preparatory steps are completed and the instrument is ready for application in impact assessments, it provides a flexible tool that can be used to support the development of ecosystem-based alternative scenarios. The practicality of the tool facilitates the iterative process of fine-tuning alternatives by a quick-scan assessment of expected effects and its transparency allows users to better understand consequences of changes and improve the design of alternatives. New habitat can also be evaluated, allowing to identify opportunities and widen the scope of avoiding negative impacts in current impact assessments practice (Goodstadt et al. 2010). The shift of focus from evaluating individual sectors to evaluating the common ground between sectors based on processes facilitates the debate among stakeholders and further enhances the development of ecosystem-based alternatives with shared objectives. Alternatives can thus be developed based on ecosystem functioning and on the interactions between ecological and socio-economic actors rather than on merely short-term goals of individual sectors.

A sensitivity analysis can be performed to identify those parameters and processes with largest impacts, for which it is most important to find alternative solutions. This can be done by changing the size and/or the direction of the related parameters and analysing how this affects the outcomes. The impacts of the marina on ecosystem services and habitats for example strongly depend on the degree to which hydrodynamics are affected. Disturbance of hydrodynamics due to the construction of the marina's jetties has an impact on multiple processes such as morphodynamics (changing sedimentation patterns around the jetties), benthic production (changing sediment composition + turbidity) and dune formation (sedimentation around the jetties). This affects the habitats upper beach and dune foot, submerged sand banks and foreshore, and ecosystem services such as recreation and protection against floods. An alternative scenario of the marina can thus be based on its location in comparison to the harbour jetties. The advantage of this method is that both the effects on other ecosystem processes and the potential impacts on habitats and ecosystem services are directly visualized in the matrix, without having to apply complex models, thus facilitating the development of alternatives.

6.5.4. Constraints and improvements

The method in its current form is in an early stage of development and could be improved in a number of ways. A score-system is used to describe impacts of changing parameters and processes on habitats and ecosystem services, because this supports quick and practical application and allows integrating effects for which quantitative information is not available. Especially for processes, detailed knowledge is sometimes lacking as they are more difficult to measure or evaluate than static properties. By using qualitative scores instead, these are avoided being ignored. A qualitative outcome however does not provide information on the precise extent of the impact or on its socio-economic significance, so that ecosystem services cannot be compared in quantitative terms. For example, one of the main purposes of the island is to protect the coast against floods by reducing tidal and wave energy. However, the results indicate that the qualitative score for fisheries production is higher than that of flood protection (Figure 5). One way to deal with this is to describe impacts using mathematical formula and/or expressing the final results in a common unit, such as a monetary value (for ecosystem services) or an existing ecological status classification (e.g. such as from the Water Framework Directive). The possibility to quantify and to have an estimate of the uncertainty may also extend the scope of the methodology, which is now more a quick-scan screening of potentially relevant impacts as a starting point for an ecosystem services-inclusive impact assessment, to a stand-alone instrument to support a final decision. Given that the lack of quantitative assessments has been identified as a shortcoming in the early years of impact assessments (Drayson et al. 2017), the feasibility to mathematically describe impacts is worthwhile investigating. Another advance lies in distinguishing between the effects on the quality and on the quantity of habitats and ecosystem services. This would, however, make the methodology more complex and difficult to interpret, while its main aim is to be an indicative instrument as a starting point for a more sound model-based assessment.

A further improvement to be explored is the possibility to set thresholds on the impacts of changing parameters on processes, so that the rate of a process will only be affected at a certain magnitude of the changing parameter. This could make the methodology more site-specific, and may also refine the assessment of cumulative effects. However, similar as for the usage of quantitative relationships, this would render the methodology more complex and change its scope.

Despite these constraints, the application of the methodology on the case-study shows the potentials of focussing on evaluating ecosystem processes and of the presented approach to advance ecosystem services-inclusive impact assessments, and potentially classic impact assessments.

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7

General discussion and synthesis

7.1. Introduction

The central focus of this thesis is on developing ecosystem-functioning based tools for assessing and managing ecosystem services. Different approaches were explored including the use of a structural indicator, Bayesian networks based on ecological models and process-based indicators. Two novel process-based approaches were developed to align ecosystem services and biodiversity targets in spatial planning and to integrate ecosystem services into impact assessments.

In the first part of this conclusion, the study is framed in relation to the evolution in ecosystem services research in the past decade. The second part highlights the main findings and achievements of this research with reference to the initial research questions. The chapter ends with suggestions for future research and management recommendations on the implementation of ecosystem services in spatial planning and environmental management.

7.2. Positioning of the thesis in relation to trends in ecosystem services research

The study was performed between 2010 and 2018, covering the era from when the concept of ecosystem services had already become a research domain in itself (cfr. the foundation of the journal 'Ecosystem Services' in 2012 and the 'International Journal of Biodiversity Science, Ecosystem Services and Management' in 2010) and applications were mostly limited to awareness-raising and accounting (Landuyt 2015), over the adoption of the Aichi biodiversity targets including ecosystem services and the founding of the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES), up to the first steps of applying the concept in decision-informing (e.g. Flanders Regional Ecosystem Assessment; Estimation of the benefits of NATURA2000, Broekx et al. 2013) and decision-making (e.g. Forest Management Plan in the land dune region in the province of Antwerp, Seynaeve J. (2015) and Van der Biest et al. (2015); cases from the ECOPLAN-project, www.ecosysteemdiensten.be). This part frames the work that was performed in this thesis in relation to some trends in ecosystem services research.

At the start of this doctoral study in 2010, one of the main conclusions in ecosystem services research was that there is a need to comprehend interactions between ecological functioning and ecosystem services (Nicholson et al. 2009; Verburg et al. 2009; Seppelt et al. 2011), but that inter- and transdisciplinary approaches were scarce (Potschin and Haines-Young 2011), potentially impeding the inclusion of ecosystem services in management and decision-making (de Groot et al. 2010). Proxy-based methods linking structural properties to ecosystem services provision were at that time commonly applied on different scales and for various purposes (Maes et al., 2011, 2012; Nedkov and Burkhard, 2012; Schneiders et al., 2012), in spite of the multiple researches that demonstrated their poor quality (Kienast et al., 2009; Eigenbrod et al., 2010; Lautenbach et al., 2011; Geijzendorffer and Roche, 2013; Hou et al., 2013). The **second chapter** of this thesis assesses the accuracy of proxy methods based on structural indicators (in this case land use) to map ecosystem services.

The complexity behind the production and consumption of ecosystem services and the lack of transdisciplinarity across disciplines and between science and society are other reasons why simple proxies have long been used to assess ecosystem services (Reyers et al. 2009). The very nature of ecosystem services requires understanding of both ecological (mostly supply-side) and socio-economical (mostly demand-site) functioning. Apart from this complexity, ecosystem services are also naturally diverse, touching upon different topics within the ecological and socio-economic fields. Regulating services for example can be related to soil, hydrological, biological, climatic science, etc.. While research points out the need to engage with this complexity and transdisciplinarity in order to be effective in management (Nicholson et al. 2009; Grêt-Regamey et al. 2017a), practitioners themselves demand pragmatic tools that enable rapid application (Villa et al. 2014). The **third chapter** constitutes an explorative study on the usage of Bayesian networks to deal with this ambiguity. Bayesian networks were at that time an established methodology for medical diagnosis (Landuyt 2015) and their advantages in ecological applications were increasingly being recognized (Aguilera et al. 2011) but not yet fully studied in detail for ecosystem services (Landuyt et al. 2013).

The release of several studies pointing to the need for scientifically underpinned methods (de Groot et al. 2010; Braat and de Groot 2012; Seppelt et al. 2012) contributed to a reorientation of efforts towards more and/or better integration of ecosystem knowledge in ecosystem services assessments. Emphasis was more and more put on distinguishing between final benefits for human well-being and processes underpinning the ecosystem services that generate those benefits (Boerema et al. 2016; La Notte et al. 2017), which were mentioned to be often mixed up, causing confusion and impeding application of the ecosystem services paradigm in practice (Nahlik et al. 2012). Recognizing, understanding and modelling the underlying processes has increasingly been acknowledged as key to predicting changes in ecosystem services (Nicholson et al. 2009; Lavorel et al. 2017). A similar evolution towards a focus on underlying processes has also taken place in biodiversity science. Research shows that conservation efforts are likely to be more beneficial for biodiversity if ecological processes are considered (Bennett et al. 2009; Klein et al. 2009; Castro et al. 2015; Pettorelli et al. 2018) and international standards are now being adapted to include processes (Watson et al. 2016). As processes are the drivers of both biodiversity and ecosystem services, they enable to find common ground between biodiversity conservation and management for ecosystem services. **Chapters 4 – 6** build on this rationale in which ecosystem processes are the nexus of environmental management.

Transdisciplinarity in ecosystem science also refers to integration of biodiversity conservation with socio-economic demands (Reyers et al. 2009). Research and management in ecosystem services and biodiversity conservation often have a biased focus either on ecosystem services or on biodiversity. The notion of ecosystem services arose as a vehicle to increase awareness that nature, besides biodiversity, also provides many benefits to humans. The concept thus aims to demonstrate those additional benefits, which explains why biodiversity per se is not included in common ecosystem services classifications (e.g. MEA 2005; TEEB 2010; CICES, EEA 2016), except if it is related to human well-being (e.g. habitat service of supporting different life stages of fish). Focussing merely on the use-side of the ecosystem however has proven to result in major losses of biodiversity (Cardinale et al. 2012). Despite it being high on the environmental policy agenda (e.g. Strategic Goal D of the Aichi Biodiversity Targets; IUCN's guidelines for protected areas including (non-)consumptive use in delineated zones; the Flemish Integral Interrelation and Support Network IVON), the incorporation of conservation and multiple societal objectives into spatial planning (see e.g. Adriaensen et al. 2005; RAMSAR) is still not yet common practice (Guerry et al. 2015). This research resulted in a set of hands-on instruments to further advance the integration of biodiversity conservation with socio-economic demands in spatial planning (**Chapter 5**) and in impact assessments (**Chapter 6**).

7.3. Main findings and contributions

7.3.1. Ignoring ecosystem functioning results in inaccurate ecosystem services predictions

Research question 1
Does a highly detailed classification increase the accuracy of land use as a single structural indicator of multiple ecosystem services?

Conclusion
Land use is insufficiently accurate to predict multiple ecosystem services, even when using a highly detailed thematic resolution.

As mentioned earlier, criticism had raised on the reliability of proxy methods such as land use (Eigenbrod et al. 2010). Research (Burkhard et al. 2012; Vihervaara et al. 2012) however suggested that increasing the thematic resolution (number of land use classes) based on an inclusion of environmental conditions in the description of the land use class might increase the accuracy of the method. The first part of this study (**Chapter 2**) tested this hypothesis and revealed that land use remains a poor indicator of ecosystem services even with increasing thematic resolution. Ecosystem services maps based on land use for three different thematic resolutions were statistically compared with maps of ecosystem services produced by biophysical models. Although an increase in the number of classes did improve correlations slightly for regulating ecosystem services, they remained weak. Part of this correlation may even be attributed to the usage of the more detailed land use map as a basis for the quantitative assessment, especially when no other land use variables are used besides this map. Correlations were higher for provisioning ecosystem services as they are less dependent of environmental conditions and for which the environment is often manipulated to optimize production (e.g. groundwater level control in agricultural areas). Despite significant correlations, probability density analysis showed that the number of erroneous predictions was high for both provisioning and regulating ecosystem services. This chapter underpins the need to consider ecosystem functioning and dynamic processes when using ecosystem services in decision-making.

7.3.2. Bayesian networks are suitable to reduce the accuracy – practicablity trade-off in ecosystem services assessments

Research question 2
How useful are Bayesian networks to develop practical and scientifically underpinned ecosystem services assessment tools?

Conclusion
Bayesian networks have several advantages that make them appealing for developing ecosystem services assessment tools: (1) they allow to integrate knowledge on ecosystem functioning, (2) they are able to combine knowledge from different sources, (3) they can be used to create scenarios optimizing multiple ecosystem services and (4) they remain practical and transparent facilitating application at different levels of decision-making. However, the difficulty to include feedbacks makes them less suitable to model dynamic ecosystem processes and interactions among ecosystem services.

As ecosystem services cover a wide variety of disciplines, the techniques for assessing ecosystem services likewise differ strongly (Seppelt et al. 2011). For some services, knowledge is advanced and numerical models are available (e.g. soil erosion). At the other end, there are services for which research is still in its infancy and stakeholder- or expert-based assessments are frequently used (e.g.

many cultural services). Bayesian networks (**Chapter 3**) allow to combine knowledge from different sources by converting it into cause-effect flows between environmental properties and ecosystem services (Landuyt et al. 2015). Quantitative data can be combined with expert elicitation, reducing the trade-off between accuracy and completeness (Figure 1.5).

Bayesian networks can be run in a belief or in a decision mode, making them suitable for multiple aspects in ecosystem management. They allow to assess impacts of a certain scenario on ecosystem service delivery (belief mode), but they can also be used to spatially allocate land use or management interventions for service optimization and to develop a benchmark to which choices can be compared (decision mode). The differences between today's land use and the benchmark uncover discrepancies between land use and its biophysical suitability.

The explicit representation of the links between environmental properties and ecosystem services in a graphical network makes the tool more transparent and user-efficient in comparison with complex ecological models (Seppelt et al. 2009). This may also facilitate the iterative science-management process (Grêt-Regamey et al. 2017a) where scenarios can be developed and revisited. To further support integration of ecosystem services in decision-making, it is recommended to embed Bayesian software within software which practitioners are familiar with, such as GIS (e.g. as further elaborated collaboratively in Landuyt et al. 2015) or web-based applications (e.g. Jongsawat and Premchaiswadi 2009). Another advantage of Bayesian networks in terms of user-friendliness is the possibility to make predictions, although with a greater uncertainty, when certain input data is unavailable. This helps to avoid that ecosystem services for which knowledge is lacking are omitted from the analysis, resulting in biased outcomes (Seppelt et al. 2012).

Although this was not elaborated upon in the research, an important advantage which make Bayesian networks particularly suitable for ecosystem services assessments is the possibility to take into account uncertainty. Addressing uncertainties is key in environmental decision-making (Grêt-Regamey et al. 2013; Long et al. 2015) and identified as one of the main challenges ecosystem services research is faced with today (Landuyt et al. 2015; Grêt-Regamey et al. 2017b). Uncertainties also provide additional information that can influence the desirability of a scenario (i.e. low in case of high uncertainty). Uncertainty can be derived from available models, or it can be assigned based on validation of the final output of the model. The rigour with which validation is performed determines for which purposes the model can be used. In case it is meant for decision-making, validation should be based on field measurements and go beyond the qualitative validation as done in this explorative study. A critical note however is that the influence of uncertainty on making environmental decisions and the potential abandoning of a choice with large uncertainties should be balanced with the way this is done when making economic decisions, where uncertainty is often accepted as is.

Although the complexity of ecosystem functioning can be incorporated in Bayesian networks to a certain extent, some simplification and thus loss of information is inevitable due to the obliged discretization of the values of the input and output parameters. An issue which may impede implementation of tools that go beyond the use of static properties as proxies and account for ecological complexity, is the amount of data that is needed to run the model, especially if multiple ecosystem services are included. Although Bayesian networks can deal with missing data, this should be reduced to a minimum in order to restrain uncertainty and achieve realistic results. An important drawback of Bayesian networks is the difficulty to include feedbacks and thus spatial and temporal interactions between ecosystem conditions and services. This makes them less suitable as a modelling environment for dynamic ecological processes. This could partly be overcome by adapting the structure of Bayesian networks (e.g. Halabi et al. 2017), but conflicts with shortcomings of Bayesian networks related to an exponential increase in the time needed for belief updating as the number of nodes increases, and memory availability (Landuyt 2015).

The prototype tool as developed in Chapter 3 was applied to support the participatory process of developing a shared vision on the management of a forest area among owners, users and neighbours in the community of Balen, Belgium (Seynaeve J. 2015; Van der Biest et al. 2015). This process supported the change in mindset of the forest owners to widen the scope of management of their private piece of forest from a focus on fire wood production, recreation and/or nature to an inclusion of multiple aspects such as carbon sequestration, pollination, landscape value, … Efforts are now being spent to plant mixes of native tree species suitable for the specific conditions of the location within the forest and to increase diversity and landscape attractiveness along forest edges. Further plans are to use the vision as a guideline to develop a forest management plan.

7.3.3. Processes are the driving mechanisms of ecosystem services and biodiversity and therefore key in environmental management

Research question 3
How can a focus on ecosystem processes advance the integration of ecosystem services at different levels of environmental management?

Conclusion
Processes are the underlying mechansisms for both biodiversity and ecosystem services and the drivers of interactions between them. Considering processes in environmental management allows to take into account these dynamic relations and to more explicitly link biodiversity with human well-being. They thus constitute the key tool to align biodiversity conservation with ecosystem services in spatial planning.

Changes in processes can affect supply of ecosystem services on different time and spatial scales. They are early indicators of changes in ecosystem services and it is therefore crucial to take them into account in spatial planning.

Ecosystem services can be produced by processes that took place in the past (e.g. historic carbon stocks in wetlands or coastal dunes formed hundreds of years ago), or by processes that are taking place now (e.g. yearly carbon sequestration or sand dynamics in coastal dunes). Assessing ecosystem services based solely on static properties may thus lead to an under- or overestimation of the service (Boerema et al. 2016). A typical example is the usage of the current stock of carbon in the soil to assess the climate regulation capacity of an ecosystem. However, to estimate the true amount of carbon which is being sequestered in and potentially emitted from the soil per year, gas flux measurements are needed. Especially in wetlands with large historic carbon stocks and methane formation, this may lead to large errors. Management for ecosystem services is thus more effective if underlying processes are considered (Nicholson et al. 2009; Lavorel et al. 2017).

The capacity to manage ecosystem services through the regulation of ecosystem processes was first tested using the case of sand transport in coastal dunes. **Chapter 4** demonstrates how multiple ecosystem services are affected by a human-induced disruption of the eolian driven process of sand transport between beach and dunes. The results show that redynamisation of coastal dunes may create up to ~50% more economic benefits in terms of ecosystem services and that these benefits are mostly on account of recreation and coastal safety. This is explained by the visitors' appraisal of the characteristic dune landscape with freshly deposited bare sand and marram grass (De Nocker et al. 2015), and the capacity of dunes to grow with sea level rise as sand gets trapped by dune vegetation (Feagin et al. 2015). It is recommended to maintain or restore connectivity between beach and dunes and allow eolian forces to move sand in order to safeguard ecosystem services. This chapter illustrates how ecosystem processes are early indicators for ecosystem services and underpins the need to consider them when planning for ecosystem services.

Focussing on processes provides a tool to align ecosystem services with biodiversity conservation in spatial planning

The approach in which ecosystem services are managed through ecosystem processes was extrapolated to integrate ecosystem services with biodiversity conservation in spatial planning (**Chapter 5**) and in environmental impact assessments (**Chapter 6**).

Chapter 5 presents an analytical tool to find common ground between biodiversity conservation and socio-economic needs in spatial planning. The method allows to incorporate multiple benefits and actively involve stakeholders early in the planning process, which is acknowledged as key to environmental planning by a growing body of literature (Hein et al. 2006; Ban et al. 2013; Beunen et al. 2013; Partidário and Gomes 2013; Bennett et al. 2017; Grêt-Regamey et al. 2017a). The method identifies the biophysical, ecological and anthropogenic processes that underlie ecosystem services and the creation of habitats, and prioritizes these based on (1) their importance in establishing (multiple) habitats and ecosystem services and (2) socio-economic needs as appraised by stakeholders. Management strategies are formulated for each process based on trade-offs and synergies among ecosystem services and biodiversity values. This provides guidance in the debate on prioritizing when trade-offs are inevitable, which is identified as key research area for more effective integration of biodiversity and ecosystem services in spatial planning (Jennings and Rice 2011; Lester et al. 2013). Focusing on processes shifts the debate from an emphasis on conflicts between sectors to a common goal of multifunctionality.

The method is based on expert elicitation, reducing its accuracy (Figure 1.5) and reliability to support decision-making. However, as mentioned, processes are much more difficult to capture than static properties. A quantitative approach may increase reliability but would lead to less services being considered. This, in turn, reduces usefulness in practice and unbiasedness in decision-making. Further needs for improvement of the method are an inclusion of non-linear effects of processes on habitats and ecosystem services (e.g. cumulative effects) and interactions between processes that reduce or increase the impact of a process on a habitat or ecosystem service. For these reasons, the method in its current form is not suitable for spatially explicit assessments and allocation of management measures but merely to identify the key processes environmental management should focus on to achieve habitat and ecosystem services targets, thus as a supportive tool for early, more strategic stages of spatial planning.

A process-based approach advances the integration of ecosystem services in impact assessments

Chapter 6 takes this process-based rationale a step further by adding a methodology for impact assessment. Including ecosystem services has several potentials to deal with current shortcomings in environmental impact assessment, such as an explicit representation of benefits for human well-being and a cross-sectoral consideration of impacts. The methodology aims to provide a solution to some additional challenges related to the application of ecosystem services in impact assessments (Bina O. 2007; Hooper et al. 2014; Baker et al. 2013; Partidário and Gomes 2013; Bigard et al. 2017). Cross-sectoral, cumulative effects can only be accounted for if the underlying ecosystem processes are taken into account and if they are considered on sufficiently large spatial and temporal scales (Tallis et al. 2015). Instead of evaluating individual aspects of the ecosystem (e.g. soil, groundwater), changes in a process are evaluated by assessing how the parameters that play a role in the process are affected by a project, and couples this with effects on ecosystem services and habitats. The methodology allows to explicitly include large-scale or long-term processes as the mechanisms behind cumulative effects, such as habitat fragmentation. It reduces the intensive resource requirements typical for quantifying ecosystem services by identifying the priority services that are affected by the project and that require a more detailed assessment. A flexible and user-friendly methodology as presented here enables the iterative process of developing and fine-tuning ecosystem-based alternatives, which would be more difficult with complex discipline-specific models and is mentioned as another shortcoming of current impact assessments (Partidário and

Gomes 2013). By focussing on the role of processes, alternatives are developed based on ecosystem functioning and on the interactions between ecological and socio-economic actors rather than on merely short-term goals of indivual sectors.

Although a quantitative approach is preferred over a qualitative approach, using qualitative scores has the advantage that it allows to take into account effects for which no quantitative information is available (Figure 1.5). As mentioned earlier, detailed knowledge on ecological processes is often lacking as they are more difficult to measure or evaluate than static properties. By using qualitative scores instead, these are avoided being ignored.

The methods in Chapters 5 and 6 were developed in the frame of the Vlaamse Baaien project in the Belgian coastal zone (Projectgroep Vlaamse Baaien 2012). This resulted in the creation of a shared vision on a healthy coastal ecosystem and sustainable ecosystem services delivery. A set of management recommendations was formulated that constitute an integrated framework for nature development in the coastal ecosystem, to which future project developments and measures can be evaluated using the accompanying impact assessment framework. Based on this research, drastic changes in coastal zone management are recommended in light of climate change, i.e. to develop strategies to advance or retreat rather than to maintain the current shoreline. The research was published as the Ecosystem Vision for the Flemish Coast (Van der Biest et al. 2017a, b).

7.4. Some future directions

7.4.1. Research

Decision-makers need practical tools to integrate ecosystem services into spatial planning and environmental management. An important criteria is that they want confidence in the tools, which is created by scientific underpinning and by the possibility to understand how their decisions affect the ecosystem (to avoid the so-called ‘black box’ effect). Besides the development of hands-on instruments that allow to deal with this ambiguity, some additional recommendations to promote ecosystem services operationalization follow from this research: (1) Decision-makers should be involved in the development of assessment tools as they know best what they need to make them usable in practice. Researchers likewise should be involved in planning practice to help overcome obstacles related to the complexity of ecosystem services assessments and guide in the interpretation of the outcomes. Training courses can be organized to familiarize planners with the concept of ecosystem services and with the tools. Processes on the interface between science and management and science and policy should be iterative so that decision-makers fully understand the consequences of their choices and strategies can be evaluated, adapted and reevaluated. (2) The availability of an integrated spatial data platform makes it more practical for practitioners to gather data that is needed to quantify ecosystem services, thus overcoming reluctance towards including ecosystem services related to intensive data requirements. (3) A standardized reference framework on the regional or national level to assess ecosystem services would help to avoid inconsistencies between methodologies, which is believed to undermine credibility of the ecosystem services concept and slow down implementation (Seppelt et al. 2012), or guide the selection among alternative methodologies. The Flemish Regional Ecosystem Assessment (2014-2018), like other national or regional ecosystem assessments, constitutes such a reference framework in which methodologies to assess ecosystem services are described. However, it does not provide ready-to-use instruments such as those developed in this research, that allow to bring theory closer to practice. A standardized ecosystem services framework should include such hands-on tools that are fit for different management endpoints and ecosystem-specificities and that are continuously updated as scientific knowledge advances. The toolbox coming forth from the ECOPLAN-project (Vrebos et al. 2017), for example, has paved the way to establish such a spatially explicit assessment framework in Flanders. (4) Efforts should continue to be spent in convincing practitioners to take the extra mile for an ecosystem services approach. Important herein is to pinpoint the added value of including ecosystem services in comparison with existing approaches.

Although the accuracy of ecosystem services assessments increases when they are based on ecosystem functioning models, the most exact way of assessing ecosystem services is by monitoring (Eigenbrod et al. 2010). Up to date very few monitoring schemes exist that include variables allowing for a precise assessment of ecosystem services. One explanation is that ecological processes which drive ecosystem services are more difficult and costly to measure than static environmental conditions, justifying the use of less reliable methods such as expert elicitation in particular cases. For climate regulation it is for example much easier to measure carbon stocks than fluxes. Sound environmental management should embrace the complexity of ecosystem functioning and invest in research and monitoring to improve understanding and decrease uncertainty related to using proxies. Environmental monitoring schemes should comprise variables specifically identified to assess ecosystem services and the underpinning ecological processes. Monitoring processes has the additional advantage that changes in ecosystem services can be detected in an early phase, avoiding drastic changes to take place in the environment.

While this study illustrates how common ground between biodiversity and ecosystem services can be identified, an important knowledge gap remains on the precise mechanisms that define the relationships between biodiversity and ecosystem services. An underrepresentation of ecosystem services in environmental monitoring is likely to be related to this lack of understanding. More research is needed that investigates links between species richness, functional diversity (Díaz et al. 2007) and ecosystem services, and monitoring schemes should include specific variables for this study. Vice versa, newly gained insights in the relationships between biodiversity and ecosystem services can improve existing monitoring (Oliver et al. 2015).

This research presents hands-on methodologies to integrate ecosystem services in spatial planning but also aims to provide inspiration for the development of new approaches by demonstrating the benefits of considering ecosystem functioning and processes. Another crucial research frontier in ecosystem services modelling is the inclusion of sociological dynamics that drive the demand for ecosystem services (Rieb et al. 2017). In this study, social values were accounted for using stakeholders' knowledge on the importance of the ecosystem to deliver certain ecosystem services (Chapters 5, 6) and stakeholders' preferences in terms of an average value score (Chapters 3, 5). However, the demand for ecosystem services may change in time, depending on the stakeholder group and on the location of the beneficiaries (Rieb et al. 2017). This will strongly affect results of assessments and desirability of planning scenarios and should be taken into account to advance current ecosystem services models.

7.4.2. Management

The concept of ecosystem services provides a tool to integrate the interests of different stakeholder groups in spatial planning and management of the ecosystem. It enables participatory development of planning processes and the creation of a shared vision, which has been identified as a prerequisite for successful conservation of nature values in spatial planning. It also allows to explicitly demonstrate the benefits for humans of the preservation of ecosystems, thus finding more support for biodiversity conservation and for ecosystem-based solutions that reduce vulnerability of humans to environmental changes.

Simplified proxies such as land use allow for rapid assessments of ecosystem services as they are straightforward and don't require large amounts of data. However, they are insufficiently precise to be used in decision-making and their usage should therefore be limited to deliberate purposes for which they constitute added value over other, more complex methodologies.

This research proves that the accuracy of ecosystem services predictions becomes higher when ecosystem functioning is considered, and that this is crucial for more effective environmental management. As ecosystem processes constitute the driving forces behind the delivery of ecosystem services, management should target the maintenance of ecosystem processes to guarantee long-term supply of ecosystem services and support of human well-being as end-products of these processes.

An understanding of how ecosystem functioning underpins ecosystem services and biodiversity in combination with a stakeholder-based identification of socio-economic demands provides unbiased guidance in choosing among trade-offs and in prioritizing management goals. Focussing on the underlying processes that produce ecosystem services and that support biodiversity enables the identification of common ground between the formerly disconnected and alternative targets of biodiversity conservation and socio-economic demands. It also helps to shift the debate around conflicts between sectors to a common goal of multi-functionality shaped by these processes that have cross-sectoral effects.

The results from this study demonstrate the advantages of a process-based approach to integrate ecosystem services in environmental management and spatial planning. The methodologies elaborated upon in Chapters 5 and 6, however, use a qualitative scoring system to describe relationships between parameters, processes and ecosystem services and habitats. Because qualitative assessments are less accurate (Figure 1.5), the applicability of this method is restricted to more strategic purposes, as a tool to support communication in spatial planning processes or as a pre-screening tool.

This research shows the capacity of Bayesian networks to deal with the ambiguity that characterizes the demand for ecosystem services assessment tools, i.e. integration of the complexity of ecosystem functioning while remaining practicable in use. Nevertheless, despite their user-friendliness when modelling multiple ecosystem services, the more variables are included within the model, the more data a user needs to provide. While Bayesian networks can be used to support decision-making, time and means need to be invested to do so.

A final recommendation following from this study is that ecosystem services and biodiversity should not be regarded as substitutes of each other but included as separate study objectives in environmental management with different evaluation methodologies. Yet, common ground between both should be evidenced as much as possible in order to maximize the scope and effectiveness of nature and ecosystem services conservation. Ecosystem services are useful as a complementary framework to define targets for ecosystem processes besides targets for specific species or habitats, in order to optimize both biodiversity values and human well-being.

7.5. References

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Appendices

Appendix A

Appendix tables of Chapter 2

Table A.1 – Conversion of the ESLUC classification system into the CORINE and governmental classification system respectively

	LAND USE CATEGORIES		
	ESLUM	**Corresponding CORINE category**	**Corresponding governmental category**
Urban	Building	Continuous urban fabric	Residential and commercial fabric
	Non-built up paved surface	Continuous urban fabric	Residential and commercial fabric
	Garden, grass or bush	Discontinuous urban fabric	Residential and commercial fabric
	Garden, trees	Discontinuous urban fabric	Residential and commercial fabric
	Park, grass or bush	Green urban areas	Park
	Park, trees	Green urban areas	Park
	Bare ground	Discontinuous urban fabric	Business and industrial site
	Cemetery	Green urban areas	Park
	Railway	Road and rail networks and associated land	Infrastructure
	Road	Road and rail networks and associated land	Infrastructure
	Dike	Road and rail networks and associated land	Agricultural grassland
	Sports- or recreational building	Sport and leisure facilities	Recreation and sports facilities
	Sports or recreational terrain	Sport and leisure facilities	Recreation and sports facilities
	Sports or recreational terrain, grass or bush	Sport and leisure facilities	Recreation and sports facilities
	Sports or recreational terrain, trees	Sport and leisure facilities	Recreation and sports facilities
	Golf terrain	Sport and leisure facilities	Recreation and sports facilities
	Quarry	Mineral extraction sites	Business and industrial site
	Dump	Dump sites	Business and industrial site
Agricultural	Cropland	Non-irrigated arable land	Cropland
	Corn	Non-irrigated arable land	Cropland
	Floriculture + nursery	Non-irrigated arable land	Cropland
	Fruit	Fruit trees and berry plantations	Cropland
	Orchard	Fruit trees and berry plantations	Cropland

	LAND USE CATEGORIES		
	ESLUM	**Corresponding CORINE category**	**Corresponding governmental category**
	Greenhouse	Non-irrigated arable land	Cropland
	Fallowland	Non-irrigated arable land	Cropland with an environmental target
	Permanent cultivated grassland	Pastures	Agricultural grassland
	Temporary cultivated grassland	Pastures	Agricultural grassland
	Forested cultivated grasland	Pastures	Agricultural grassland (with an environmental or a biodiversity target)
	Natural grassland, agricultural use	Pastures	Agricultural grassland (with an environmental or a biodiversity target)
Forest & semi-natural	Mixed deciduous forest	Broad-leaved forest	Multifunctional forest
	Mixed deciduous forest, nature management	Broad-leaved forest	Forest (biodiversity management)
	Mixed coniferous forest	Coniferous forest	Multifunctional forest
	Mixed coniferous forest, nature management	Coniferous forest	Forest (biodiversity management)
	Mixed forest	Mixed forest	Multifunctional forest
	Mixed forest, nature management	Mixed forest	Forest (biodiversity management)
	Alder	Broad-leaved forest	Multifunctional forest
	Ash	Broad-leaved forest	Multifunctional forest
	Beech	Broad-leaved forest	Multifunctional forest
	Birch	Broad-leaved forest	Multifunctional forest
	Linden	Broad-leaved forest	Multifunctional forest
	Poplar	Broad-leaved forest	Multifunctional forest
	Willow	Broad-leaved forest	Multifunctional forest
	Oak	Broad-leaved forest	Multifunctional forest
	Pine	Coniferous forest	Multifunctional forest
	Dry heathland	Moors and heathland	Heathland (biodiversity management)
	Wet heathland	Moors and heathland	Heathland (biodiversity management)
	Natural grassland (excl. valley and marsh natural grassland)	Natural grassland	High nature value grassland (biodiversity management)
	Other bush	Transitional woodland-shrub	High nature value grassland (no biodiversity management)
	Other shrub	Transitional woodland-shrub	Multifunctional forest
	Bare land dune	Moors and heathland	Beaches and dunes (biodiversity management)
	Vegetated, non-forested land dune	Moors and heathland	Beaches and dunes (biodiversity management)
	Forested land dune	Mixed forest	Multifunctional forest
Wetland	Alder brook forest	Inland marshes	Marshland (biodiversity management)
	Other valley and marsh forest	Inland marshes	Marshland (biodiversity management)
	Fen	Inland marshes	Marshland (biodiversity management)
	Reed	Inland marshes	Marshland (biodiversity management)
	Valley and marsh bush	Inland marshes	Marshland (biodiversity management)

	LAND USE CATEGORIES		
	ESLUM	**Corresponding CORINE category**	**Corresponding governmental category**
	Valley and marsh natural grassland (excl. reed)	Inland marshes	Marshland (biodiversity management)
	Valley and marsh shrub	Inland marshes	Marshland (biodiversity management)
	Vegetated intertidal flat ('schorre')	Intertidal flats	Mud flat and salt marshes
Water body	Navigable water course	Water courses	Water
	Non-navigable watercourse	Water courses	Water
	Pond	Water bodies	Water
	Water basin	Water bodies	Water
	Dock	Water bodies	Port area

Table A.2– Capacity of the ESLUC land use types to deliver ecosystem services

ESLUC land use	Crops	Livestock	Wood fuel	Global climate regulation	Groundwater recharge	Nutrient retention	Denitrification	Pollination
Building	0	0	0	0	0	0	0	1
Non-built up paved surface	0	0	0	0	0	0	0	0
Garden, grass or bush	2	0	0	1	4	2	2	3
Garden, trees	0	0	3	2	3	2	3	3
Park, grass or bush	0	1	0	1	4	2	2	4
Park, trees	0	0	3	2	3	3	3	2
Bare ground	0	0	0	0	4	0	0	0
Cemetery	0	0	1	1	1	1	0	2
Railway	0	0	0	0	0	0	0	1
Road	0	0	0	0	0	0	0	0
Dike	0	3	1	1	0	2	2	2
Sports- or recreational building	0	0	0	0	0	0	0	0
Sports or recreational terrain	0	0	1	1	0	0	0	0
Sports or recreational terrain, grass or bush	0	0	0	1	4	2	2	2
Sports or recreational terrain, trees	0	0	3	2	3	2	3	2
Golf terrain	0	0	1	1	4	2	2	2
Quarry	0	0	0	0	3	0	0	0
Dump	0	0	0	0	0	0	0	0
Cropland	5	0	0	1	2	0	2	1
Corn	5	0	0	1	1	0	1	1
Floriculture + nursery	0	0	0	1	2	0	2	3
Fruit	5	0	0	1	2	1	2	4
Orchard	5	1	2	1	2	1	2	4
Greenhouse	5	0	0	0	0	0	0	0
Fallowland	0	0	0	1	2	1	2	4
Permanent cultivated grassland	0	3	0	2	3	1	2	0
Temporary cultivated grassland	0	2	0	2	3	1	2	0
Forested cultivated grasland	2	3	1	2	3	1	2	3
Natural grassland, agricultural use	0	2	0	2	3	2	2	4
Mixed deciduous forest	0	0	2	3	3	3	3	2
Mixed deciduous forest, nature management	0	1	1	4	3	3	4	3
Mixed coniferous forest	0	0	4	2	2	2	2	1
Mixed coniferous forest, nature management	0	0	1	3	2	2	3	1
Mixed forest	0	0	3	3	3	3	3	2
Mixed forest, nature management	0	1	1	4	3	3	4	3
Alder	0	0	3	2	3	3	2	2
Ash	0	0	3	2	3	3	2	2
Beech	0	0	3	2	3	3	2	1
Birch	0	0	3	2	3	3	2	2
Linden	0	0	3	2	3	3	2	3
Poplar	0	0	3	2	3	3	2	2
Willow	0	0	3	2	3	3	2	2
Oak	0	0	3	2	3	3	2	2
Pine	0	0	3	2	2	2	2	1
Dry heathland	0	1	0	2	4	1	1	3
Wet heathland	0	0	0	4	1	2	1	3
Natural grassland (excl. valley and marsh natural grassland)	0	1	0	3	3	2	2	4
Other bush	0	0	0	3	3	3	2	4
Other shrub	0	0	1	3	3	3	3	3
Bare land dune	0	0	0	0	5	0	0	0
Vegetated, non-forested land dune	0	1	0	1	4	1	1	2
Forested land dune	0	1	2	2	3	2	1	1
Alder brook forest	0	0	1	5	3	4	4	2
Other valley and marsh forest	0	0	1	5	3	4	4	2
Fen	0	0	0	5	1	4	4	2
Reed	0	0	0	4	3	4	4	2
Valley and marsh bush	0	0	0	4	3	3	3	4
Valley and marsh natural grassland (excl. reed)	0	0	0	4	3	3	3	4
Valley and marsh shrub	0	0	1	4	3	4	4	3
Vegetated intertidal flat ('schorre')	0	1	1	3	0	3	3	3
Navigable water course	0	0	0	1	3	1	2	0
Non-navigable watercourse	0	0	0	2	3	2	3	1
Pond	0	0	0	3	3	3	2	1
Water basin	0	0	0	1	2	1	1	0
Dock	0	0	0	1	2	1	1	0

Table A.3 – Capacity of the CORINE land use types to deliver ecosystem services

CORINE land use	Crops	Livestock	Wood fuel	Global climate regulation	Groundwater recharge	Nutrient retention	Denitrification	Pollination
Continuous urban fabric	0	0	0	0	0	0	0	0
Discontinuous urban fabric	1	0	0	0	0	0	0	0
Green urban areas	0	0	1	1	4	2	2	3
Road and rail networks and associated land	0	0	0	0	0	0	0	1
Sport and leisure facilities	0	0	0	1	2	1	1	1
Mineral extraction sites	0	0	0	0	0	0	0	0
Dump sites	0	0	0	0	0	0	0	0
Non-irrigated arable land	5	0	0	1	2	0	2	1
Fruit trees and berry plantations	5	0	0	1	2	1	2	4
Pastures	0	3	0	2	3	1	2	0
Broad-leaved forest	0	0	2	3	3	3	3	2
Coniferous forest	0	0	4	2	2	2	2	1
Mixed forest	0	0	3	3	3	3	3	2
Moors and heathland	0	1	0	3	3	2	1	3
Natural grassland	0	1	0	3	3	2	2	4
Transitional woodland-shrub	0	0	1	3	3	3	3	3
Inland marshes	0	0	1	4	3	4	4	3
Intertidal flats	0	1	1	3	0	3	3	3
Water courses	0	0	0	1	3	1	2	0
Water bodies	0	0	0	3	3	3	2	1

Table A.4 – Capacity of the governmental land use types to deliver ecosystem services

Governmental land use	Crops	Livestock	Wood fuel	Global climate regulation	Groundwater recharge	Nutrient retention	Denitrification	Pollination
Residential and commercial fabric	1	0	2	1	1	1	1	1
Park	0	1	2	2	3	2	2	3
Business and industrial site	0	0	0	0	0	0	0	0
Infrastructure	0	0	0	0	0	0	0	0
Agricultural grassland	0	3	0	2	3	1	2	0
Recreation and sports facilities	0	0	2	1	3	2	1	1
Cropland	5	0	0	1	2	0	0	1
Cropland with an environmental target	4	0	0	1	3	0	1	1
Agricultural grassland (with an environmental or a biodiversity target)	0	2	0	2	3	2	2	4
Multifunctional forest	0	1	3	3	2	3	4	3
Forest (biodiversity management)	0	1	1	4	2	3	3	2
Heathland (biodiversity management)	0	1	0	3	3	2	1	3
High nature value grassland (biodiversity management)	0	1	0	3	3	3	3	4
High nature value grassland (no biodiversity management)	0	1	0	3	3	3	3	4
Beaches and dunes (biodiversity management)	0	1	0	0	3	1	1	1
Marshland (biodiversity management)	0	0	1	5	3	3	4	3
Mud flat and salt marshes	0	1	0	2	0	2	3	2
Water	0	0	0	2	3	2	2	1
Port area	0	0	0	1	2	1	1	0

Appendix B

Background information on the variables and processes driving the ecosystem services quantitative models in Chapter 2.

Agricultural production and livestock

Agricultural production is expressed in % yield. Areas with optimal soil conditions (soil texture, profile development and groundwater level) for agricultural practices have an estimated yield of 100 %, while areas with less suitable soil conditions suffer from certain yield losses. Information on yield losses is derived from experimental studies on the effects of desiccation and rewetting on crop-specific productivity (Moens et al. 2008 and Brouwer and Huininck 2002). How sensitive a soil is to these effects depends on soil texture, where sandy soils are more sensitive to hydrological effects than loamy soils. Furthermore, the effects of profile development and substrate on chemical and physical soil fertility are accounted for (Bollen 2012).

Wood production

Wood production is expressed in mean annual volume increment (m^3 ha^{-1} yr^{-1}). It depends on soil characteristics (De Vos 2009) and the harvest regime applied. The specific potential of some species to produce wood volumes (Moonen et al. 2011; Jansen et al. 1996) can be found in Table S.1 where differentiation is made according to soil suitability (soil texture and moisture).

Table B.1 - Overview of the relationships between soil suitability and the maximal mean growth of stemwood (m^3 ha^{-1} yr^{-1}), based on Moonen et al. (2011) and Jansen et al. (1996)

	Tree species							
Soil suitability	*Fagus sylvatica*	*Quercus* (*Robur* and *Rubra*)	*Populus sp.*	*Larix decidua*	*Pinus sylvestris*	*Pinus Nigra*	*Picea abies*	*Pseudotsuga menziesii*
4	4.0	3.0	9.0	6.0	4.0	6.0	6.0	8.0
3	6.7	5.0	11.0	8.7	6.0	9.3	9.0	10.7
2	9.3	7.0	13.0	11.3	8.0	12.7	13.0	13.3
1	12.0	9.0	15.0	14.0	10.0	16.0	16	16.0

Depending on management and ownership structure (private or public), a harvest factor is applied that estimates the proportion of the maximal mean growth that is harvested annually. The harvested volumes are available from recent data (2009-2012) on timber selling from public (state-owned) forests and from forest owner cooperatives (privately-owned but the management is state-coordinated). This database has about 80,000 records of volumes sold per tree species and circumference. For state-owned forests, the harvest factor is 0.54. Privately-owned forests and forests under nature management often have a lower (0.15) harvest factor. For private forests, there is an unknown fraction of harvest for private use and informal markets (especially for firewood).

Infiltration

Water infiltration is the movement of water from the soil surface into the soil profile ($l.m^{-2}$). Potential infiltration is determined by soil infiltration capacity (influenced mostly by soil texture) and groundwater level (Batelaan and Desmedt 2007). Data on soil texture was obtained from the Belgian national soil classification system (AGIV 2001). Soil hydrology data was generated by updating the drainage classification system from the soil map based on the most recent digital elevation model with 25m resolution (AGIV 2011). Groundwater depth has a limited effect. More important variables are interception by vegetation and runoff from paved surface to storm drain infrastructure. Both parameters were mapped at high resolution (5m cells), using detailed data on vegetation (INBO 2010), paved surface (NGI 2007) and sewerage infrastructure (Dirckx et al. 2009; Vrebos et al. 2013).

Carbon sequestration in soils

Soils in unmanaged, natural vegetation types typically have larger carbon stocks than in managed vegetation types. Soil hydrology also plays a crucial role in the creation of soil organic carbon (SOC) stocks. Soil organic carbon is especially high in forests and/or hydric soil. The potential equilibrium state for soil organic carbon stocks can be calculated using the regression formula developed by Meersmans et al. (2011), which includes as parameters soil water content, soil texture and vegetation type. De Vos B. (2009) revealed that this function systematically underestimated SOC stocks in forest soils by 32 %. This correction factor to Meersmans's regression formula is applied to all forests. Peatlands, wetlands and freshwater ecosystems can sequester higher carbon stocks than terrestrial ecosystems (Altor and Mitsch 2008).

Nutrient storage in soils

Changes in soil organic carbon also affect soil nutrient stocks. It is known that soil organic matter contains a certain percentage of nitrogen (N). Land use change and especially drainage of historically water-logged soils can increase mineralization of the SOC and result in additional supply of N to the environment. The ratio of carbon to nitrogen (C/N ratio) is typically associated with vegetation type and land use. Higher C/N ratios in the SOC can be explained by litter production that is more difficult to decompose (high C/N, high lignin concentration). Parameter values for the C/N ratio in SOC can be found in Table S.2. Estimates of the total N stocks in the soil are corrected for input through manuring, based on the data of nutrient leaching in Flemish soils (Coppens et al. 2007).

Table B.2 – SOC C/N ratio for several vegetation types

Vegetation type/ land-use	Upper and lower estimates	Central value
Cropland and production grassland	8-12	10
Floristic and species rich grasslands	10-14	12
Broad leaf forest	15-25	20
Mixed forest	20-25	22
Coniferous forest	25-30	27
Heathland	25-35	30
Phragmites wetlands	25-35	30
Wetlands (sedges and tall herbs)	15-25	20
Eutrophic alluvial forests	15-20	17
Mesotrophic wetland forest	20-25	22
Oligotrophic wetland forest	25-30	27
Peat bogs	25-35	30

Nutrient removal by denitrification

Nutrient removal by denitrification is expressed in % removal of nitrogen (N). In conditions of (temporary) waterlogging, bacterial processes enable the removal of nitrogen from ground and surface water. The most important variables are the soil moisture, supply of nitrate and soil organic carbon. As a proxy for nitrate removal efficiency, combinations of the mean highest (MHG) and mean lowest groundwater (MLG) levels were converted to an estimated nitrate removal efficiency (% of available nitrate removed), based on the results from Pinay et al. 2007.

Table B.3 - Estimated removal efficiency (%) for combinations of mean highest (MHG) and mean lowest groundwater (MLG) levels (in cm below soil surface).

MHG	MLG	>50	45	40	35	30	25	20	15	10	5-0
>50		10	13	17	20	23	27	30	33	37	40
45			20	23	27	30	33	37	40	43	47
40				30	33	37	40	43	47	50	53
35					40	43	47	50	53	57	60
30						50	53	57	60	63	67
25							60	63	67	70	73
20								70	73	77	80
15									80	83	87
10										90	93
0-5											100

Pollination

Pollination is a regulating service crucial to sustain certain types of agricultural production. In situ (cell-based) quantification of pollination is not straightforward as it depends on the presence of pollination-dependent crops near pollinator-providing habitat. As this study evaluated the suitability of a habitat to provide services, independent of the surrounding land use, a semi-quantitative method was applied. This method is based on only one input parameter (vegetation), for which experts provided a score on the capacity of the habitat to provide pollinators. This was done for the nearly 1000 classes of the biological valuation of Flanders (INBO 2010).The pollination probability by wild pollinators for a particular habitat is derived by mapping the suitability score onto a relative surface area. The surface of high quality pollinator habitat (score

4) accounts for 100 % of the surface area, while low quality pollinator habitat (e.g. score 2) would only be accounted for 50 % of the surface area.

References

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Appendix C

Appendix tables of Chapter 3

Table C.1 – Abstract from expert judged knowledge rules for agricultural production (Provincie Antwerpen, 1998).

Soil texture	Mean lowest/highest groundwater (cm)	Suitability agr. prod.
Clay	200/300	1
Clay	112/112	1
Clay	75/175	4
Clay	40/130	3
Clay	40/100	1
Clay	40/60	1
Clay	20/20	1
Sandy loam	200/300	1
Sandy loam	112/112	5
Sandy loam	75/175	5
Sandy loam	40/130	5
Sandy loam	40/100	2
Sandy loam	40/60	1
Sandy loam	20/20	1

Table C.2 - Expert judgment on levels of food production (ES_a) ranging from zero (0, white) to very high (5, dark grey) for every combination of states of agricultural production potential (P_a) and crop type (CT).

Food Production (ES_a)																	
Crop type (CT)	Agricultural Potential (P_a)																
	A1	A2	A3	A4	A5	A6	A7	A8	A9	A10	A11	A12	A13	A14	A15	A16	A17
Cropland	1	1	2	2	2	3	3	3	3	4	4	4	5	5	5	5	5
Cropland envi	1	1	1	1	1	2	2	2	2	3	3	3	4	4	4	4	4
Cropland nature	1	1	1	1	1	1	1	1	1	2	2	2	3	3	3	3	3
Grassland prod	2	2	2	2	2	2	2	2	2	3	3	3	3	3	3	3	3
Grassland prod nature	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2
Grassland nature	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
non regi Grassland nature	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2
non regi Cropland	2	2	2	2	2	2	2	2	2	3	3	3	3	3	3	3	3
None	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table C.3 – Abstract from expert judged knowledge rules for wood production (De Vos, 2000).

Soil texture	Mean lowest/highest groundwater (cm)	Soil profile	Suitability wood prod.
Clay	200/300	no profile	2
Clay	200/300	non defined	2
Clay	112/112	no profile	4
Clay	112/112	non defined	4
Clay	75/175	no profile	4
Clay	75/175	non defined	4
Clay	40/130	no profile	4
Clay	40/130	non defined	4
Clay	40/100	no profile	3
Clay	40/60	no profile	2
Clay	40/60	non defined	2
Clay	20/20	no profile	1
Clay	20/20	non defined	1
Sandy loam	200/300	stained Bt horizont	2
Sandy loam	112/112	stained Bt horizont	4
Sandy loam	75/175	stained Bt horizont	4
Sandy loam	75/175	no profile	5
Sandy loam	40/130	stained Bt horizont	4
Sandy loam	40/130	no profile	4
Sandy loam	40/100	stained Bt horizont	3
Sandy loam	40/100	no profile	3
Sandy loam	40/60	stained Bt horizont	2
Sandy loam	40/60	no profile	2
Sandy loam	20/20	stained Bt horizont	1

Table C.4 – Conditional Probability Table (CPT) for assigning levels of wood production (ES_f) based on forest production potential (P_f) and forest type (FT). This table was constructed by converting the food production table (Table C.1) to a similar, deterministic CPT and assigning a 100% probability to the selected production level for every combination of input states. This CPT was populated on expert judgment and is deterministic.

Forest type (FT)	Forest potential (P_f)	Wood production (ES_f)					
		Zero (0)	Very low(1)	Low (2)	Average (3)	High (4)	Very high (5)
Nature management	F1	0	100	0	0	0	0
Nature management	F2	0	0	100	0	0	0
Nature management	F3	0	0	0	100	0	0
Nature management	F4	0	0	0	100	0	0
Nature management	F5	0	0	0	0	100	0
Timber management	F1	0	100	0	0	0	0
Timber management	F2	0	0	100	0	0	0
Timber management	F3	0	0	0	100	0	0
Timber management	F4	0	0	0	0	100	0
Timber management	F5	0	0	0	0	0	100
None	F1	100	0	0	0	0	0
None	F2	100	0	0	0	0	0
None	F3	100	0	0	0	0	0
None	F4	100	0	0	0	0	0
None	F5	100	0	0	0	0	0

SOC regression formula (Meersmans et al., 2008): $SOC_{cropland} = (a \times H_2O_{max}) + [(b \times clay\%) + (c \times Dg) + (d \times (silt\% + sand\%) \times H_2O_{min}) + (e \times Dg^2 \times H_2O_{min}) + (f \times H_2O_{min})] + \gamma_{cropland}$

$SOC_{pasture} = (a \times H_2O_{max}) + [(g \times clay\%) + (c \times Dg) + (h \times Dg^2 \times H_2O_{min}) + (f \times H_2O_{min})] + \gamma_{pasture}$

$SOC_{forest} = (a \times H_2O_{max}) + [(g \times clay\%) + (i \times Dg) + (j \times H_2O_{min})] + \gamma_{forest}$

$SOC_{heath} = (a \times H_2O_{max}) + [(g \times clay\%) + (i \times Dg) + (j \times H_2O_{min})] + \gamma_{heath}$ where H_2O_{max}: maximum depth of the ground water table (reduction horizon); H_2O_{min}: minimum depth of the ground water table (oxidation horizon); Clay%: percentage of clay; Silt%: percentage of silt; Sand%: percentage of sand; Dg: geometric mean particle size; $a, \ldots, j$, γ_i = model parameters.

Table C.5 – Predicted value, standard error and 95% confidence interval of model parameters (Meersmans et al., 2008).

	Parameter	Value	Std. error	95% confidence interval
a	−10.120	0.292	−9.537	−10.704
b	0.074	0.011	0.097	0.052
c	−2.643	0.646	−1.351	−3.936
d	0.031	0.005	0.040	0.021
e	10.025	1.005	12.034	8.015
f	−2.925	0.394	−2.137	−3.712
g	0.168	0.011	0.189	0.146
h	8.491	1.433	11.357	5.625
i	2.334	0.558	3.451	1.217
j	−1.997	0.410	−1.176	−2.818
γa	22.471	0.566	23.604	21.339
γb	24.090	0.467	25.023	23.156
γw	25.490	0.591	26.672	24.309
γh	24.584	0.743	26.070	23.098

Appendix D

Data sources Chapter 3

Table D.1 –Data source used in the ecosystem services models

GIS-layer	***Map***	***Year***	***Source***	***Format***	***Nr. of classes***	***Classes - Ranges***
Elevation	Digital Elevation Model Flanders	2005	AGIV	raster 25m	float	8 - 58m
Hydrology	Set of GIS operations on DEM and drainage class from soil map	2012	J. Staes	raster 25m	9	a (>125cm), b (245-90cm), c (120-60cm), d (75-40cm), e (45-20cm), f (25-0cm), g (<0cm)

GIS-layer	***Map***	***Year***	***Source***	***Format***	***Nr. of classes***	***Classes - Ranges***
Land use	Land use map Flanders and Brussels	2011	L. Van Esch; Poelmans, L.; Engelen, G.; Uljee, I.	raster 100m	37	Non registered grassland with nature management, grassland with nat. man., production grassland with nat. and environmental man., production grassland, forest with nat. man., forest with forest man., non-registered cropland, cropland with nat. man., cropland with environmental man., cropland, wetland without nat. man., wetland with nat. man., heath without nat. man., heath with nat. man., coastal dune without nat. man., coastal dune with nat. man., tidal flat, open water, residential, light industry, heavy industry, water distribution, mining, energy, transport, horeca, recreation, park, military, infrastructure, administration, education and health, commercial, marine port, residential Brussels, industry Brussels
Soil texture	Digital Soil Map Flanders	2001	AGIV	vector	11	sand, loamy sand, light sandy loam, sandy loam, clay, peat, dune
Profile development	Digital Soil Map Flanders	2001	AGIV	vector	13	strongly spotted or fragmented clay enrichment horizon, weakly developed humus and/or iron enrichment horizon, well developed humus and/or iron enrichment horizon, deep antropogene humus A horizon, no profile development, non-defined profile dev.

Appendix E

Appendix table of Chapter 4

Table E.1 – Description of habitat types

Name	Description
Embryonic and shifting dunes along the shoreline	First row of dunes covered by pioneer and dynamic vegetation (often dominated by marram grass), alternated with patches of bare sand, and where active sand movement takes place ("white dunes")
Shifting dunes with marram grass	Inner dunes dominated by marram grass and where active sand movement takes place ("white dunes"), certain portion of bare sand
Dunes with moss or grass	First stage in ecological succession after marram grass, start of dune fixation ("grey dunes")
Dunes with *Hippophae rhamnoides*	Fixed dunes dominated by *Hippophae Rhamnoides* (sea buckthorn)
Dune slacks	Depressions in the dune system where the water table comes to the surface. Permanently wet soils, usually rich in species
Dunes with *Salix repens*	Dunes dominated by shrub of creeping willow. Associated with dune slacks
Wooded dunes	Final stage of ecological succession in dunes, often dominated by oaks

Appendix F

Appendix tables of Chapter 5

Table F.1 – Definition and interpretation of ecological and anthropogenic processes with their abbreviations used in Chapter 5. Column "S/L" indicates whether the process – as defined in this chapter - is restricted to the marine part (S), the terrestrial part (L) or takes place both in the marine and in the terrestrial part (S/L)

Process	S/L	Definition
Ecological processes		
Hydrodynamics (HD)	S	Movement of water by tides, currents or waves
Morphodynamics (MD)	S	Movement and dispersal of sand, fine sediment and organic matter by tides, currents and waves.
Ecological engineering (EE)	S	Processes that are important for habitat development and which take place autonomously by biota. Self-strengthening of creation of suitable conditions take place by active and passive trapping of sediment, nutrient retention and attraction of food.
Benthic production (BeP)	S	Primary and secondary production in the lowest zone of the water column (including sediment surface and sub-surface layers).
Pelagic production (PeP)	S	Primary and secondary production in the water column.

Process	S/L	Definition
Transfer (T)	S	The transfer of energy and nutrients through primary and secondary production.
Primary dune formation (DUNE)	L	The accumulation of sand on the upper beach resulting from the interaction of wind and wave action and the subsequent colonization by pioneer species and dune building species such as marram grass.
Large-scale wind dynamics (LW)	L	Wind induced transport of sand in dunes with a sufficiently high intensity that prevents the formation of a continuous vegetation cover. Large-scale wind dynamics are essential for the optimal development and maintenance of young dunes.
Small-scale wind dynamics (SW)	L	Wind induced transport of sand in dunes with a sufficiently low intensity that allows vegetation to keep up with wind induced bare sand.
Infiltration (INF)	L	Downward movement of water through the soil.
Evapotranspiration (ET)	L	The sum of evaporation and plant transpiration resulting in the loss of water from the ecosystem to the atmosphere.
Soil development (SOIL)	L	The interaction between sediment, topography, climate and biota. Soil development is the result of processes such as leaching, nutrient accumulation, decalcification and vegetation development.
Gas emissions (GHG)	S/L	Emission of methane, carbon dioxide and nitrous oxide caused by the accumulation of carbon and nitrogen in soils and subsequent decomposition in aerobic or anaerobic conditions.
Denitrification (DEN)	S/L	Conversion of nitrate to nitrogen gas by bacteria resulting in a loss of nitrogen from the ecosystem.
Vegetation development (VEG)	L	Development of and succession into a specific composition of different plant species under the influence of (micro)climate, geomorphology, topography, water cycle, soil, land use, management, etc.
Primary production (land) (PP)	L	The synthesis of organic compounds from atmospheric or aqueous carbon dioxide, principally occurring through the process of photosynthesis.
Population dynamics (POP)	S/L	Changes in the size and age composition of populations driven by biological and environmental processes (i.e. birth, growth and death, immigration and emigration).
Anthropogenic processes		
Sediment extraction (SED)	S	The extraction of sediment (mostly sand and gravel) from submarine sand banks, for use in the construction sector and in coastal defence, or to maintain (maritime) access (channel deepening).
Dumping (DUM)	S	Deposition of sediment (mostly sand and mud) on the sea floor resulting from dredging activities (mostly for the maintainenance of maritime access or for building infrastructure on the sea floor).
Bottom disturbing fishing (BeF)	S	Catch of fish living near the seafloor using techniques that disturb the sea floor, such as beams.
Pelagic fishing (PeF)	S	Catch of fish from the water column using techniques such as angles and floating nets.
Artificial reef formation (ARF)	S	Artificial deposition of hard substrate (concrete, metal, rocks, …) in the sea (below the low water line), including jetties, windmill foundations, groynes, ship wrecks, …
Sand nourishing (NOUR)	S/L	Artificial deposition of sand on the sea floor, beach or in the dunes as a measure to protect against sea level rise.

Process	S/L	Definition
Artifical infiltration (AINF)	L	Infiltration of pre-treated sewage or canal water in dunes, as a measure to avoid lowering of groundwater tables resulting from drinking water extraction.
Drainage (DRA)	L	Removal of water from the soil that is being experienced as excessive or disturbing, mainly for agricultural purposed or housing, by means of artificially dug channels, resulting in an artificial lowering of the groundwater table in the vicinity of the channel.
Water extraction (EXTR)	L	Groundwater extracted from the freshwater lens underneath dunes for the purpose of drinking water, crop production or industrial usage.
Manuring (MAN)	L	The direct addition of nutrients to the soil (mostly nitrogen and potassium compounds) to increase production of crops or other plants.
Nature management (NAT)	S/L	All human interventions that aim to stimulate vegetation development or population dynamics, including mowing, grazing, burning, clear cutting, sod cutting, exclusion of certain activities, … .
Intensive grazing (GRZ)	L	The grazing of livestock for production purposes (on average 5 livestock units per hectare, thus differentiating from the extensive grazing as nature management technique).
Cropping (CRP)	L	The usage of soils for the production of crops for human consumption, ornamental purposes, energy production or fodder.
Biological invasions (INV)	S/L	The intensive areal expansion of species introduced by humans and with adverse effects on at least one of the following aspects: biodiversity and ecosystems, agriculture and horticulture, public health or infrastructure.
Trampling (TR)	L	The physical entering of habitats by people, livestock or machines.
Disturbance (DIS)	S/L	Disturbance of animals.
Paving (PAV)	L	Making soil surface impermeable using material such as concrete, asphalt, tiles, … (buildings, roads, paths, parkings, dikes, …).

Table F.2 – Experts involved in Step 2 and 4 of the elaboration of the method on the Belgian coastal ecosystem. E = expert; FA = funding agency; PP = project partner

Last name	Name	Organisation	Expertise	Role
Lindeboom	Han	IMARES	North Sea ecosystem	E
Degraer	Steven	Royal Belgian Institute of Natural Sciences - Directorate Natural Environment	North Sea marine biology	E
Van Lancker	Vera	Royal Belgian Institute of Natural Sciences - Directorate Natural Environment	North Sea sedimentology and bathymetry	E
Pirlet	Hans	VLIZ (Flanders Marine Institute)	North Sea marine policy	E
Provoost	Sam	Flemish Research Institute Nature and Forest (INBO)	Landscape ecology coastal dunes	E
Maelfait	Hannelore	Province of West-Vlaanderen	Sustainable coastal managament	E
Van Hoestenberghe	Thomas	Fluves	Hydraulic processes	E
Van Oyen	Tomas	Flanders Hydraulics Research	Coastal morphology	E
Vanderheiden	Stijn	Spatial Planning Flanders	Spatial planning	E

Last name	Name	Organisation	Expertise	Role
Demarest	Leni	Flemish Nature and Forestry Institute	Environmental planning	E/FA
Roose	Frederik	Maritime Acces Division Flemish Government	Maritime access	E/FA
Van Wonterghem	Miran	Coastal Division of the Flemish Government	Maritime access	E/FA
Vanagt	Thomas	eCOAST	North Sea marine biology	E/PP
Meire	Patrick	Ecosystem Management Research Group, Antwerp University	Sustainable ecosystem management	E/PP
Ysebaert	Tom	Marine Research University Wageningen	Marine and estuarine benthic ecosystem	E/PP
Schellekens	Tim	eCOAST	Marine and estuarine benthic ecosystem	E/PP
Van der Biest	Katrien	Ecosystem Management Research Group, Antwerp University	Coastal ecosystem services	E/PP
D'hont	Bram	Terrestrial Ecology Unit, University Ghent	Coastal ecology	E/PP
Bonte	Dries	Terrestrial Ecology Unit, University Ghent	Coastal dune spatial ecology	E/PP
Cosyns	Eric	Intercommunal West-Vlaanderen	Nature and forest ecology	E/PP
Lemey	Emile	eCOAST	Marine biology	E/PP

Table F.3 - Stakeholders involved in Step 3 of the elaboration of the method on the Belgian coastal ecosystem

Last name	Name	Organisation	Sector
Hansen	Krien	Natuurpunt	Environment
schroé	pual	Haven Zeebrugge	Port
Desloovere	*Steven*	*Jachthaven Nieuwpoort*	Tourism
Overmeire	Ann	Flanders' Maritime Cluster	Marine and Maritime Industry
Sterckx	Tomas	DEME	Dredging
Malherbe	Bernard	Jan de Nul	Dredging
Barbery	Stefaan	Province of West-Vlaanderen	Government (province)
Vanderheiden	Stijn	Ruimte Vlaanderen	Government (region)
Degloire	Mieke	FOD Leefmilieu	Environment
Moulaert	Ine	DEME	Dredging
Versluys	Willy	Shipping company Versluys	Fisheries
Devalck	Steven	Toerisme Vlaanderen	Tourism

Appendix G

Appendix tables of Chapter 6

Table G.1 – Effects of changing parameters on ecological processes related to the development of an artificial island resulting from the degradation of the existing habitat of lower beach and from the newly created habitat tidal flats and marshes. Dark blue: marine changing parameters, brown: terrestrial changing parameters, light blue: changing parameters at sea and on land.

		Ecological processes																
Changing parameters	**Expec-ted change**	Hydrodynamics	Morphodynamics	Ecological engineering	Benthic production	Pelagic production	Transfer	Primary dune formation	Large-scale wind dynamics	Small-scale wind dynamics	Infiltration	Evapotranspiration	Soil development	GHG	Denitrification	Vegetation development	Primary production (land)	Population dynamics
Hydrodynamics	**-2**	-2	-2	+1	+1	+1	-1	+2	0	0	0	0	0	0	0	0	0	0
Morphodynamics	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Organic mirco pollution	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Residence time	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Exposition time	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Upwelling	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Flood frequency (existing tidal flats & marshes)	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tidal amplitude	**-1**	-1	-1	0	0	0	0	-1	0	0	0	0	0	0	0	-1	0	0
Inflow pollutants	**+1**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	-1
Inflow nutrients	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
pH sea water	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Turbidity	**+1**	0	0	-1	-2	0	0	0	0	0	0	0	0	0	0	0	0	0
Fisheries	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sediment extraction	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sand suppletion - new habitat	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Climate - Storm surge and frequency	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Climate - Sea level	**+1**	0	0	0	0	0	0	-1	0	0	0	0	0	0	0	-1	0	0
Surface area gravel beds	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area tidal flats	**+2**	0	0	0	+2	+1	+2	0	0	0	0	0	+2	+2	+2	+2	+2	+2
Surface area reefs	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area submerged sandbanks & foreshore	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area estuary	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area lower beach & emerged sand banks	**-2**	0	0	0	-2	0	-2	-1	0	0	0	0	0	0	0	0	0	-2
Surface area upper beach & dune foot	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Habitat fragmentation - existing habitat	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Habitat fragmentation - new habitat	**-2**	0	0	+1	+1	+1	+1	0	+2	+2	0	0	0	0	0	0	0	+2

Changing parameters	Expec-ted change	Ecological processes																
		Hydrodynamics	Morphodynamics	Ecological engineering	Benthic production	Pelagic production	Transfer	Primary dune formation	Large-scale wind dynamics	Small-scale wind dynamics	Infiltration	Evapotranspiration	Soil development	GHG	Denitrification	Vegetation development	Primary production (land)	Population dynamics
Nature management - existing habitat	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nature management - new habitat	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Noise and visual disturbance	**+1**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Soil disturbance (no habitat loss)	**+1**	0	+1	-1	-1	0	0	0	0	0	0	0	0	0	0	0	0	+1
Atmospheric nitrogen deposition	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Groundwater extraction	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Drainage	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Artifical infiltration	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Climate - Evapotranspiration	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Soil acidification	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Manuring	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Disturbance by access (no habitat loss)	**+1**	0	0	0	0	0	0	-1	0	0	0	0	0	0	0	0	0	+1
Accesibility urban areas	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface hardening	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area dynamic dunes	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area herbaceous dunes	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area dunes with shrub	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area wet dunes	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sum processes (existing habitat)		**-3**	**-2**	**-1**	**-4**	**+1**	**-3**	**-2**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**-2**	**0**	**-1**
Sum processes (new habitat)		**0**	**0**	**+1**	**+3**	**+2**	**+3**	**0**	**+2**	**+2**	**0**	**0**	**+2**	**+2**	**+2**	**+2**	**+2**	**+4**

Table G.2 – Effects of changing parameters on anthropogenic processes related to the development of an artificial island resulting from the degradation of the existing habitat of lower beach and from the newly created habitat tidal flats and marshes. Dark blue: marine changing parameters, brown: terrestrial changing parameters, light blue: changing parameters at sea and on land.

		Anthropogenic processes																
Chaning parameters	**Expec-ted change**	Sediment extraction	Dumping	Bottom disturbing fishing	Pelagic fishing	Artificial reef formation	Sand nourishing	Artifical infiltration	Drainage	Water extraction	Manuring	Nature management	Intensive grazing	Cropping	Biological invasions	Disturbance by access	Noise and visual disturbance	Surface hardening
Hydrodynamics	**-2**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Morphodynamics	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Organic mirco pollution	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Residence time	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Exposition time	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Upwelling	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Flood frequency (existing tidal flats & marshes)	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tidal amplitude	**-1**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Inflow pollutants	**+1**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Inflow nutrients	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
pH sea water	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Turbidity	**+1**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fisheries	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sediment extraction	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sand suppletion - new habitat	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Climate - Storm surge and frequency	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Climate - Sea level	**+1**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area gravel beds	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area tidal flats	**+2**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area reefs	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area submerged sandbanks & foreshore	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area estuary	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area lower beach & emerged sand banks	**-2**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area upper beach & dune foot	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Habitat fragmentation - existing habitat	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Habitat fragmentation - new habitat	**-2**	0	0	+1	0	0	0	0	0	0	0	+1	0	0	-1	0	0	0
Nature management - existing habitat	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nature management - new habitat	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Noise and visual disturbance	**+1**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	+1	0

		Anthropogenic processes																
Chaning parameters	**Expec-ted change**	Sediment extraction	Dumping	Bottom disturbing fishing	Pelagic fishing	Artificial reef formation	Sand nourishing	Artifical infiltration	Drainage	Water extraction	Manuring	Nature management	Intensive grazing	Cropping	Biological invasions	Disturbance by access	Noise and visual disturbance	Surface hardening
Soil disturbance (no habitat loss)	**+1**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	-1	0
Atmospheric nitrogen deposition	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Groundwater extraction	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Drainage	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Artifical infiltration	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Climate - Evapotranspiration	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Soil acidification	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Manuring	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Disturbance by access (no habitat loss)	**+1**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	+1	+1	0
Accesibility urban areas	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface hardening	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area dynamic dunes	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area herbaceous dunes	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area dunes with shrub	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Surface area wet dunes	**o**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sum processes (existing habitat)		**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**+1**	**+1**	**0**
Sum processes (new habitat)		**0**	**0**	**+1**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**+1**	**0**	**0**	**-1**	**0**	**0**	**0**

Table G.3 – Effects of processes on habitats and ecosystem services in the Belgian coastal zone. Dark blue: marine processes, brown: terrestrial processes, light blue: processes taking place at sea and on land (based on Table 5.3)

				HABITATS												ECOSYSTEM SERVICES									
		Sum processes (existing habitat)	**Sum processes (new habitat)**	Pelagic	Gravel beds	Tidal flats and marshes	(Artifical) reefs	Submerged sandbanks and foreshore	Estuary	Lower beach and emerged sand banks	Upper beach and dune foot	White dunes	Grey dunes - herbaceous	Grey dunes - shrub	Dune slacks	Agricultural production	Fisheries production	Aquaculture production	Sediment extraction	Drinking water provisioning	Flood protection	Climate regulation	Water quality regulation	Wind energy	Recreation and tourism
ECOLOGICAL PROCESSES	Hydrodynamics (HD)	**-3**	**0**	+2	+1	+1	+1	+1	+2	+1	-1	-1	0	0	0	0	+2	+2	0	0	-2	0	+1	+2	0
	Morphodynamics (MD)	**-2**	**0**	0	-1	+2	+1	+2	-1/+1	+2	0	0	0	0	0	0	-1/+1	-0,5	+1	0	+1	0	0	0	0
	Ecological engineering (EE)	**-1**	**+1**	+1	+2	0	+2	+2	+1	0	0	0	0	0	0	0	+2	+1	-0,5	0	+1	+1	+2	0	0
	Benthic production (BeP)	**-4**	**+3**	+1	+2	+2	+2	+2	+2	+2	0	0	0	0	0	0	+2	0	0	0	0	+1	+1	0	+1
	Pelagic production (PeP)	**+1**	**+2**	+2	+2	+1	+1	+1	+2	+1	0	0	0	0	0	0	+2	+2	0	0	0	+1	+1	0	-1/+1
	Transfer (T)	**-3**	**+3**	+2	+2	+2	+2	+2	+2	+2	0	0	0	0	0	0	+2	+2	0	0	0	+2	+2	0	+1
	Primary dune formation (DUNE)	**-2**	**0**	0	0	0	0	0	0	0	+2	+0,5	0	0	+1	0	0	0	0	+1	+2	0	0	0	+2
	Large-scale wind dynamics (LW)	**0**	**+2**	0	0	-0,5	0	0	0	0	0	+2	+1	-1	+1	-1	0	0	0	-1	+2	0	+1	0	+1
	Small-scale wind dynamics (SW)	**0**	**+2**	0	0	0	0	0	0	0	0	+1	+2	0	+1	0	0	0	0	0	+1	0	+1	0	0
	Infiltration (IF)	**0**	**0**	0	0	0	0	0	0	0	0	0	0	+2	+2	+2	0	0	0	+2	0	0	+1	0	0
	Evapotranspiration (ET)	**0**	**0**	0	0	0	0	0	0	0	0	+0,5	0	0	-2	-1	0	0	0	-2	0	-1	0	0	0
	Soil development (SOIL)	**0**	**+2**	0	0	0	0	0	0	0	0	-1	+1	+2	+1	+1	0	0	0	-1/+1	-1	+0,5	+1	0	0
	Vegetation development (VEG)	**0**	**+2**	0	0	+2	0	0	0	0	+2	+2	+2	+2	+2	0	0	0	0	0	+2	+1	0	0	+1
	Primary production (land) (PP)	**0**	**+2**	0	0	+1	0	0	0	0	+1	+1	+1	+2	+2	0	0	0	0	0	0	+2	+2	0	0

		Sum processes (existing habitat)	Sum processes (new habitat)	HABITATS												ECOSYSTEM SERVICES									
				Pelagic	Gravel beds	Tidal flats and marshes	(Artifical) reefs	Submerged sandbanks and foreshore	Estuary	Lower beach and emerged sand banks	Upper beach and dune foot	White dunes	Grey dunes - herbaceous	Grey dunes - shrub	Dune slacks	Agricultural production	Fisheries production	Aquaculture production	Sediment extraction	Drinking water provisioning	Flood protection	Climate regulation	Water quality regulation	Wind energy	Recreation and tourism
	Gas emissions (GHG)	-2	+2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	-2	+1	0	0
	Denitrification (DEN)	0	+2	+1	0	+1	0	+1	+1	+1	+1	0	+1	+1	+1	-2	0	0	0	0	0	-1	2	0	0
	Population dynamics (POP)	-1	+4	+2	+2	+2	+2	+2	+2	+2	+2	+1	+2	+1	+2	-1/+1	+2	-0,5	0	0	+0,5	0	0	0	+1
ANTHROPOGENIC PROCESSES	Sediment extraction (SED)	0	0	-1	-2	0	-2	-1	0	0	0	0	0	0	0	0	-1	0	+2	0	0	0	0	-1	0
	Sediment dumping (DUM)	0	0	-1	-2	-1	-2	-0,5	-1	0	0	0	0	0	0	0	-1	-1	0	0	0	0	0	0	0
	Bottom disturbing fishing (BeF)	0	+1	0	-1	0	-2	-2	0	0	0	0	0	0	0	0	+1	0	0	0	0	0	-1	0	+1
	Pelagic fishing (PeF)	0	0	-1	0	0	0	0	0	0	0	0	0	0	0	0	+1	0	0	0	0	0	0	0	+1
	Artificial reef formation (ARF)	0	0	-1/+1	0	0	+2	-1	-1	0	0	0	0	0	0	0	+1	+2	-1	0	+1	0	+2	0	+1
	Artifical infiltration (AIF)	0	0	0	0	0	0	0	0	0	0	0	0	0	+1	+1	0	0	0	+2	0	0	+1	0	+0,5
	Drainage (DRA)	0	0	0	0	0	0	0	0	0	0	0	-0,5	-0,5	-2	+2	0	0	0	-1	0	-2	-2	0	+0,5
	Water extraction (EXTR)	0	0	0	0	0	0	0	0	0	0	0	-1	-1	-2	-1	0	0	0	+1	+0,5	-2	-2	0	0
	Manuring (MAN)	0	0	-1	0	-1	0	0	-2	-1	+1	-2	-2	-1	-2	+2	0	0	0	0	-1	-1	-2	0	0
	Grazing (GRZ)	0	0	0	0	-1	0	0	0	0	0	0	+1	-1	+1	+2	0	0	0	-2	0	-2	-1	0	0
	Cropping (CRP)	0	+1	0	0	+1	0	0	0	0	0	0	-2	-2	-2	+2	0	0	0	-2	0	-1	0	0	0
	Disturbance by access (TR)	0	0	0	0	0	0	0	0	0	-2	+1	0	-1	-1	0	0	0	0	0	-1	0	0	0	+2
	Surface hardening (PAV)	0	0	0	0	0	0	0	0	-2	-2	-2	-2	-2	-2	-2	0	0	0	-2	-2	-2	-2	0	-1/+1

	Sum processes (existing habitat)	Sum processes (new habitat)	HABITATS												ECOSYSTEM SERVICES									
			Pelagic	Gravel beds	Tidal flats and marshes	(Artifical) reefs	Submerged sandbanks and foreshore	Estuary	Lower beach and emerged sand banks	Upper beach and dune foot	White dunes	Grey dunes - herbaceous	Grey dunes - shrub	Dune slacks	Agricultural production	Fisheries production	Aquaculture production	Sediment extraction	Drinking water provisioning	Flood protection	Climate regulation	Water quality regulation	Wind energy	Recreation and tourism
Sand nourishing (NOUR)	**0**	**-1**	-1	-2	0	-2	+1	-1	-1	+1	+1	0	0	0	0	-1	0	+1	+1	+2	0	0	0	+2
Nature management (NAT)	**+1**	**0**	+1	+2	+2	+1	+1	+1	+1	+2	+1	+2	+1	+2	0	+2	0	0	+1	+1	-1	+1	0	+2
Biological invasions (INV)	**+1**	**0**	-2	-2	-2	-2	-2	-2	0	0	0	-1	-2	-1	-2	-2	-2	0	-1	-1	+0,5	+0,5	0	-1/+1
Noise and visual disturbance (DIS)	**0**	**0**	-2	-2	-2	-2	-2	-2	-2	-2	-2	-2	-2	-2	0	0	0	0	0	0	0	0	0	-1/+1
IMPACTS EXISTING HABITAT			**+2**	**0**	**0**	**+1**	**+1**	**0**	**-21**	**-4**	**-1**	**0**	**0**	**0**	**0**	**-14**	**-2,5**	**0**	**0**	**-11,5**	**-11**	**-9**	**0**	**-12**
IMPACTS NEW HABITAT			**+7**	**-1**	**+32**	**-2**	**-2**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**+29**	**+10**	**0**	**0**	**+8**	**+10,5**	**+22,5**	**0**	**+15**

Curriculum Vitae

Education and research

2010 – 2018	Doctor in Sciences: Biology Universiteit Antwerpen
2007 – 2009	Researcher Flanders Hydraulics Research, Antwerpen
2004 – 2006	Master in Oceanography (Master of Advanced Studies) Université de Liège Degree : *Greatest distinction*
1998 – 2002	Master in Geography, option Physical Geography (Licentiate) Universiteit Gent Degree: *Distinction*

Journal papers A1

2017

Van der Biest K., De Nocker L., Provoost S., Boerema A., Staes J. and Meire P. (2017). Dune dynamics safeguard ecosystem services. Ocean and Coastal Management 149, 148-158

Staes J., Broekx S., Van der Biest K., Vrebos D., Beauchard O., De Nocker L., Liekens I., Poelmans L. Verheyen K. and Meire P. (2017). Quantification of the potential impact of nature conservation on ecosystem services supply in the Flemish region : a cascade modelling approach. Ecosystem services 24, 124-137

2015

Van der Biest K., Vrebos D., Staes J., Boerema A., Bodí M.B., Fransen E. and Meire P. (2015). Evaluation of the accuracy of land-use based ecosystem service assessments for different thematic resolutions. Journal of Environmental Management 156, 41-51

Landuyt D., Van der Biest K., Broekx S., Staes J., Meire P. en Goethals P. (2015). A GIS plug-in for Bayesian belief networks: Towards a transparent software framework to assess and visualise uncertainties in ecosystem service mapping. Environmental Modelling & Software 71, 30-38

Vrebos D., Staes J., Struyf E., Van der Biest K. and Meire P. (2015). Water displacement by sewer infrastructure and its effect on the water quality in rivers. Ecological Indicators 48, 22-30

Keune H., Dendoncker N., Florin P., Jacobs S., Kampelmann S., Boeraeve F., Dufrêne M., Bauler T., Casaer J., Cerulus T., De Blust G., Denayre B., Janssens L., Liekens I., Panis J., Scheppers T., Simoens I., Staes J., Turkelboom F., Ulenaers P., Van der Biest K. and Verboven J. (2015). Emerging ecosystem services governance issues in the Belgium ecosystem services community of practice. Ecosystem services 16, 212-219

2014

Van der Biest K., D'Hondt R., Jacobs S., Landuyt D., Staes J., Goethals P. and Meire P. (2014). EBI: An index for delivery of ecosystem service bundles. Ecological Indicators Volume 37, Part A, February 2014, Pages 252–265

2009

Verwaest T., Van der Biest K., Vanpoucke P., Reyns J., Vanderkimpen P., De Vos L., De Rouck J. and Mertens T. (2009). Coastal flooding risk calculations for the Belgian coast. 31st International onference on Coastal Engineering. Coastal Engineering VOLS 1-5 Pages: 4193-4201

Journal papers A3

2018

van Bodegom P., van Oudenhoven A., Van der Biest K., Pijpers B., van 't Zelfde M. and Besteman B. (2018). Sleutelfactor Context. Ecosysteemdiensten van watersystemen inzichtelijk gemaakt. Landschap 35(1), 43-49

2017

Boerema A., Van der Biest K. and Meire P. (2017). Towards sustainable port development. Terra et Aqua 149, 5-17

2016

Boerema A., Van der Biest K. and Meire P. (2016). Maritime meets ecosystem services. The monetary valuation of the Kruibeke polder. Terra et Aqua 141, 5-14

2015

Van der Biest K., De Blust G., Staes J. and Meire P. (2015). Inrichten van het landduinenlandschap met ecosysteemdiensten. ANTenne 2, 17-23

Book chapters

2014

Jacobs S., Spanhove T., Thoonen M., De Smet L., Boerema A., Van der Biest K. and Landuyt D. (2014). Hoofdstuk 9 – Interacties tussen aanbod, gebruik en vraag van ecosysteemdiensten in Vlaanderen. In Stevens M. et al. (Eds.), Natuurrapport – Toestand en trend van ecosystemen en ecosysteemdiensten in Vlaanderen. Instituut voor Natuur- en Bosonderzoek, Brussel, 1-61

Van der Biest K., Van Gossum P., Struyf E. en Van Daele T. (2014). Hoofdstuk 21 - Ecosysteemdienst regulatie van erosierisico. In Stevens M. et al. (Eds.), Natuurrapport – Toestand en trend van ecosystemen en ecosysteemdiensten in Vlaanderen. Instituut voor Natuur- en Bosonderzoek, Brussel, 1-50

2013

Liekens I., Broekx S., Smeets N., Staes J., Van der Biest K., Schaafsma M., De Nocker L., Meire P. and Cerulus T. (2013). Chapter 2.8 – The ecosystem services valuation tool and its future developments. In Jacobs S. et al. (Eds.), Ecosystem Services: Global Issues, local practices, San Diego

Van der Biest K., D'hondt R., Jacobs S., Landuyt D., Staes J., Goethals P. and Meire P. (2013). EBI: an index for delivery of ecosystem service bundles. In Jacobs S. et al. (Eds.), Ecosystem Services: Global Issues, local practices, San Diego, 263-272

Turkelboom F., Raquez P., Dufrêne M., Raes L., Simoens I., Jacobs S., Stevens M., De Vreese R., Panis J., Hermy M., Thoonen M., Liekens I., Fontaine C., Dendoncker N., Van der Biest K., Casaer J., Heyrman H., MEiresonne H. and Keune H. (2013). CICES Going Local: Ecosystem Services Classification Adapted for a Highly Populated Country. In Jacobs S. et al. (Eds.), Ecosystem Services: Global Issues, local practices, San Diego, 263-272

Other research output

Sleutelfactor Context Analyse Instrument. ArcGIS 10.2 tool to assess the effects of measures in freshwater ecosystems on ecosystem services. Available through www.deltares.nl

ECOPLAN-SE toolbox. Q-GIS plugin for spatially explicit quantification of ecosystem services in Flanders. Version 1.0, March 2017. Available upon request (dirk.vrebos@uantwerpen.be)

ECOPLAN map database. Online map tool to analyse ecosystem services in Flanders. www.ecosysteemdiensten.be

ESDKANSEN. Online map tool to analyse potentials for climate adaptation and ecosystem services in Flanders. https://esdkansen.marvin.vito.be/

Nature Value Explorer. Online tool for qualitative, quantitative and monetary assessment of ecosystem services in Flanders (Natuurwaardeverkenner). www.natuurwaardeverkenner.be

Coastal ecosystem impact assessment tool. Excel tool to assess effects of changes on habitats and ecosystem services in the Belgian coastal zone. Available upon request (frederik.roose@mow.vlaanderen.be)

Oral presentations at national and international conferences (as speaker)

2018

The ecosystem goods and services of coastal dunes and their benefits for people and the economy. Keynote at International LIFE+ Flandre Workshop, June 12-14, Dunkirk, France

Ecosystem vision for the Belgian coastal zone. International LIFE+ Flandre Workshop, June 12-14, Dunkirk, France

2016

Building a coastal ecosystem vision using ecosystem services. European ESP Conference, September 19-23, Antwerp, Belgium

Building a coastal ecosystem vision using ecosystem services. 56th International ECSA Conference, September 4-7, Bremen, Germany

2015

Developing an ecosystem services based vision for the Belgian coastal zone. 8th International ESP Conference, November 9-13, Stellenbosch, South-Africa

Linking GIS with Bayesian networks for ecosystem service assessment and vision building. 8th International ESP Conference, November 9-13, Stellenbosch, South-Africa

Dynamic dunes safeguard ecosystem services. International Dunes and Estuaries Conference, September 16-18, Brugge, Belgium

Developing an ecosystem services based vision for the Belgian coastal zone. International IALE-D Coastal Ecosystem Services Workshop, March 22-25, Kiel, Germany

2014

Ecosysteemdiensten als inspiratie voor de landduinenregio. Studie- en netwerkmoment Ecosysteemdiensten, November 18th, Laakdal, Belgium

Ecosystem service delivery versus plant functional diversity. 7th International ESP Conference, September 8-12, San José, Costa Rica

Ecosystem services: practical usability in validating N2000 sites. International workshop NATURA2000 People, April 2, Brugge, Belgium

2013

Ecosysteemdiensten als ondersteuning van ecologische maatregelen. Keynote at 10th Waterforum, September 23, Brussels, Belgium

An integrated model to assess the effects of land use change on the delivery of multiple ecosystem services. International ESP – IALE-D joint workshop, May 6-8, Kiel, Germany

2011

Scale effects of ecosystem services within catchments. BEES workshop II, March 23, Leuven, Belgium

Wat biedt de Nete aan ecosysteemdiensten en kunnen we die beheren? Keynote at 14th ANKONA-day, Antwerpse Koepel voor Natuurstudie, February 12, Antwerp, Belgium

Research projects [1]

2018 – 2022	PROWATER – Protecting and restoring raw water sources through actions at the landscape scale. Funded by Interreg2Seas (EU).
2017 – 2019	SMARTSEDIMENT – Development of an ecosystem services toolkit for the optimization of sediment management. Funded by Interreg Vlaanderen-Nederland (EU) and Maritime Access (Flemish government).
2014 – 2018	FLANDRE – Evaluation of the socio-economic impact of the FLANDRE-project on the local economy, society and ecosystem services. Funded by ANB (Flemish government), Conservatoire du Littoral (FR) and Département du Nord (FR).
2017	Biodiversity, climate change adaptation, nature education and ecosystem services in Santuario Nacional de Ampay (Peru). Interchange project funded by Broederlijk Delen, in cooperation with Regionaal Landschap Zuid-Hageland.
2016 – 2017	Maarkebeek – Applying ecosystem services in developing a climate adaptation strategy for the Maarkebeek valley. Funded by Provincie Oost-Vlaanderen.
2016 – 2017	Ecological Key Factors – Key factor ‘Context – Ecosystem services’. Funded by STOWA (NL).

[1] The list does not include research from before 2010.

2012 – 2017	SOGLO - The soil system under global change. Funded by BELSPO, IAP program.
2015	Ecosystem services: towards integrated maritime infrastructure project assessments. Funded by IADC.
2014 – 2016	Ecosystem vision for the Flemish Coast. Funded by Maritime Access and ANB (Flemish Government).
2013 – 2016	ECOPLAN – Planning for ecosystem services. Funded by IWT.
2013 – 2014	Ecosystem Services of the Campine Land Dune Area. Subsidised by Province of Antwerp.
2012 – 2014	ESSENSE – Mapping regulating services using remote sensing. Funded by BELSPO, STEREOII program.
2012 – 2013	Valuation of ecosystem services: the nature value explorer 2.0 (Natuurwaardeverkenner). Funded by LNE (Flemish government).
2012 – 2013	Estimate of the benefits delivered by the Flemish NATURA2000 network. Funded by ANB (Flemish government).
2009 – 2012	ECOFRESH – Ecosystem Services of freshwater ecosystems. Funded by BELSPO, Science for a Sustainable Development program.

Lectures and master thesis supervision

2015 – 2016

Guest lecture “Services écosystémiques” (2015-2016). Integraal Waterbeheer – Protos, 23 Octobre 2015, Antwerp.

2014 – 2015

Guest lecture “Ecosysteemdiensten” (2014 – 2015). Industrieel Ingenieur in de Biowetenschappen, Natuur en Milieu, Thomas More Hogeschool Kempen, Geel.

2013 – 2014

Jacobs S. (2013 - 2014). Master thesis. Microbial activity as a proxy for soil-plant interactions. Interactions between microbes, plants and ecosystem services. Master Biologie, Universiteit Antwerpen.

2012 – 2013

Van Geel T. (2012 - 2013). Master thesis. Ecosysteemdiensten van de Kempense Landduinen. Focus houtproductie. Master in Milieuwetenschappen, Universiteit Antwerpen.

Coen A. (2012 - 2013). Schakelscriptie. Linking Remote Sensing with Plant Functional Traits (PFT) and Ecosystem Services (ES). Schakelprogramma Master in Milieuwetenschappen, Universiteit Antwerpen.

De Bondt C. (2012 - 2013). Master thesis. Ecosysteemdiensten in de landduinenregio Middenkempen. Focus recreatie. Master in Milieuwetenschappen, Universiteit Antwerpen.

Van den Brande P. (2012 - 2013). Master thesis. Ecosysteemdiensten in de landduinenregio Middenkempen. Het verband tussen biodiversiteit en het ecosysteemdienstenconcept. Master in Milieuwetenschappen, Universiteit Antwerpen.

Vantilt G. (2012 - 2013). Master thesis. Ecosysteemdiensten in de landduinenregio Middenkempen. Focus landbouw en voedselproductie. Master in Milieuwetenschappen, Universiteit Antwerpen.

Course “Landscape ecology” (2012-2013). Master Biologie, Bachelor Bio-ingenieurswetenschappen, Universiteit Antwerpen.

2010 – 2011

Spoelders P. (2010 - 2011). Master thesis. Ecosysteemdiensten: van mondiaal tot lokaal. Analyse van historische, huidige en potentiële ecosysteemdiensten van het Malesbroek. Master in Milieuwetenschappen, Universiteit Antwerpen.

Vanhastel E. (2010 - 2011). Master thesis. Ecosysteemdiensten: van mondiaal tot lokaal. Ecologisch functioneren, successie en beheer van moerasecosystemen. Master in Milieuwetenschappen, Universiteit Antwerpen.