Norwegian University of Life Sciences
Faculty of Environmental Sciences
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Philosophiae Doctor (PhD)
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Preface

This thesis is submitted in fulfilment of the requirements for the degree of Philosophiae Doctor (PhD) at the Faculty of Environmental Sciences and Natural Resource Management of the Norwegian University of Life Sciences. The research presented in this thesis is part of BIOWATER, a Nordic Center of Excellence funded by Nordforsk under project number 82263.

The thesis consists of three papers, preceded by a synopsis that synthesizes the work into a whole, summarising the problem statement, the current state of knowledge, the aims and relation between the papers, the applied methods and main findings, and a discussion which covers the main conclusions, the contribution to the field and policy implications and an outlook for the future.

Acknowledgements

I could not have written this thesis without the guidance, knowledge, assistance, resources and support of all the people involved in the work, and of those close to me. At the risk of forgetting some, I will attempt here to thank everyone that made its completion possible.

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Finally, I want to thank Sophie, for giving me a loving home, for having complete confidence in me every day, for sharing the journey with me and making it so much better.

Oslo, June 2021

Bart Immerzeel

List of papers

Paper I

Immerzeel, B., Vermaat, J.E., Juutinen, A., Pouta, E. and Artell, J.

Appreciation of Nordic landscapes and how the bioeconomy might change that: results from a discrete choice experiment.

Revised manuscript under review at Land Use Policy.

Paper II

Immerzeel, B., Vermaat, J.E., Riise, G., Juutinen, A. and Futter, M.

Estimating societal benefits from Nordic catchments: An integrative approach using a final ecosystem services framework.

Published in PLOS ONE.

Paper III

Immerzeel, B., Vermaat, J.E., Collentine, D., Juutinen, A., Kronvang, B., Skarbøvik, E. and Vodder Carstensen, M.

The value of change: a scenario assessment of the effects of bioeconomy driven land use change on ecosystem service provision.

Manuscript ready for submission.

Abbreviations

CICES	Common International Classification of Ecosystem Services
CORINE	COoRdination of INformation on the Environment
DCE	Discrete Choice Experiment
EU	European Union
FES	Final Ecosystem Service
GIS	Geographic Information System
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
MA	Millennium Assessment
MAES	Mapping and Assessment of Ecosystems and their Services
MXL	Mixed Logit
NBP	Nordic Bioeconomy Pathway
NEA	National Ecosystem Assessment
NEP	New Ecological Paradigm
NESCS	National Ecosystem Services Classification System
OECD	Organisation for Economic Co-operation and Development
SEEA	System of Integrated Environmental and Economic Accounting
TEEB	The Economics of Ecosystems and Biodiversity
TEV	Total Economic Value
UN	United Nations
WFD	Water Framework Directive

Summary

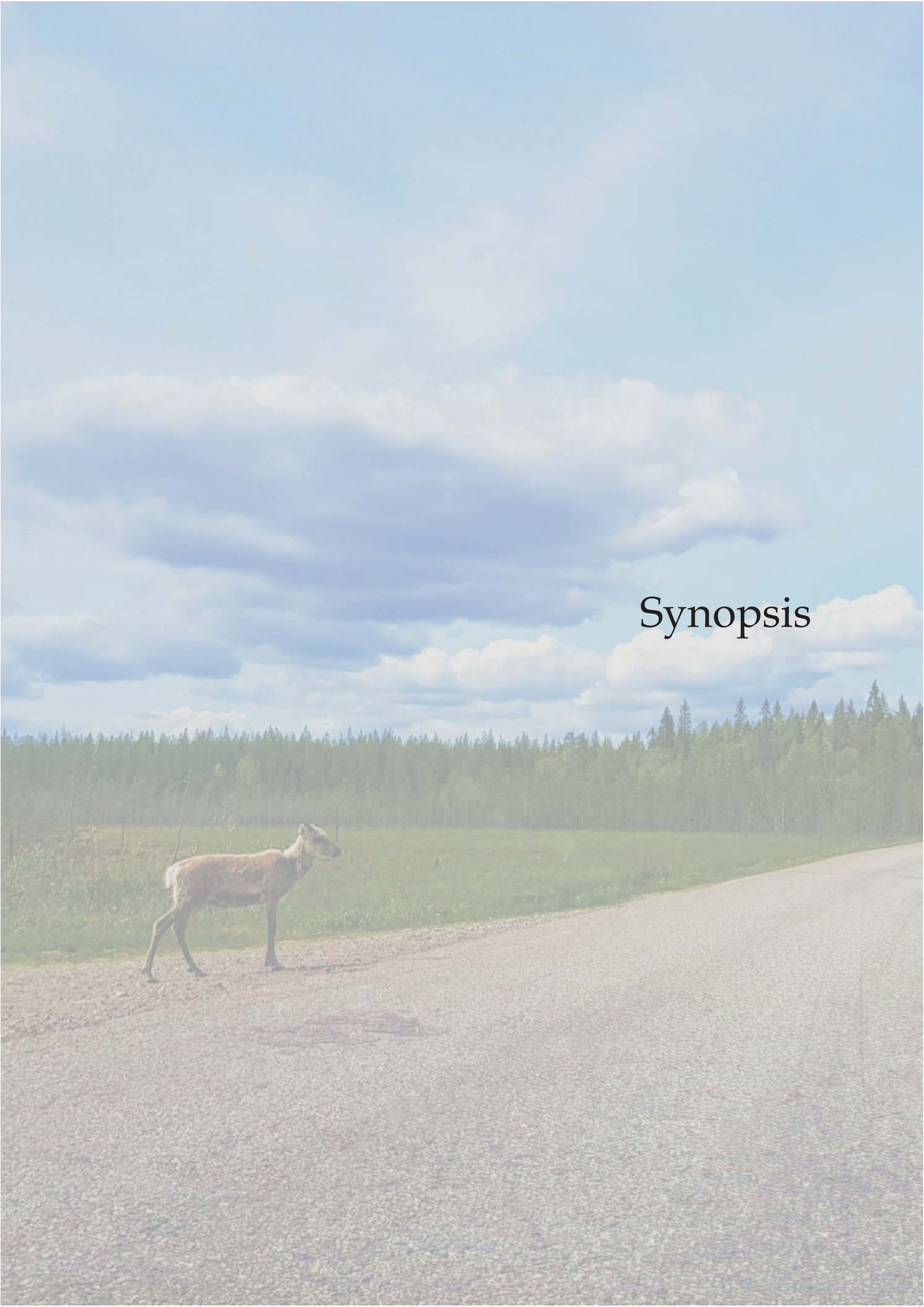
In light of the increasing pressures from human activities on ecosystems and the global climate, the Nordic countries have decided that a green shift is necessary to ensure the future wellbeing of society. The transition to a bioeconomy is defined by a shift from fossil-based goods and energy to renewable, bio-based ones. This implies that resource extraction from ecosystems, which generate the biological resources for a bioeconomy, needs to increase. At the same time, we benefit from ecosystems in a wide variety of ways, often quantified as ecosystem services, ranging from the capacity to produce food to the regulation of water quality and possibilities for recreation. How a green shift would impact the value of ecosystem services generated in Nordic catchments is unknown, and this thesis aims to address this knowledge gap, based on three papers. The study subjects were six Nordic catchments, in Denmark, Finland, Norway and Sweden. The first paper presents a study on the relationship between landscape attributes and preference for recreation, using a discrete choice experiment. The results showed that, on average, respondents in the catchments prefer a more balanced mix between agriculture and forestry, neither more intensive nor extensive land management, an increase in water clarity, nature reserve areas and local employment from agriculture, forestry and fishing, and a decrease in flood frequency. However, the results varied among catchments as well as among different types of respondents. The second paper presents an estimation of the current total societal value of ecosystem services generated in the six catchments and an analysis of its variability. Average total value estimates ranged from roughly €400 ha^{-1} $year^{-1}$ in the Finnish Simojoki catchment, to €7,000 ha^{-1} $year^{-1}$ in the Norwegian Orrevassdraget catchment. Most of the value was generated by active nature appreciation, such as recreation, but there was large spatial variability among and within catchments. Other major ecosystem services were the supporting environment for agriculture, forestry and carbon sequestration. Soil type, slope, landscape diversity, population density and access to water all showed significant correlations to ecosystem services values. The third paper presents an analysis of the effects of transitioning to a bioeconomy on the value of ecosystem services. It applied five bioeconomy scenarios to the framework developed in Paper II, and for each assessed its effects on land use change, socio-geographic change and subsequently on the ecosystem services generated in each catchment. It found that a developed bioeconomy is likely to increase the value of ecosystem services as a whole, with the sustainability-focused scenario and the scenario aimed at maximising economic output generating most benefits. However, the effects vary among catchments, as well as among stakeholder groups benefiting from ecosystem services. This suggests that bioeconomy policy will not only affect total societal value, but also the distribution of value within society.

Sammendrag

I lys av det økende presset fra menneskelig aktivitet på det globale klimaet og klodens økosystemer, har de nordiske landene blitt enige om nødvendigheten av et grønt skifte for å sikte fremtidens samfunn. Overgangen til en bærekraftig bioøkonomi defineres av et skifte fra produksjon av varer og energi basert på fossile ressurser, til fornybare, biobaserte alternativer. Dette antyder at vi må øke uttaket av økosystemenes biologiske ressurser. Samtidig drar vi nytte av disse økosystemene på andre måter, ofte kvantifisert som økosystemtjenester. Disse strekker seg gjennom alt fra dets kapasitet til å produsere mat, regulere vannkvalitet og dets muliggjøring av ulike former for rekreasjon. Det er ukjent på hvilken måte det grønne skiftet vil påvirke de nordiske områders økosystemtjenester. Denne avhandling søker å gi en bedre forståelse av nettopp dette gjennom tre forskningsartikler. Avhandlingen undersøker seks nordiske nedbørfelt, i Danmark, Finland, Norge og Sverige. Den første artikkelen er en studie i sammenhengen mellom landskapets attributter og menneskers preferanser når de velger rekreasjonsområde. Forskningsmaterialet er basert på et diskret valgekspriment. Resultatene viser at respondenter i gjennomsnitt foretrekker en balansert blanding av jordbruk og skog, verken mer eller mindre intensiv landforvaltning, økt vannkvalitet, naturreservater og lokale arbeidsplasser i landbruket, skogbruk og fiske, og et ønske om mindre forekomst av oversvømmelse i vassdrag. Resultatene viser også noe variasjon mellom de ulike områdene, og mellom ulike typer respondenter. Den andre forskningsartikkelen presenterer et estimat av økosystemtjenestenes totale samfunnsverdi av i dag i de seks områdene, samt en analyse av dets variasjoner. Gjennomsnittsestimater av totalverdien strekker seg fra omtrent €400 ha^{-1} year^{-1} i det finske Simojoki, til €7,000 ha^{-1} year^{-1} i det norske Orrevassdraget. Mesteparten av verdien kommer fra aktiv verdsettelse av naturen i form av eksempelvis rekreasjon, men funnene viser stor variabilitet mellom og innad i områdene. Andre store økosystemtjenester er støtteområdene for landbruksvirksomhet, skogbruk og karbonbinding. Jordsmonnstype, skråninger, landskapsvariasjon, befolkningstetthet og tilgang på vann, viser alle signifikant korrelasjon til økosystemtjenestenes verdi. Den tredje artikkelen presenterer en analyse av potensielle effekter det grønne skiftet kan ha på verdien av økosystemtjenestene. Det er presentert fem ulike bioøkonomiske senarioer til rammeverket utviklet i artikkel II. Hver av disse senarioene er analysert for å finne hvilke endringer de villede til i henholdsvis landbruksendringer, sosio-geografiske endringer og økosystemtjenestene undersøkte nedbørfelt tilbyr i dag. Analysen finner at en fremskreden bioøkonomi med høy sannsynlighet vil øke verdien av økosystemtjenestene i sin helhet. Det er det bærekraftsfokuserte senarioet, og senarioet fokusert på å maksimere økonomisk produksjon som gir størst verdiøkning. Likevel er det også for disse senarioene stor variasjon mellom de ulike områdene, så vel som mellom ulike interessegrupper som på ulikt vis drar nytte av økosystemtjenestene. Dette antyder at bioøkonomisk politikk ikke bare vil påvirke den totale sosiale verdien av nedbørfelt, men også fordelingen av verdier innad i samfunnet.

Whether the universe is a concourse of atoms, or nature is a system, let this first be established: that I am a part of the whole that is governed by nature; next, that I stand in some intimate connection with other kindred parts.

- *Marcus Aurelius, 175 C.E.*



Synopsis

1 Introduction

1.1 Background

Humans and the natural world have had a strained relationship for as long as we have existed. As we grew in population and in technological abilities, so have the pressures that we put on our living environment. In consequence, about 150 years after the first societies started moving into a fossil-fuel based industrial revolution, it started to become apparent that we might be getting ourselves into serious trouble (Carson et al. 1962, Robinson 1973). In our search for materials and energy to keep economic growth on its upward trajectory, ecosystems, as suppliers of the resources we needed, took the toll. Deforestation for the creation of agricultural land and the harvest of timber became one of the staples of 20th century economic development, reaching a global peak of 151 million hectares of net loss during the 1980s (Williams 2003, Houghton 2016). This is an area half the size of India being cut down in a decade. Meanwhile, biodiversity drastically reduced across the world. Haddaway and Leclère (2020) report that in the period between 1970 and 2016, global species abundance declined by 68%, based on monitoring of 20,811 populations representing 4,392 species. They see the main causes for decline in changes in land and sea use, species overexploitation, spread of invasive species and disease, pollution and climate change.

As the impacts of human activity on ecosystems became more pronounced, ecologists and environmentalists became increasingly aware of the complex links between the healthy functioning of ecosystems and the underpinnings of human wellbeing. This suggested that our continued harvesting of resources from ecosystems would eventually severely damage our wellbeing. Alarm bells were rung, most famously by The Club of Rome in its 1973 'The Limits to

Growth' (Robinson 1973), but maximising economic growth remained the world economy's first priority. However, concern over the global degradation of ecosystems, species loss and climate change led to a shift in focus in environmental research and policy, from managing the limited supply of food, energy and mineral resources to the idea that we might be placing more pressure on ecosystems than their inherent resilience can withstand (Colombo 2001). This shift in focus towards what is now called sustainability gave rise to the concept of ecosystem services (World Commission on Environment and Development 1987).

1.2 A short history of ecosystem services

The term 'ecosystem services' originates from a paper by Westman (1977) in Science, titled 'How Much are Nature's Services Worth?' In this paper, Westman aims to answer the titular question by applying economic and accounting concepts and terms to our interactions with ecosystems. He concludes that instead of focusing on quantifying stocks of resources, we should aim to quantify flows stemming from ecosystem functioning. He then argued for closer understanding of these flows and how they impact human wellbeing. He also warned against using monetary measures to estimate value, because of our limited knowledge on ecosystem function and their societal benefits, making for unfair comparisons when measuring them on the same scale as other economic outputs.

Before that seminal paper, concerns already existed about the tense relationship between short term economic gain and the long-term degradation of ecosystems (Gómez-Baggethun et al. 2010), but these were mostly researched in the separate spheres of ecology and environmental economics (Costanza et al. 2017). In the years after the publication of Westman (1977), a newly integrated field of ecology and economics produced ecosystem services as a separate research topic, which since then has seen exponential growth (Costanza et al. 2017). A subsequent landmark was the publication of Costanza et al. (1997), a meta-analysis of global ecosystem services valuation studies, which brought ecosystem services into the research mainstream with a controversial estimate: that the societal value generated by the global biosphere is within the range of US$ 16 - 54 trillion per year. This conclusion evoked not only methodological questions, but also more fundamental ones: is it ethically right to put a monetary estimate on nature? What is the use of throwing together numbers sourced from various valuation methods? And if so, how do we deal with knowledge gaps and lack of data? How do we integrate the value of natural capital into economic decision making? Fundamental questions and continuous debate became a mainstay of the field since that landmark publication, but at the same time the valuation of nature became a productive research topic (Christie et al. 2008). In part due to the magnitude of the value estimates that this first global assessment made, policy makers also increasingly showed interest in the concept of ecosystem services (Braat and de Groot 2012). This resulted in a next landmark effort, the Millennium Ecosystem Assessment (MA 2005). This was the result of four years of study by 1,300 scientists at the behest of the United Nations. It concluded that degradation of ecosystems presents a threat to human wellbeing due to reduced generation of ecosystem services. A second international study, The Economics of Ecosystems and Biodiversity (TEEB 2010), was undertaken by the UN Environment Programme and garnered extensive news coverage, further pulling the concept of ecosystem services into the public sphere.

Since then, researchers have made attempts to find consensus in what an ecosystem service is, how to measure its value and what to do with these values. Multiple frameworks have been developed, from the European Environment Agency's Common International Classification of Ecosystem Services (Haines-Young and Potschin 2017), to the United States Environmental Protection Agency's Final Ecosystem Goods and Services and the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services' Nature's Contributions to People (Díaz et al. 2018), and ecosystem services studies have been applied to a variety of topics, from local river restoration projects to national natural capital accounts and global assessments of ecosystem degradation. However, all of these frameworks follow their own methodology and concepts, ranging from differences in detail to fundamental disagreement on the definition of what an ecosystem service is, or if we should even use that term (Díaz et al. 2018). Agreement on the questions that Westman (1977) posed does not yet appear in sight.

This short history serves to show not only the origins of the concept of ecosystem services, but also how much is unresolved, ranging from basic questions on our relationship to nature, to debate on the use and application methods of valuation of ecosystem services. This thesis does not aim to answer these questions, but it uses the language and tools of ecosystem services because it has become a mainstay in policy makers' vocabulary, and ecosystem services frameworks have proved invaluable as tools for estimation and communication of our complex connections and dependencies to our living environment.

1.3 Bioeconomy as a solution to our problems

Around the turn of the century, when the concept of ecosystem services developed its exponential growth into the science and policy mainstream (Fisher et al. 2009), the term 'bioeconomy' started appearing. The term gained popularity after the European Union's Biotechnology Strategy was launched in 2002, which was linked to the goal of reaching a 'Knowledge Based Bio-Economy' (Patermann and Aguilar 2018). As this origin suggests, the concept was grounded in technological and industrial development. The strategy aimed at developing new drugs, foods and chemicals for industrial use based on biological resources. These developments would require specialised production chains, forming a new bio-based economy, or bioeconomy. The Organisation for Economic Co-operation and Development in 2004 also published a document, 'Biotechnology for sustainable growth and development', that defined a biobased economy as 'a concept that uses renewable bioresources, efficient bioprocesses and industrial clusters to produce sustainable bioproducts, jobs and income' (OECD 2004). This definition made clear the link between a bioeconomy and our complicated relationship with the natural world: by requiring bioresources on an industrial scale at the one hand, but on the other hand providing the potential to rid ourselves of our addiction to fossil resources, a bioeconomy could transform how we relate to the natural environment.

Since then, multiple countries have implemented national bioeconomy strategies (Dietz et al. 2018). In 2012 the EU launched its Bioeconomy Strategy (Geoghegan-Quinn 2012). In it, the concept as defined by the OECD was elaborated on as a means to reduce reliance on fossil resources and create a more sustainable economy. It was awarded a budget of close to €2 billion. However, what exactly a bioeconomy would look like was not clear. Bugge et al. (2016) recognised this ambiguity and performed a literature review, finding that there are multiple visions for what a bioeconomy is, ranging from a focus on bio-technology research to the promotion of ecologically sustainable land use. Ambiguity notwithstanding, since the launch of the EU Bioeconomy Strategy, European countries have made their intention to further develop the bioeconomy explicit. Germany for instance has a National Bioeconomy Strategy, overseen by the German Bioeconomy Council and with the aim of transitioning Germany to a bioeconomy (Issa et al. 2019).

In 2017, the Nordic countries (Denmark, Finland, Iceland, Norway and Sweden) also launched a cooperative strategy for transitioning to a bioeconomy (Belling 2017). This strategy focuses on replacing fossil resources with biological ones, upgrading current production chains for efficiency, making the economy more circular and realising closer collaboration between stakeholders. What this transition would look like in practice raises questions closely linked to our interactions with our living environment, and these questions form the starting point of this thesis.

1.4 Problem statement

Now that the Nordic countries have committed to a green shift to a bioeconomy, new questions arise. First and foremost: what goals should we set to create a bioeconomy? Does this mean the complete elimination of all fossil fuel-based goods and energy sources? And if so, how will societies achieve this? Will this need to come with a reduction in production of new materials and energy, or can we continue to increase these flows by solely relying on renewable, biological resources? And where will these resources come from? In 2018, the bioeconomy in the Nordic countries was mostly focused on the food and forest industry (Refsgaard et al. 2018). Rönnlund et al. (2014) estimated that total turnover of the bioeconomy sectors in the Nordic countries is about €184 billion per year, which is 10% of the total economy. The renewable energy share in total energy production varies from 100% in Iceland, which predominantly produces geothermic energy, to 6% for Norway, which is one of the largest oil producers in Europe. The fact that the bioeconomy in 2014 constituted about 10% of the total economy suggests that a further, major transformation is necessary to reach a state of bioeconomy as described in the Nordic Bioeconomy Strategy. Even if in the long term resource efficiency and the implementation of a circular bioeconomy would reduce our dependency on large amounts of biological resources (Ellen MacArthur Foundation 2013), on shorter time scales it is likely that agricultural production and forestry will need to expand and intensify (Issa et al. 2019), once again changing our relationship with the land around us.

This leads us back to ecosystem services. If we intensify land management in the Nordic countries, what will happen to its ecosystems, and in turn, what will happen to the ecosystem services that we generate by interacting with them? Nordic catchments, or river basins, are core geographic entities that generate ecosystem services (Barton et al. 2012). The wellbeing of those that live in them and visit depends on access to clean water for drinking and recreation, healthy soils suitable for forestry and agriculture, flood protection, and carbon sequestration by the biota living in Nordic catchments. These flows of ecosystem services can be altered by changing land management to accommodate a growing bioeconomy, and as Dietz et al. (2018) point out, so far the transition to a bioeconomy has not yet been strongly linked to the concept of ecosystem services in policymaking. To allow for a societally optimal bioeconomy, it is therefore essential to develop a better understanding of the links between land management and the value of ecosystem services in Nordic catchments.

The questions I pose are therefore:

- Can we apply the concept of ecosystem services to successfully estimate the effects of a transition to a bioeconomy on a Nordic scale?
- If so, what are potential effects of such a transition on the societal value of generated ecosystem services from Nordic catchments?

In the following chapter, I will describe the current state of knowledge on ecosystem services estimation, on the value of ecosystem services in the Nordic countries and on what a bioeconomy might look like in the Nordics. In chapter 3, I describe how I planned to answer the research questions, by setting up a series of operationalised aims within the scope of three linked original research papers. In chapter 4 I describe and justify the methods we have used, and in chapter 5 I describe the main findings per paper. In the final chapter, I discuss these findings by answering the research questions, assessing how this work can advise policy makers, how it fits in the current scientific body of work, and conclude with the implications of this work for further research.

2 The state of knowledge

2.1 Ecosystem services – definitions and methods

The field of ecosystem services is broad, with researchers applying a variety of basic definitions and assumptions onto an even wider variety of quantification and valuation methods (Boyd and Banzhaf 2007, Bouma and Van Beukering 2015, Boerema et al. 2017, Potschin-Young et al. 2018, DeWitt et al. 2020). At its core, the concept is about the relationship between human wellbeing and our natural environment, but from there the divergence starts (Table 1).

The groundwork for many early ecosystem services frameworks is based on the Millennium Ecosystem Assessment (MA 2005). This conceptual framework was the first to divide ecosystem services into provisioning, regulating, cultural and supporting services, which became a cornerstone of subsequent frameworks and the most common way of categorising ecosystem services. Provisioning services are flows of goods and energy, such as food production and timber, regulating services are those that regulate effects, such as flood regulation and soil retention, cultural services are related to experience, such as recreation and cultural heritage value, and supporting services are those that support any of the other service types. The MA took the starting point of a linear relationship, from stocks of natural capital that generate flows of ecosystem services as presented in Costanza et al. (1997), and added feedback loops and drivers of change, more closely linking human activity to ecosystem condition (Schreckenberg et al. 2018) and thus to the societal benefits of the services they generate. The MA also opened the door to the application of systems approaches, instead of simple analytical methods to take into account complex system behaviour, such as thresholds, feedbacks, non-linearities and phase shifts (Schreckenberg et al. 2018).

The Economics of Ecosystems and Biodiversity (TEEB 2010) was developed as an expansion of the MA, focusing more on economic valuation of ecosystem services, but it was not widely taken up in practice (Wegner and Pascual 2011, Schreckenberg et al. 2018). As opposed to the MA it separated services from benefits, to clearly distinguish the value produced by the ecosystem (service), from a final benefit that can also include human input (Finisdore et al. 2020). TEEB was also the first major framework to incorporate another influential concept in ecosystem services quantification: the cascade model by Haines-Young and Potschin-Young (2010). This concept aims to capture the relationship between ecosystems and human well-being through a series of quantifiable steps, each flowing into the next: from ecosystem structures and processes, to ecosystem functions, to ecosystem services which can finally be translated into concrete human benefits.

Table 1. An overview of key ecosystem services frameworks. This table shows a list of frameworks that are widely applied, their years of first publication, main organisation supporting its development, and key concepts that each framework introduced or applied.

Short name	Year	Main organisation	Key concepts
MA	2005	United Nations	Provisioning, regulating, cultural, supporting services
TEEB	2010	United Nations Environment Programme	Focus on economic value Split services from benefits
NEA	2011	Government of the United Kingdom	National application Spatial analysis
SEEA	2012	United Nations Statistical Commission	Natural capital accounting
CICES	2013	European Environment Agency	Hierarchical structure Final services
MAES	2013	European Commission Joint Research Chair	Spatial analysis
IPBES	2015	United Nations Environment Programme	Nature's benefits to people
NESCS-Plus	2020	United States Environmental Protection Agency	Final services Direct link to beneficiaries

While MA and TEEB were meant as generic frameworks first applied to global assessments of ecosystem services, national and regional adaptations soon followed, which further crystallised definitions and methods. In the United Kingdom, the National Ecosystem Assessment (NEA) was an adaption of the MA framework to estimate the value of ecosystem services generated by the entirety of Great Britain (Bateman et al. 2013). It also used the four categories of provisioning, regulating, cultural and supporting services, but an extra step here was adding a spatial dimension by linking its estimates to spatially referenced environmental data across all of Great Britain.

The United Nations in the meantime attempted to apply their System of National Accounts to ecosystem services as directly as possible, using the same structure, concepts, definitions and classifications, producing the System of Environmental-Economic Accounting (SEEA). Its central framework was first published in 2012 (United Nations et al. 2017), and in 2021 the SEEA was adopted by the United Nations Statistical Commission, enabling countries to incorporate natural capital into their official national capital accounting.

Another intergovernmental collaboration under the umbrella of the United Nations is IPBES, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Díaz et al. (2015) developed a framework under the IPBES banner, broadly following the provisioning-regulating-cultural services structure, but expanding the framework to include more than these traditional ecosystem services, under the label 'nature's benefits to people'. The aim of this expansion is to widen the scope from the ecosystem services terminology of western science, to a concept of benefits that is more inclusive of indigenous values, and to the idea that the effects of nature on achieving a good quality of life differ for different people and in different contexts.

In the same period, The European Commission developed CICES, the Common International Classification of Ecosystem Services (Haines-Young and Potschin 2017). This framework is designed using the same basic concepts as the MA, TEEB and IPBES classifications, but it is the first to apply a hierarchical structure for classifying ecosystem services (Finisdore et al. 2020). It also incorporates the concept of final ecosystem services as first defined by Boyd and Banzhaf (2007). In that original publication, final services are defined as 'components of nature, directly enjoyed, consumed, or used to yield human well-being'. CICES applies a basic form of that definition by removing the MA's supporting services, since they are not directly linked to human wellbeing. The CICES structure has been used in MAES (Mapping and Assessment of Ecosystems and their Services), another framework for ecosystem services quantification, which applied it using a spatial analysis on a European scale (Maes et al. 2016).

In the United States, the Environmental Protection Agency developed a framework based on the same definition of final services as CICES, only using a more orthodox application. Its most recent framework is NESCS-Plus (Newcomer Johnson et al. 2020), and in it, all ecosystem services are directly linked to a type of ecosystem as well as to a (human) beneficiary, in an attempt to cut out ambiguity. The strict application of Boyd and Banzhaf (2007)'s definition of a final service in NESCS-Plus also excludes some services that in other frameworks are classified as regulating: regulation of soil quality for example, a final service under CICES, is here only a supporting process for final services such as the supporting environment for growing crops (which farmers directly benefit from). Under CICES, both cultivated plants and regulation of soil quality are final services, which can lead to double counting of benefits, since the latter contributes to the former.

Taking a step back to look at the proliferation of ecosystem services frameworks described here, the concept of a relationship between human wellbeing and the natural environment remains at the core to this day. Over time, from the MA to NESCS Plus, definitions have been polished and reshaped, but for the most part these are all attempts to quantify value in traditional economics and accounting terms: an ecosystem is a stock of capital that generates a flow of returns on investment. This notion has been challenged, most explicitly by the IPBES, but never overturned and it is still the dominant paradigm in the field. So looking back on that original paper by Westman (1977) and his advice to consider the quantification of flows stemming from ecosystem functioning while warning for the risks of monetary valuation, the fundaments have not changed. For someone working in ecosystem services today, finding these conclusions in Westman's paper might suggest we have not progressed much since then: these are still unresolved points of debate in the scientific community (Schröter et al. 2014, Díaz et al. 2015, Boerema et al. 2017, Costanza et al. 2017, La Notte et al. 2017, Kenter 2018, Potschin-Young et al. 2018). In the meantime, however, more sophisticated frameworks and modelling tools, as for instance in the MAES and NEA's spatial analyses have aided in creating better understanding of the dynamics of the interactions between human activity and ecosystems. On the other hand, opposing viewpoints (Díaz et al. 2015, Braat 2018, Kenter 2018, Dasgupta 2021) suggest that the field is not converging on its understanding of key concepts, and possibly equally important, of its goals. This thesis operates within this fractured field, and in it I do not aim to create a new conceptual basis for ecosystem services estimation, but rather to apply existing concepts, definitions and methods that are most suitable to estimating the effects of a green shift on ecosystem services value in Nordic catchments. In chapter 3 I describe how this application took shape and was informed by the current state of knowledge.

2.2 Ecosystem services in Nordic catchments

Catchments, watersheds, or river basins are geographic entities bound by hydrology. The European Environment Agency defines a catchment as an area from which surface runoff is carried away by a single drainage system[1].Catchments are suitable units of study for ecosystem services estimation because many services are directly linked to water quality and quantity, making them naturally bounded, semi-closed systems. This suitability shows in the number of ecosystem service studies specifically targeting catchments: Kaval (2019) found a total of 103 published studies with specific reference to ecosystem services and rivers or catchments in the period 2010-2016 alone.

In the Nordics, catchments are known to provide a wide variety of ecosystem services (Barton et al. 2012, Vermaat et al. 2020). In 2011, the Nordic Council of Ministers acknowledged a knowledge gap in light of the growing body of work on ecosystem services flowing from the publications of MA (2005) and TEEB (2010): not enough was known of the value of ecosystem services provided by Nordic catchments. It therefore commissioned a study to fill this gap (Barton et al. 2012). However, this study's budget did not allow for in-depth assessment of ecosystem services value. Rather, it was a compilation of valuation work done previously. The study looked at five common types of valuation methods for ecosystem services: stated preference, revealed preference, production/damage functions, cost-based valuation, and benefit transfer, all common valuation tools for non-market goods and services. They found that in Norway, value estimates for food production, water flow regulation and purification, and opportunities for recreation were most common. In Sweden and Finland, valuation of food and water production, water purification, opportunities for recreation and landscape aesthetics were most common. In Denmark, the focus was on water purification and opportunities for recreation and landscape aesthetics. The authors' main findings were that food and water supply, as well as recreation, had been the main focus in the aggregate of valuation studies. They found many studies valuing recreational possibilities in the context of water quality under the Water Framework Directive (WFD 2000), which requires surface waters to have a "good ecological status", but they warn that benefit transfer of such estimates to other catchments produces results with limited reliability. They advised to perform more primary valuation studies across representative Nordic populations, and to focus on spatially explicit studies to show conflicts of interest between different stakeholders.

[1] https://www.eea.europa.eu/archived/archived-content-water-topic/wise-help-centre/glossary-definitions/catchment-area

Since then, no concerted efforts were made to compile knowledge on ecosystem services value from Nordic catchments in general, but separate studies of various scopes and scales were performed. Lankia et al. (2015) estimated the value of nature-based recreation in different Finnish regions using travel cost analysis based on survey data, finding values per recreational trip in a range between €2 and €252 person^{-1}. Crop production as an ecosystem service has been valued in Odense, Denmark, by Lehmann et al. (2020) at €1,067 ha^{-1} year^{-1}. Nikodinoska et al. (2018) took a more integrative look by estimating total value of several ecosystem services, but they only studied one specific region in Sweden. They estimated total economic value at around €1,200 ha^{-1} year^{-1} from forest areas and €600 ha^{-1} year^{-1} from agricultural areas. On a wider geographic scale, Bartlett et al. (2020) made an assessment of carbon storage in Norwegian ecosystems, though they did not assess the economic value of this service. Similarly, Odgaard et al. (2017) estimate ecosystem services from wetlands across all of Denmark, but did not include a valuation. This type of study illustrates a general trend, also pointed out by Magnussen et al. (2014) in relation to freshwater management: ecosystem services are studied increasingly in the Nordic countries, but relatively few studies value them, and more focus is placed on relating them to other concepts of ecosystem management, such as the Water Framework Directive's requirements for good ecological status of surface waters.

This overview shows that ecosystem services generated by Nordic catchments have been extensively studied, but significant gaps remain. Studies incorporating economic value are relatively rare, and those that are performed typically focus on one specific region, or on a small selection of ecosystem services. Not all ecosystem services received equal attention, and the uncertainties that come with value transfer (Navrud and Ready 2007, Bateman et al. 2011) suggest that estimates for those that have been studied extensively, such as recreation, cannot easily be extrapolated to other areas. Due to the wide variety of concepts, definitions and methods applied in the field of ecosystem services, the mosaic of separate studies outlined in this paragraph cannot be integrated into a consistent overview of ecosystem services value. This thesis then aims to help filling the gap that was already described by the Nordic Council of Ministers (Barton et al. 2012) ten years ago, by using a consistent set of definitions and methods on a complete set of ecosystem services generated in catchments across four Nordic countries.

2.3 Bioeconomy and its implications for Nordic catchments

Even if the precise aims of a green shift to a bioeconomy are not defined, the Nordic Bioeconomy Initiative was set up under the Nordic Council of Ministers to set a sustained trajectory in motion (Gíslason and Bragadóttir 2017). They acknowledged the need for clear targets and indicators (NCM 2017), but none exist so far. A key element of such a transition is apparent though, even if its magnitude is unclear: replacing flows of fossil materials and energy will require increased growth and harvesting of biomass. Currently, the bioeconomy makes up around 10% of the Nordic economy (Gíslason and Bragadóttir 2017), so expansion only depends on technological and economic viability of further developing existing and new bioeconomic production chains. The start of these production chains, collecting raw materials as inputs, will likely be the main process affecting ecosystems and is therefore the focus of this thesis. Refsgaard et al. (2018) name 'fisheries, aquaculture, forestry, agriculture and bioenergy' as the likeliest sources of new raw materials for the bioeconomy, and out of these forestry, agriculture and bioenergy will most likely impact Nordic catchments. I will therefore focus here on the possible implications of increased resource extraction from these three sources.

Nordic forests supply 28% of the Nordic bioeconomy, and 73% of Finland and 69% of Sweden are covered in forest (Refsgaard et al. 2018). Both increasing the area of production forest and intensifying biomass harvesting from current productions forests will likely alter forest ecosystems. Eyvindson et al. (2018) examined trade-offs between increasing timber extraction and biodiversity and non-wood ecosystem services in seventeen catchments in Finland. Biodiversity was evaluated as habitat availability, while carbon storage and bilberry yield were used as ecosystem services. They found that increasing timber flows decreases habitat availability and both these ecosystem services, as well as variation between landscapes. They also found that such losses can be limited with careful landscape planning, for instance by targeting increased timber harvesting to those sites with high production potential and low biodiversity and other ecosystem service provision. When shifting focus to expansion of wood production areas instead of intensification, Dimitriou and Mola-Yudego (2017) studied the establishment of poplar and willow plantations on agricultural land in Sweden. These tree species are known for their fast growth and dense plantation tolerance, especially for willows: in Sweden, they are planted at up to 16,000 trees ha^{-1} (Dimitriou and Mola-Yudego 2017). This study found that not only do plantations with fast growing trees produce large amounts of biomass, they also result in significantly lower nutrient leaching compared to agricultural crop production, especially for willows, as well as higher soil carbon storage. Since expansion of high intensity forestry is more efficient on relatively fertile soils, which are currently predominantly used for agriculture, new production forest will likely be planted on what are currently (marginal) agricultural fields, if an increase of wood production is part of the green shift (Kumm and Hessle 2020).

Growing crops puts more pressure on its surrounding environment than forestry (Carpenter et al. 1998, Bechmann et al. 2005), so increasing biomass production from agriculture, either through intensification or expansion of areas, can have significant consequences on ecosystems, soil quality and water quality. Marttila et al. (2020) estimated the potential impacts of increased biomass production on surface water quality in Nordic catchments. They recognised eutrophication, brownification and biodiversity loss as the main threats to aquatic ecosystems, and suggest that increased fertilisation and increased use of marginal land areas can lead to increased accumulation of phosphorus in soils, adding pressure on watercourses due to nutrient loading. Historical trends show increased intensification of some farming regions, while the more extensively farmed regions are being increasingly abandoned. Marttila et al. (2020) state that increased need for biomass can exacerbate this process, putting already strained ecosystems under even more pressure. This can be of special significance for Denmark, where over 60% of land cover is already agriculture (Marttila et al. 2020). They acknowledge that here as well measures can be taken to mitigate negative effects on water quality, for instance by constructing wetlands, ponds and buffer zones to reduce nutrient runoff, which have already proven effective in Danish agriculture (Vodder Carstensen et al. 2020).

Finally, changes in production of bioenergy use can change Nordic catchments beyond the effects of changing agriculture and forestry. For example, peat extraction from mires and bogs is an issue in the Nordic countries (Kløve et al. 2017, Juutinen et al. 2019, Saarikoski et al. 2019). Peat is typically considered a fossil fuel due to its large regeneration time and high carbon emissions when burnt, so a green shift will likely reduce or eliminate peat extraction from Nordic catchments (Kløve et al. 2017). Juutinen et al. (2019) studied the effects of peat extraction in Finland, the predominant location of peat extraction in the Nordics. They found that a small reduction in extraction can lead to substantial decreases in biodiversity loss and water loss from the peatland (which typically contains high concentrations of dissolved organic carbon, causing brownification of rivers and lakes). This suggests that a green shift can have positive effects on water quality and its related ecosystem services in Nordic catchments where peat is currently extracted.

The overview in this paragraph suggests that the effects of a green shift on Nordic catchments are uncertain, likely strongly spatially dependent, but potentially significant. Since suitable areas for biomass production, especially agriculture, are limited in the Nordic countries, expansion can possibly come with increased pressures on already sensitive catchments. The uncertainties in effects not only stem from the complexity of ecosystems, but also from the complexity of society: the shape of the bioeconomy is still unclear. In an attempt to outline this shape, Rakovic et al. (2020) constructed five scenarios of possible bioeconomy development, called the Nordic Bioeconomy Pathways (NBPs). Based on the Shared Socioeconomic Pathways (O'Neill et al. 2014), these scenarios describe possible states of a bioeconomy in the Nordics in 2050. They differ in the way society changes, ranging from a focus on efficient resource use and consideration of environmental impacts, to fully prioritising economic output within a global economy. These NBPs are qualitative storylines which cannot be used directly to assess effects on catchments, but can be used as the basis for a quantitative articulation. The transformation of the NBP storylines into a set of quantitative variables is part of this thesis and forms the basis for its scenario analysis.

3 Relation between the papers

To answer the main questions arising from our problem statement and to fill in the knowledge gaps described in the previous chapter, I broke the research up into several connecting parts. Each part resulted in a paper which served as input into the next, to finally be able to test a framework of ecosystem services estimation on a set of scenarios for transition to a bioeconomy in the Nordic countries (Figure 1).

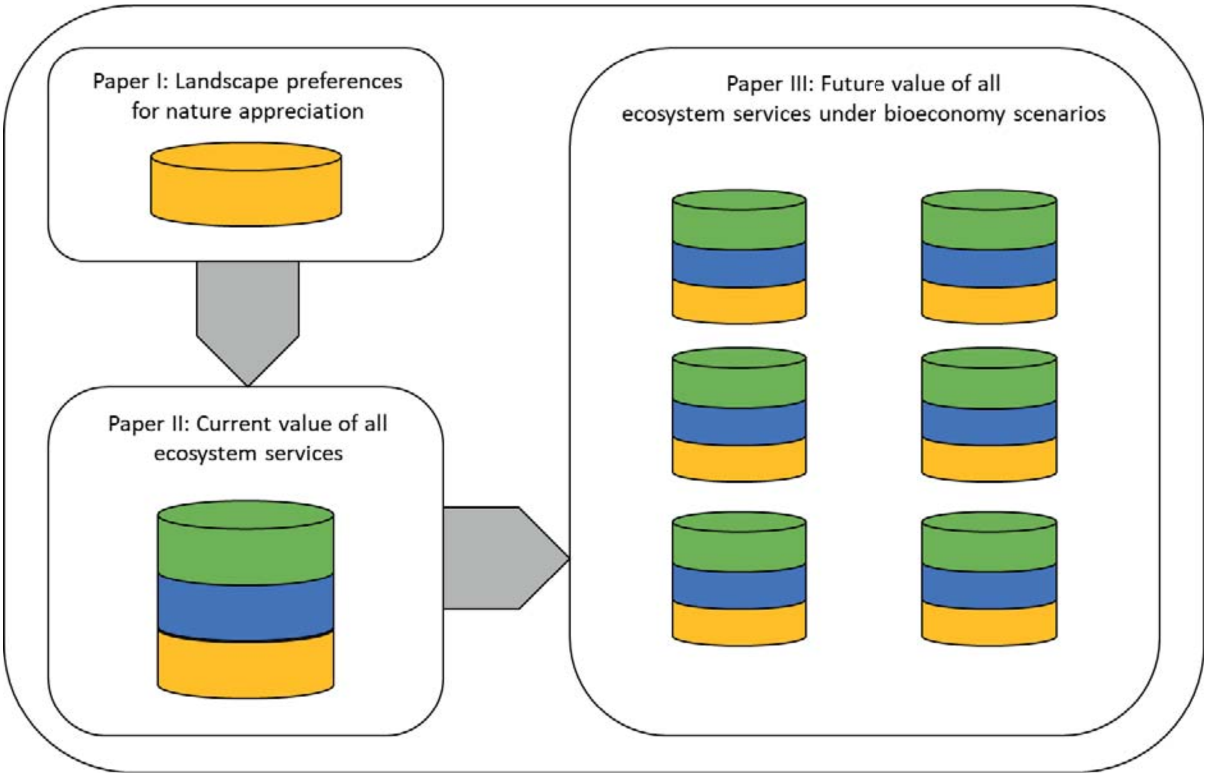


Figure 1. The thesis structure. This figure shows how the three papers at the core of this thesis connect to answer the main research questions. Each coloured cylinder represents a category of ecosystem services. Each stack of cylinders represents a complete set of relevant ecosystem services.

In order to start estimating the societal value of ecosystem services generated in a study area, data needs to be available on all relevant ecosystem services. Depending on quantification method, most ecosystem services can be quantified using publicly available data on environmental quality and flows of resource production. However, quantification of one type of ecosystem services requires data that is more specific, less generally applicable, and therefore typically not collected for other purposes: the appreciation of nature in a specific area by the general public. A large knowledge base of research on the value of cultural ecosystem services, such as recreation, already exists (Martin-Lopez et al. 2009, Boerema et al. 2014, Van Berkel and Verburg 2014, Juutinen et al. 2017, Pokki et al. 2018), but a key issue with such estimates is transferability (Bateman et al. 2011). The value of cultural services like recreation depends on location-specific variables, such as the socio-demographic profile of the population, cultural traits in society, access for visitors, types of recreational possibilities and landscape aesthetics (Garcia-Martin et al. 2017). For this reason, transferring values found in previous work to other sites comes with large uncertainties (Bateman et al. 2011, Brown et al. 2016), and collecting data specific to the study area is generally preferred. This implied that, even though for other ecosystem services I could rely on publicly available data and previous research, for the appreciation of nature by inhabitants and visitors, I would do better to collect the data myself. This first step resulted in Paper I: 'Appreciation of Nordic landscapes and how the bioeconomy might change that: results from a discrete choice experiment'. The aims of this paper were:

1. To quantify the preference and willingness to pay for landscape changes that can arise from the transition to a bioeconomy for consumers of cultural ecosystem services.
2. To explain the observed variation in these preferences from catchment and population characteristics.

With the completion of the survey work, I had access to enough data to start estimating a baseline of total ecosystem services value. The logic behind this is that to be able to estimate the effects of change due to a bioeconomy, I needed a quantified starting point: the current societal value of ecosystem services generated by Nordic catchments. Moreover, to know how land use change and societal change can affect ecosystem services value, I needed information on the relationship between landscape and socio-geographic characteristics and the generation of ecosystem services in these catchments. A final point of interest before moving onto bioeconomy effects was the distribution of ecosystem services value across societal stakeholders. This would allow for later analysis of variation in distributional effects under bioeconomy scenarios. Estimating these baseline values and relationships thus became the goal of Paper II, titled 'Estimating societal benefits from Nordic catchments: An integrative approach using a final ecosystem services framework'. For this paper we collected data on the same six Nordic catchments as in Paper I, and we operationalised the study aim into the following questions:

1. Which services are most important in these six Nordic catchments, and what underlying environmental and societal factors explain the variation in ecosystem services value?
2. Which stakeholder groups benefit from which services and do we observe potential spatial conflicts in their interests?

The findings from this paper then formed the basis for the final step: estimating the effects of transitioning to a bioeconomy on the total economic value of ecosystem services generated by Nordic catchments. To do so, we needed two sets of inputs: the results of the previous paper that estimated the current value of ecosystem services, and a set of quantified bioeconomy scenarios that could be linked to the same ecosystem services framework that was developed in the previous paper. We described the results of this exercise in Paper III, named ' The value of change: a scenario assessment of the effects of bioeconomy driven land use change on ecosystem service provision'. In it, we aimed to answer the following research questions:

1. What are the effects of the NBPs on ecosystem services value generated by our six Nordic catchments?
2. How do scenario effects vary among and within our study areas?
3. How are scenario effects distributed across different stakeholder groups and where might conflicts arise?

This paper thereby answers the overarching research questions of this thesis by showing an application of an ecosystem services framework to estimate the effects of socio-geographic and land use change caused by the green shift to a bioeconomy across six Nordic catchments.

4 Methods

4.1 Study area selection

I aimed to quantify flows of ecosystem services under various types of land use across a region covering over 15 degrees of latitude and five climatic zones (Kottek et al. 2006), but was limited by time, budget and data availability. This meant that I needed a strict set of selection criteria for which catchments to study. I used the following:

1. Each of the four mainland Nordic countries (Denmark, Finland, Norway and Sweden) has to be represented by at least one catchment.
2. Each catchment needs enough human habitation to allow for survey work on ecosystem services related to public appreciation of nature.
3. Since the bioeconomy will mainly affect forestry and agriculture, one or both of these need to exist in each catchment.
4. The catchments in total need to cover the majority of the geographic spread of Fennoscandia.
5. When more than one catchment is studied in a single country, there should be a distinct contrast in land use and population density between them.
6. We cannot study more catchments than we can survey over the course of two summers, as the possibility for recreation, an important ecosystem service, had to be quantified via surveys.
7. For each catchment, data needs to be available on the required environmental and economic indicators for ecosystem services estimation.

This set of criteria led to the selection of six catchments: Haldenvassdraget, Orrevassdraget, Odense, Simojoki, Sävjaån and Vindelälven. These six catchments cover most of the latitudinal range of the four continental Nordic countries, have varying human population densities (though be sufficiently populated to allow for fieldwork on recreational visits), each contains a mixture of land covers that includes forest and agriculture, and each is monitored for environmental variables including water quality, water quantity and a variety of economic activities related to the natural environment (Table 2, Figure 2). To allow for a clear distinction in effects of transitioning to a bioeconomy on contrasting types of catchments, we divided them into two types:

1. Peri-urban catchments where at least 30% of the total area is used for agriculture and population density is more than 40 people per km^2.
2. Rural catchments where at least 67% of the total area is covered by forest and population density is lower than 20 people per km^2.

Table 2. Study area descriptions. This table shows size and land use for forest, agriculture, water bodies, urban area and nature reserves as percentage of the total area, as well as average population density and the proximity of the closest city to the catchment. We took land use values for forest, agriculture, water bodies and urban area from 2016 CORINE land cover data (Buttner et al. 2000). We took the area of nature reserve from GIS-databases of the national environmental agencies. We used population data from 2019 estimates by WorldPop (worldpop.org). We defined cities as having more than 50,000 inhabitants. Table from Paper I.

	Halden-vassdraget[2]	**Orre-vassdraget**	**Odense**	**Simojoki**[3]	**Sävjaån**	**Vindelälven**
Country	Norway	Norway	Denmark	Finland	Sweden	Sweden
Catchment size (km^2)	1,006	102	1,199	1,178	733	778
Forested area (%)	67	3	6	76	60	75
Agricultural area (%)	17	70	80	2	32	6
Water area (%)	6	15	1	1	1	2
Urban area (%)	1	8	12	0	2	1
Nature reserve area (%)	3	10	0	14	2	1
Population per km^2	16	167	205	1	41	5
Closest city (with distance from catchment in km)	Oslo (20)	Stavanger (15)	Odense (0)	Oulu (70)	Uppsala (0)	Umeå (20)

[2] Northern end, approximately from Bjørkelangen to Ørje
[3] Western end, between Hosio and Simo.

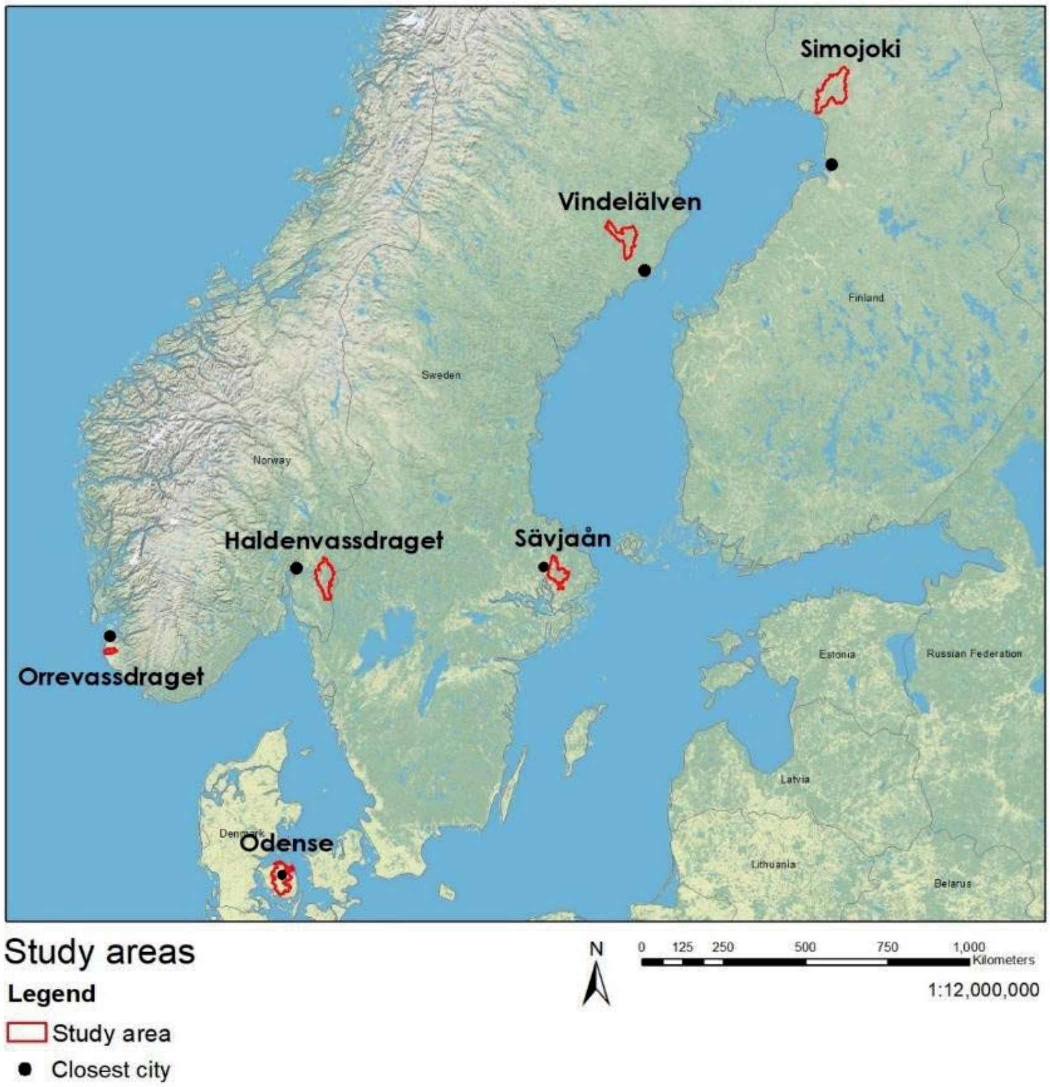


Figure 2. A map showing the positions of the different catchments across the Nordic countries. The basemap is provided by ESRI[4]. Study site boundaries are shown in red. Black dots show the city closest to the sub-catchment as described in Table 2. This map illustrates the spatial range of study sites across the Nordic countries, as well as the range of dominant land use types. Orrevassdraget, Odense and Sävjaån are close to cities and in areas with relatively large proportion of agricultural land, while Haldenvassdraget, Vindelälven and Simojoki are further from densely populated areas and contain relatively little agricultural land. Figure from Paper I.

[4] **Esri. "World Topo Base". February 5, 2020. https://www.arcgis.com/home/item.html?id=3a75a3ee1d1040838f382cbefce99125. (September 14, 2020).**

4.2 Estimating the relationship between landscape and recreation

The aim of Paper I was to estimate preferences of inhabitants and recreational visitors of the catchments for attributes of the landscape. Quantifying the relationship between characteristics of a catchment and its value generated by recreational opportunities and passive nature appreciation requires statistical analysis. We chose to use a discrete choice experiment (DCE) for this. DCEs are suitable for estimating preference among alternatives, in which the alternatives consist of a set of attributes (Adamowicz et al. 1998). This made the method suitable for our goal as well, since this allowed us to ask respondents to state their preference for various elements of the landscape and its management within our catchments. In a DCE, respondents are asked to choose between a set of alternatives (typically three), each of which has a different combination of attribute levels. A key element in most DCEs is the addition of a monetary attribute (Bennett and Blamey 2001), by having respondents choose a set of variables including a certain level of tax, representing a cost to the respondent. Monetary value for each attribute level can then be inferred using statistical analysis if the sample size is large enough.

We designed the DCE around a set of catchment attributes that fulfilled the following requirements:

1. Each attribute will potentially be affected by the transition to a bioeconomy.
2. Each attribute likely contributes to the value people place on recreating in the area.
3. The above two requirements need to be valid for each of the six catchments.
4. There can be no more than six non-monetary attributes (to minimise respondent stress).

We tested attributes on requirement 1 and 3 by consulting literature and experts on each catchment within the Norwegian Institute for Bioeconomy Research, the Norwegian University of Life Sciences, Aarhus University, the Swedish University of Agricultural Sciences and the Natural Resources Institute Finland. We tested attributes on requirement 2 by consulting literature and local partners (see Paper I). This process led to a complete list of attributes (Table 3), after which we set appropriate levels for each of them in all catchments using current levels as a starting point. We designed the survey so that each respondent had to fill out five choice cards, in an attempt to strike a balance between collecting enough data points and not overwhelming respondents. For each catchment, we designed six different configurations of choice cards, based on a D-efficient design using NGene (version 1.2.0). Aside from the DCE, the survey also contained questions on number of visits, types of recreation, opinion on the current state of the landscape and various socio-demographic questions, to be used as covariates in the statistical analysis.

Table 3. Landscape attributes used in the DCE. This gives a qualitative description of each of the attributes presented to respondents. Table adapted from Paper I.

Attribute	Description
Share of agriculture and forest	The percentage share of agricultural land and forested land in total land use in the study area. In Orrevassdraget, this was replaced by the shares of cultivated and uncultivated land due to the absence of forested area.
Agricultural and forest management intensity	The intensity of land use management, qualitatively described as the labour and machinery used, as well as the rate of biomass production and harvesting.
Water clarity	Qualitative levels of the clarity of water in rivers and lakes in the study area. In Simojoki the clarity was changed to water colour, since total organic carbon concentrations and related effects on colour have increased significantly due to changing climate and land use here (Lepistö et al. 2014).
Nature conservation	The percentage share of land used as natural conservation area in total land use in the study area.
Flood frequency	The frequency of floods that cause damage to land, infrastructure and property in the study area, described as one flood per a certain amount of years.
Local rural employment	The percentual change in employment in agriculture, forestry and fishery.

We collected the data during two summer seasons of on-site fieldwork, in 2018 and 2019, using paper questionnaires. By performing face-to-face interviews, we minimised risk of misinterpretation, since a qualitative pre-test in Haldenvassdraget had shown that some respondents could struggle with the complexity of the DCE. It would also allow us to reach respondents that would be unreachable using panel data, such as temporary visitors. In all six catchments, we used similar data collection tactics: we visited local recreation hotspots, public spaces, cafés, museums, municipal offices and went door-to-door, to cover as wide a range of respondent types as possible.

We then analysed preference for the levels of the various attributes using mixed logit (MXL) models in NLOGIT 6 (Greene 2016). An MXL model is a more complex version of a conditional logit model, in which the coefficients for preference can be random according to any distribution, so as to take into account preference heterogeneity (Train 2009, Hensher et al. 2015). We estimated a mixed logit model for the pooled dataset of all six catchments, and additionally included dummy variables for each catchment as interaction variables to analyse differences among them. We also estimated a model using respondent characteristics as interaction variables. Finally, we estimated separate models for each catchment to quantify marginal willingness-to-pay for each attribute as the negative of the attribute coefficient divided by the tax variable coefficient, as described in Hanemann (1982).

These analyses allowed for analysis of preference for different types of land use and land management across our six catchments, which served as further input for estimating the effects of bioeconomy scenarios on the value of active and passive nature appreciation.

4.3 Developing an ecosystem services framework

With the aim of making a quantitative, comparative estimate of ecosystem services value came the need for a consistent framework, applicable over all six catchments. Additionally, the framework needed to allow for scenario analysis. This meant that socio-geographic and landscape variables that might be altered by the bioeconomy needed to be directly linked to ecosystem services generation within the framework.

Before doing this however, clear boundaries of what to measure were necessary. The concept of ecosystem services lacks a clear definition, owing to the wide range of interpretations, methodological underpinnings and applications that the research community has assigned to it (see chapters 1 and 2). In preparation for the work on Paper II, we therefore started by considering the definition of an ecosystem service, keeping in mind the desired end point of the framework: a list of ecosystem services that can be quantified using the data we had at our disposal, that can be linked to socio-geographic and landscape characteristics of the catchments, as well as to direct beneficiaries of these services. The concept of final ecosystem services (FES), introduced by Boyd and Banzhaf (2007) and further expanded upon by Wallace (2007), fit our needs best. Recall from Chapter 2 that its definition is 'components of nature, directly enjoyed, consumed, or used to yield human well-being'. The key distinguishing feature of this definition compared to other definitions is the exclusion of indirect benefits. In contrast, some other definitions of ecosystem services: 'the benefits human populations derive, directly or indirectly, from ecosystem functions' (Costanza et al. 1997), 'the aspects of ecosystems utilised (actively or passively) to produce human well-being' (Fisher et al. 2009), 'the direct and indirect contributions of ecosystems to human well-being' (TEEB 2010). The focus of FES on direct enjoyment, consumption or use has implications on what to quantify. Quantification of a FES requires a direct link between an ecosystem and a beneficiary in society, which fits very well with my aim to estimate the effects of change on different groups in society. It also meant that double counting, an issue frequently discussed in the valuation literature (Bateman et al. 2011, Johnston and

Russell 2011, Keeler et al. 2012), would be minimised by providing a clear link between ecosystem process and benefit through direct interaction with this process.

Basing our framework on this definition, we created a list of FES that are generated in the six selected catchments. We then considered how to quantify their flows and the monetary value of these, using the information we had available, either through published research or publicly available statistics and GIS datasets (Table 4). This led to a framework structure informed by Boerema et al. (2014) and Mononen et al. (2016).

The decision to quantify using monetary valuation came from the need for comparative analysis. If the aim is to quantitatively compare the societal benefits generated by different Nordic catchments, and to compare the effects of bioeconomy scenarios on these benefits, a common indicator of value that can be applied to all benefits is necessary. Monetary valuation has proven to be an effective indicator for this, in part because of its strength as a communication tool (de Groot et al. 2012, Acuna et al. 2013). However, it is also a controversial method, with methodological issues related to the compilation of different valuation methods, from market pricing to stated preference valuation (Gómez-Baggethun et al. 2010, Bateman et al. 2011). In choosing to use this method for its comparative and communicative strengths, I acknowledged that it comes with uncertainty and the need for transparency in methodology. Table 4 shows how we used various valuation methods, depending on the type of ecosystem service, further explained in the following paragraph.

Table 4. List of selected final ecosystem services. This table shows for each ecosystem service who benefits, what we quantified and how we valued these quantified services. Table compiled from Papers II and III.

Final ecosystem service	Beneficiary	What to quantify	Valuation method
Supporting environment for crop production	Crop producers	Grains, grass and fodder and other crops produced	Producer prices with ecosystem contribution coefficients
Supporting environment for forestry	Foresters	Roundwood removed	Producer prices with ecosystem contribution coefficients
Availability of game	Hunters	Hunted game	Producer prices
Availability of peat	Peat extractors	Peat extracted	Producer prices with ecosystem contribution coefficients
Potential for hydropower generation	Electricity generators	Electricity generated	Producer prices
Availability of berries and mushrooms	Foragers	Berries and mushrooms gathered	Producer prices
Availability of water for drinking and processing	Water extractors	Water extracted	Producer prices
Active nature appreciation	Recreating visitors	Hunting and fishing licenses sold	License prices
		Days of inhabitant and visitor recreation	Travel cost
Passive nature appreciation[5]	Global society	Area of nature reserve	Willingness-to-pay for nature reserves
Mitigated climate change	Global society	Carbon sequestered in biomass and lake beds	Social cost of carbon
Prevented flood damage	Downstream property owners	Downstream area prevented from flooding	Land values and damage curves

4.4 Estimating the current value of ecosystem services

Estimating the current value of ecosystem services generated in the six catchments was the core work of Paper II. We started with an analysis of land use, using spatial data, combined with collecting statistics on the production and extraction of crops, wood products, wild plants and animals, peat, hydropower, and water. Additionally, we collected data on recreation by using the same survey data we analysed in Paper I, supplemented with statistics on the sale of licenses for

[5] Added in Paper III.

hunting and fishing, the annual growth of biomass to convert to quantities of carbon sequestration and spatial data on areas at risk of flooding. These data could then be converted to monetary value in € ha^{-1} y^{-1} using common methods in value estimation, depending on the type of ecosystem service. First are services that are inputs into production processes of goods that can be traded on markets. An example is the supporting environment for the production of crops. These crops have market prices, so we used the prices that their producers get for selling them as a basis. However, these producer prices also include the value of labour and man-made capital input used to produce these crops, so to separate the value of the ecosystem service's contribution we applied an ecosystem contribution coefficient to the producer price, based on Vallecillo et al. (2019). Then there are ecosystem services that in themselves generate goods or energy, without human input necessary for their production. Examples are game meat, berries, and mushrooms. For these we used producer prices, the monetary value that those extracting them from the ecosystem receive for their sale. For active nature appreciation, we used survey data collected for Paper I for a travel cost analysis, a well-established revealed preference method for value estimation (Haab and McConnell 2002), supplemented with the price of licenses sold for hunting and fishing. For passive nature appreciation, we used the DCE data from Paper I to estimate willingness-to-pay for an increase in nature reserves. For mitigated climate change through carbon sequestration, we used the social cost of carbon as a monetary value estimate (Tol 2005), and for the value of prevented flood damage we used the method described by de Moel and Aerts (2011), using land values and damage curves. Compiling these data into a common spreadsheet led to a complete list of annual flows of ecosystem services value for each catchment.

To further analyse what drives variation in value, we then used high resolution spatial data to distribute the value estimates over hectare cells in each catchment. We performed multiple linear regression using sub-catchments as observations, to see which spatially explicit socio-geographic and landscape variables correlate to the generation of value. Finally, we performed a basic analysis of the distribution of effects among different stakeholders, by altering land use and estimating the effects on value generated per stakeholder group.

4.5 Estimating the effects of a bioeconomy on ecosystem services

While Paper II focused on the current situation, in Paper III we looked at the potential effects of a future bioeconomy on ecosystem services generation. To do so, we needed three building blocks:

1. A baseline of ecosystem services value.
2. A set of quantified scenarios of what a bioeconomy can look like.
3. A framework that links these quantified scenarios to the generation of ecosystem services.

Building block 1 was provided to us by Paper II, which provided data on annual value generated in each catchment under the current situation.

We constructed building block 2 from Rakovic et al. (2020), as described in Chapter 2. The Nordic Bioeconomy Pathways (Table 5) provided qualitative narratives of five bioeconomy scenarios for the Nordic countries in 2050, built up of elements such as population growth, economic growth, bioeconomy policy orientation, energy use, crop production and forestry. We split these elements up into quantified sub-elements, for example, we split crop production into tonnes of crops produced, productivity per hectare and amount of phosphorus fertilisation per hectare. We based these quantifications on statistics and projections combined with expert judgement from colleagues at the Norwegian Institute for Bioeconomy Research, the Norwegian University of Life Sciences, Aarhus University, the Swedish University of Agricultural Sciences and the Natural Resources Institute Finland.

Table 5. Summary of the NBP storylines. This gives a short qualitative summary of each NBP storyline. Table adapted from Paper III.

NBP name	Summary of storyline
NBP1: Sustainability first	Development shifts to a more sustainable path, which respects perceived environmental boundaries and places human well-being ahead of economic growth. Lower and more efficient resource use, stronger reliance on renewables.
NBP2: Conventional first	Typical recent historical patterns with uneven development and income growth.
NBP3: Self-sufficiency first	The world is characterized by rising regional rivalry driven by growing nationalistic forces and the Nordic countries have become allies in a fragmented Europe. Nordic bioeconomy and self-sufficiency become matters of regional security.
NBP4: City first	Unequal investments in human development and rising differences in economic opportunity and political power, a gap widens across and within countries between a small affluent elite and underprivileged lower-income groups.
NBP5: Growth first	Spurred by high economic growth and rapid technological development, this society trusts that competitive markets, new technology and investments in human capital is the path to sustainable development.

Building block 3, a spreadsheet-based framework that links the previous two together, worked by connecting the quantified sub-elements of the NBPs to attributes of the catchment, such as the size of cropland area, built-up area and nature reserves (Figure 3). Since these catchment attributes directly impacted ecosystem services generation, we could quantify for each NBP what the value of each ecosystem service in each catchment would be, allowing for comparison with the current situation. Next, we made the effects spatially explicit by creating a set of knowledge rules that defines where land use would change within logical boundaries. For instance, forest will only become agriculture where the soil is suitable, and built area will expand from those areas that are currently already built up. Using this spatially explicit set of bioeconomy scenarios for each catchment, we could then interpret the effects within catchments, among catchments and among various stakeholder groups.

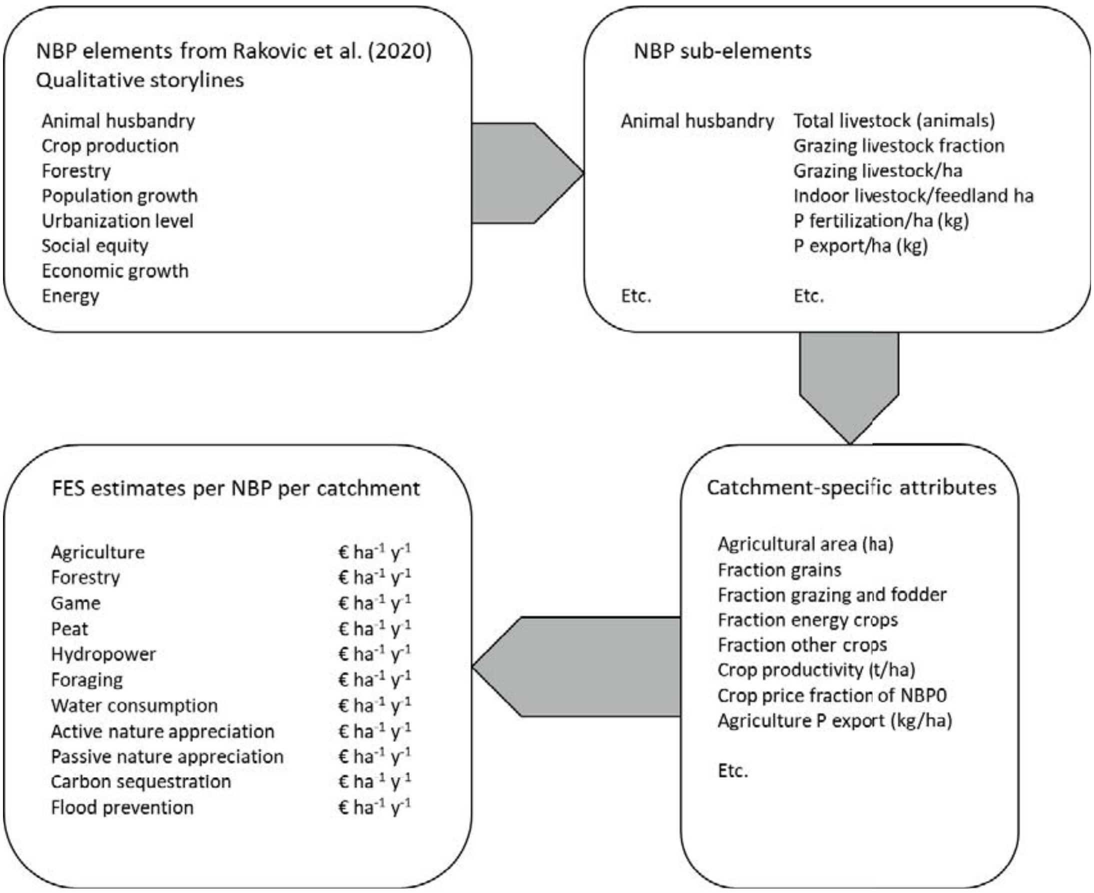


Figure 3. Flowchart showing how the NBP elements were translated to FES value estimates. This figure shows how the NBP elements from Rakovic et al. (2020) served as inputs for a set of quantitative variables, the NBP sub-elements. These were then translated into physical attributes in each catchment, which are transformations of the current values that were used for NBP0 in Paper II. These catchment attributes are directly linked to FES value estimates, generating a unique set of estimates for each NBP. The full spreadsheets are available as Supplement 1 to Paper III (available on request from the first author).

5 Main findings

5.1 Paper I - Appreciation of Nordic landscapes

An MXL model of the pooled dataset for all six catchments showed significant ($p<0.01$) coefficients for preference for all variables in the model. Respondents positively favoured an increase in agricultural area in the balance between agriculture and forest. They showed negative preference for both more intensive and more extensive land management, as well as for an increase in flood frequency. Increase in water clarity, the area used for nature reserves and local employment from agriculture, forestry and fishery were all preferred.

When differentiating among catchments, we found stronger positive preference for having more agriculture in Haldenvassdraget and Vindelälven, which are both rural, forested catchments, and stronger preference for having more forest in Sävjaån, which is a peri-urban, agricultural catchment. In Haldenvassdraget we also found a stronger negative preference for more small scale, extensive management, as well as for an increase in nature reserves. In both Swedish catchments preference for improved water clarity was stronger than in the other catchments. In Odense, in which the main stream runs through a city, there was a stronger negative preference for increased flooding, while in Sävjaån we found a less strong negative preference for the same.

Finally, we estimated similar MXL models but using socio-demographic characteristics as interaction variables instead of dummy variables for each catchment. Here we found that non-local visitors have stronger preference for change in land management intensity as well as for water clarity improvements. People that have a stronger concern for environmental issues, rated on the NEP-scale (Dunlap and Vanliere 1978, Dunlap et al. 2000), showed stronger preference for improving water clarity and an increase in the area used for nature reserves. Age had a positive effect on preference for agriculture over forest, and a negative effect on preference for increased water clarity. Respondents with higher education showed the opposite: they had stronger preference for forest over agriculture, as well as for increased water clarity. Income only appeared to have an effect on a stronger negative preference for more intensive land management, while those employed in agriculture, forestry and fishery appeared to have lower preference for increased water clarity.

5.2 Paper II - Estimating societal benefits from Nordic catchments

Monetary estimates of FES value showed variation among the six catchments (Figure 4). Total value generated annually was highest in Odense (around €125 million year^{-1}) and lowest in Simojoki (around €20 million year^{-1}). However, when normalising value over area, a different picture arises. Orrevassdraget, a small agricultural catchment on the west coast of Norway, generates by far the highest value, at over €7,000 ha^{-1} year^{-1}, with the other catchments varying between about €400 and €1,100. When normalising over inhabitants, yet another picture appears, where Simojoki, the least densely populated catchment, generates the highest value, with around €14,000 inhabitant^{-1} year^{-1}, compared to a lowest value of around €500 inhabitant^{-1} year^{-1} in Odense. A large part of the total FES value estimate, especially in Orrevassdraget, comes from recreational activities, which explains why densely populated catchments like Odense and Orrevassdraget generate the largest flow of value. When comparing among catchments, agriculture and water for drinking and industrial use are most substantial in Odense, while forestry and carbon sequestration generate most value in Sävjaån. Peat that can be extracted is only available in Simojoki, where it generates about a third of the catchment's annual value.

Spatial analysis showed that value was mostly generated in the main river valleys, where agriculture and recreation concentrated, as well as near more densely populated, most clearly visible in the concentration of value around the city of Odense (Figure 5). More remote agricultural areas and forest generated least value. When considering the spatial distribution of value for separate stakeholder groups, we found that large extractors (water companies, peat extractors, energy companies) dominate in built-up areas and peatland areas under production, landowners dominate in croplands and production forests far from inhabited areas, recreating visitors dominate in more densely populated areas that are well connected and are close to water, and global society dominates in remote forest and nature, where carbon sequestration is the dominant ecosystem service.

Statistical analysis using multiple linear regression showed that several socio-geographic and landscape attributes correlated significantly with monetary value of FES: the availability of clay soils correlated positively with agricultural value, while topographic slope correlated negatively with it, as well as with recreational value. Landscape diversity, measured using the Shannon Diversity Index, appeared to correlate negatively with agricultural value and recreational value, and positively with forestry value. Population density showed positive correlations with agricultural value and recreational value, and a negative correlation with forestry value. Finally, the fraction of surface water of total area positively correlated with recreation and forestry, and negatively with agriculture.

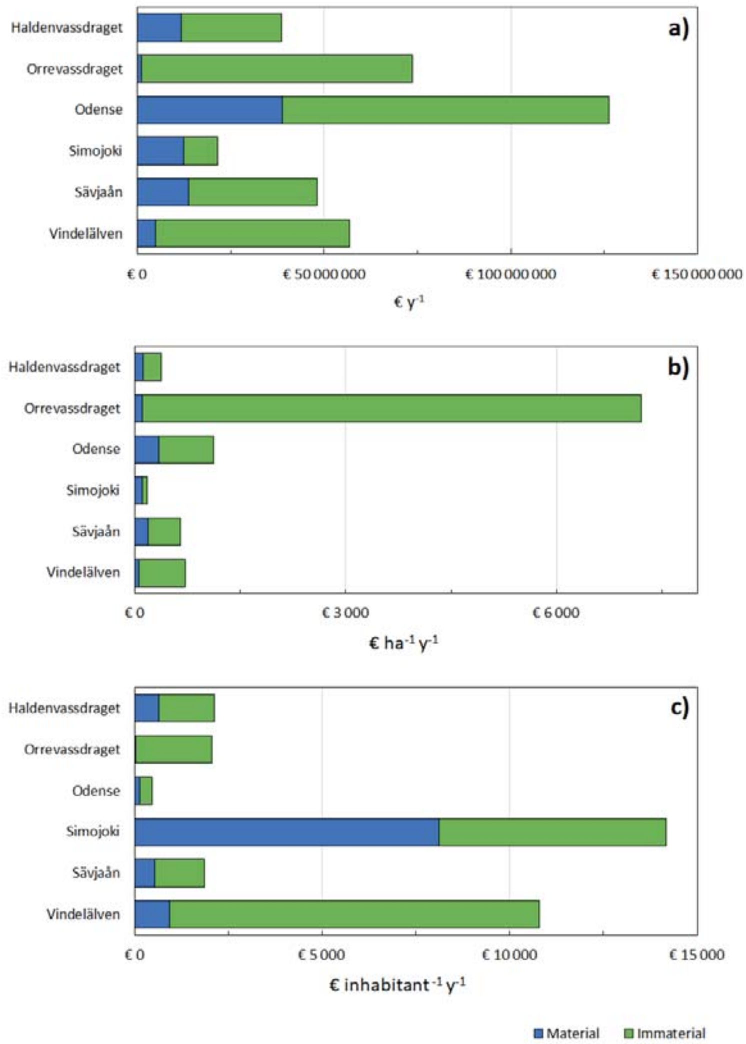


Figure 4. Total economic value per study site, split out over material and immaterial ecosystem services. a: The sum of all value consumed from ecosystem services per year in each study site. b: The same values, only divided by study site area in hectares. c: The same values, only divided by study site population. Figure from Paper II.

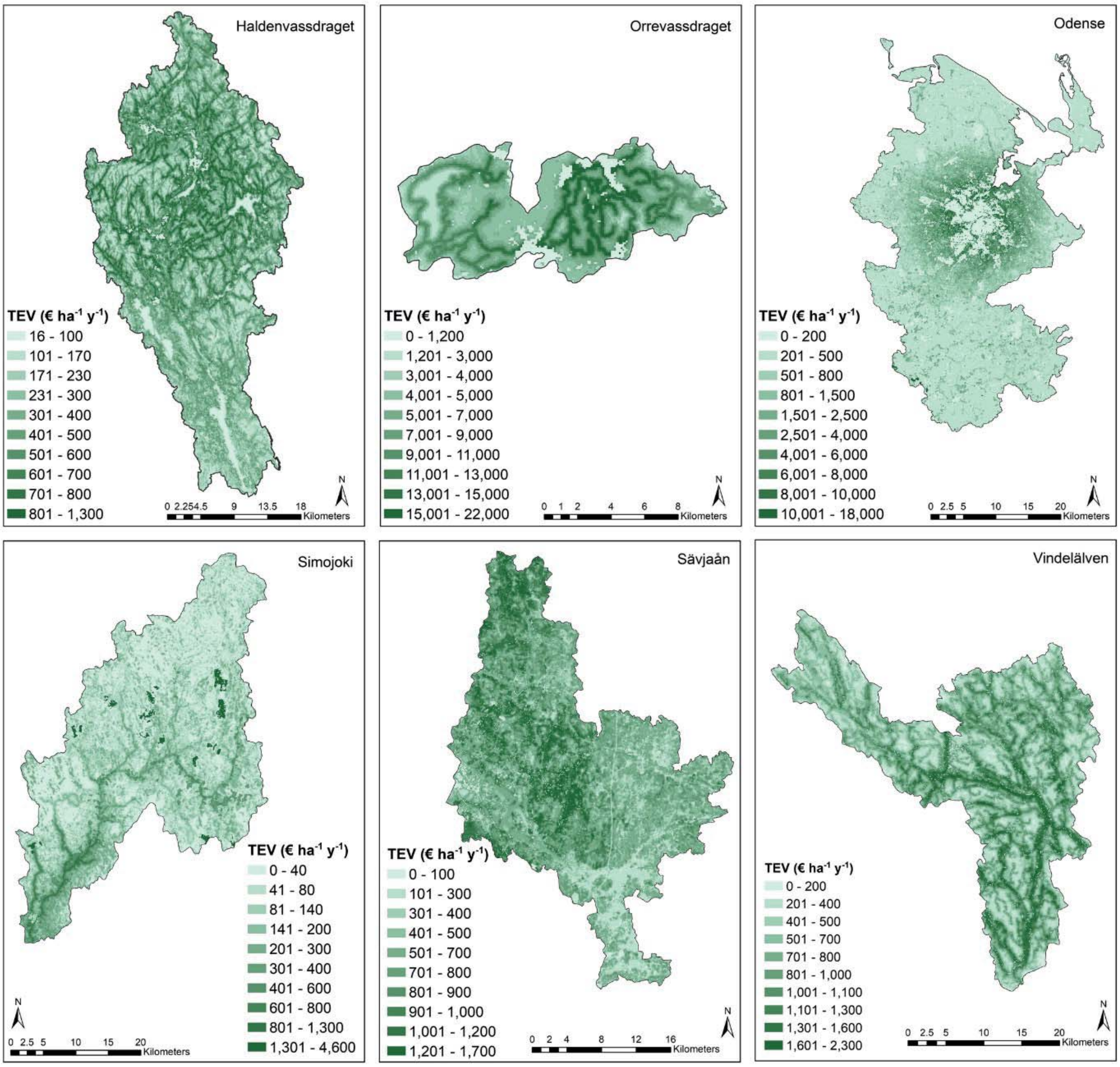


Figure 5. Total economic value estimates per hectare per year for each study area. Note the different colour scales. This reduces comparability among study areas, but increases the resolution of values shown within each study area. Figure from Paper II.

5.3 Paper III - The value of change

Using a framework that links FES value estimates to the NBPs showed that value generation varies within catchments among NBPs, but the effects of the NBPs also vary among the catchments (Figure 6). In general, NBP1 (Sustainability First) and NBP5 (Growth First) generated the highest total value. Simojoki is an exception because of its current reliance on peat extraction, which ends under NBP1. NBP4 (City First) showed greatest variation in effects among catchments: rural, forested catchments all do worse than currently under this scenario, while all peri-urban, agricultural catchments do better than currently.

The distribution of value over the separate FES also varies among NBPs. Under NBP1 for example, the relative value of ecosystem services used in the production of goods, such as from agriculture and forestry, declines, while active and passive nature appreciation gain a larger share of the value. Changes in relative value also illustrate a divide between rural and peri-urban catchments: under NBP4, in rural catchments produced goods gain in relative value and active nature appreciation loses in relative value, while in peri-urban catchments the opposite happens.

Different groups in society also benefit differently from the NBPs: landowners benefit most under NBP5 (Growth First), large extractors benefit most under NBP4 (City First), and visitors and global society benefit most from NBP1 (Sustainability First).

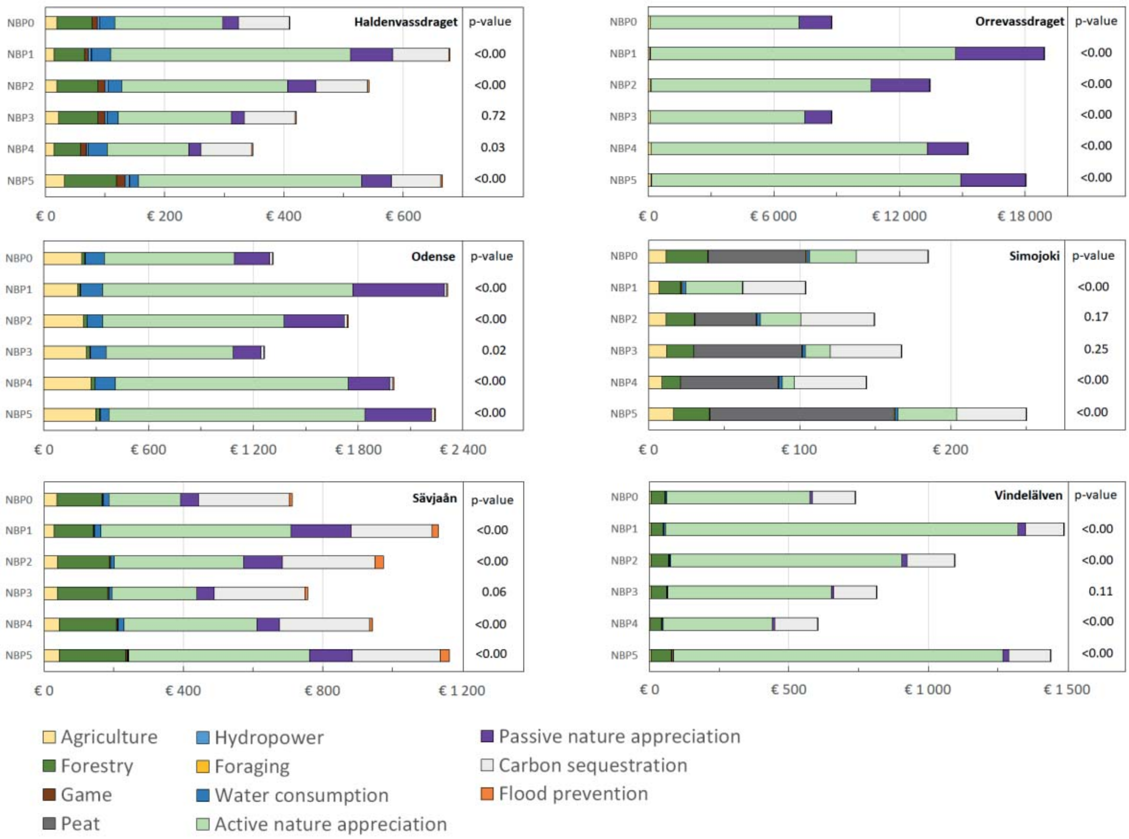


Figure 6. Economic value of groups of ecosystem services generated in our study areas for each NBP, in € ha^{-1} year^{-1}. This shows per study area the economic value of all estimated ecosystem services. Next to each bar we give a p-value for the chi-square test statistic, indicating whether there is a statistically significant difference in distribution over the different services compared to NBP0. With an appropriate Bonferroni correction, these comparisons are significant when p< 0.01. Figure from Paper III.

6 Discussion

6.1 Answering the research questions

Can we apply the concept of ecosystem services to successfully estimate the effects of a transition to a bioeconomy on a Nordic scale?

In this thesis, I attempted to show how society benefits from the ecosystem services generated by Nordic catchments using a framework that generates monetary value estimates. I specifically designed this framework to allow for incorporation of the effects of a green shift, by linking socio-geographic and landscape variables to the value estimates. To assess how successful the estimates of the effects of a bioeconomy are, three questions need to be answered.

Firstly, how reliable are the estimates of current value? The first concern is the reliability of the baseline: the value being generated under the current situation. In Paper II, we attempted to answer this question by testing the framework's quality, as well as by comparing the estimates to findings from studies with similar methods in similar geographic regions. We used criteria defined by Boerema et al. (2017) to test our framework, and found that it is suitable for ecosystem services quantification, because it is explicit in what is quantified, uses clear definitions of final ecosystem services, differentiates between supply and demand of ecosystem services by explicitly incorporating beneficiaries of the services, and uses traceable methods and data sources. We then tested the reliability of the data sources, by comparing our data to similar valuation studies in similar regions, and found that our estimates typically fall within the same value ranges, regardless of where we applied the framework.

Then, how plausible are the bioeconomy scenarios applied? We based our scenarios on the NBP storylines, as presented in Rakovic et al. (2020). These are in turn based on the Shared

Socioeconomic Pathways (O'Neill et al. 2014), which are well-established scenarios for future socio-economic change (Popp et al. 2017, Riahi et al. 2017). The NBPs are scaled down applications of the SSPs, tailored to the Nordics and focused on bioeconomy development. Their storylines were designed using expert judgment from a group of thirty researchers specialised in land, water and ecological management across the Nordic countries. From these storylines, we then quantified likely effects on six Nordic catchments in a similar manner (Paper III): we quantified various NBP sub-elements that would impact ecosystem services values in our six catchments, based on a combination of trend projections and boundary conditions from published research and statistics reports, as well as consultation workshops with a subset of the same experts that were involved with the design of the NBP storylines. It is inherent of future scenarios that their plausibility can never be formally tested (Berkhout et al. 2002), but this combination of a basis in established research, tailored to a specific need using expert consultation on both the level from SSP to NBP, and the level from NBP to catchments, at least forms a basis in understanding of the current state and processes that is internally consistent, transparent and traceable.

Finally, how realistic are the effects of these scenarios on ecosystem services value? This depends on three things: the realism of the estimates of the current state, the realism of the scenarios, and the realism of the interactions between the current state and the changes defined by the scenarios. The first two are covered in the previous paragraphs. The final point depends on the design of the estimation framework. As we described in Paper III, our framework uses a dataset based on statistics, monitoring data and published research, and does not include dynamic catchment modelling. The main advantage of this method is that the values are based on traceable, empirical measurements with minimal underlying assumptions, but a disadvantage is that we cannot include complex dynamic processes. Time effects and interaction effects between the elements within the framework can at best be rudimentary incorporations using knowledge rules in spreadsheets. This has implications for scenario effects, since in reality, changing land management will likely alter dynamic processes within the ecosystem and hydrological cycle. This means that the estimates in this thesis come with uncertainty, the ranges of which cannot be estimated. The value of the estimated scenario effects is then not in the precise values, but in the relative differences and the larger picture they present: which ecosystem services are likely to become more prominent in certain scenarios, which type of catchment will likely change most, which stakeholder group will benefit most, and will this come at a cost for other stakeholders? I argue that these questions are more relevant than precise values when considering how to direct the green shift, and that our basis in transparently traceable empirical data, combined with multiple rounds of stakeholder assessment on multiple levels of scenario building, justify our use of an ecosystem services framework to successfully estimate the effects of a transition to a bioeconomy on an international scale.

If so, what are potential effects of such a transition on the societal value of generated ecosystem services from Nordic catchments?

When considering the effects of all NBPs in all studied catchments, the results indicate that the green shift will likely lead to an increase in societal value generated by our interactions with ecosystems in Nordic catchments. How large these benefits are and how they are distributed over society depend on the shape of the green shift.

NBP1 would yield the greatest net benefits when summing over all six catchments. This is a scenario in which society increasingly recognises the environmental, social and economic costs of current production and consumption patterns, and chooses to shift to a more sustainable path that respects environmental boundaries and places human well-being over economic growth. However, even under the scenario that is likely to produce the largest net gains, some regions will not benefit: our estimates of change in value range between 57% of current value under NBP1 in Simojoki, to 216% of current value under NBP1 in Orrevassdraget. For Simojoki that means a decrease of about €9 million year^{-1}, while in Orrevassdraget that means an increase of about €100 million year^{-1}. This not only illustrates the differences in effects between different types of catchments, but also the scale of the effects of implementation of a bioeconomy. An increase in net benefits is not the only effect of this scenario though: in all six catchments, it would also produce a statistically significant rearrangement of distribution of value over separate ecosystem services, mainly due to an increase in value from active nature appreciation such as recreational activities, and a decrease in value from what are typically defined as provisioning services: benefits from agriculture, forestry and peat extraction. This means different stakeholder groups might see significantly different effects, and indeed both landowners and large resource extractors will see a net reduction in benefits under NBP1. This suggests that simply following total economic value as a guideline for policy making, though possibly optimising net societal benefit, can have severe negative effects for specific groups in society. Complicating matters further, the effects on both total benefit and on distribution of benefits over society will vary according to catchment type, as well as within catchments, which can also have implications for effective policy decisions on a green shift, which I will discuss further in the next paragraph.

If the bioeconomy takes another shape, the effects can be significantly different. NBP4 is a scenario in which differences in economic opportunity and political power increase, leading to widening gaps between urban and rural areas, between those with high incomes and those with low incomes, and between progressives and conservatives. If the bioeconomy is shaped around these trends, the estimates from this thesis suggest that the results will be profoundly different from those under NBP1. All the rural catchments we studied will see net decreases in ecosystem services value, while all the peri-urban catchments will see net increases. In all catchments except Haldenvassdraget, the distribution of value over the different services will also significantly change, but under this scenario the change will depend on the type of catchment: rural catchments will become more dependent on peat extraction and carbon sequestration, while the benefits from active nature appreciation, forestry and agriculture will decrease. This means that even within rural areas, the benefits that are left over will increasingly flow to those not living there, while inhabitants, depending on forestry, agriculture and recreation, will do disproportionally worse. In the peri-urban catchments in the meantime, net benefits will increase, and this is mainly generated by an increase in value of active nature appreciation, both because inhabitants will have better opportunity to recreate in nature, and because these areas will attract more visitors than before. This development is in line with the expected trends in the NBPs (Rakovic et al. 2020), as well as in the SSPs that they are based on (Jiang and O'Neill 2017).

I describe here only two of the five NBPs in some detail, because this is enough to illustrate that the shape of the bioeconomy will have a large effect on societal value of ecosystem services generated in Nordic catchments. Overall, net benefits are likely to increase compared to the current situation, but if, how and where this will indeed become reality will depend on the choices that are made to shape the green shift, as well as possible lock-in effects (Klitkou et al. 2015, Scarlat et al. 2015).

6.2 Policy implications

The results presented in this thesis suggest that NBP1 will produce the largest net benefit from ecosystem services generated in Nordic catchments. Should we then strive to follow this pathway to a bioeconomy? That depends on the goals. If we do want to aim for net benefits, then aiming for something like NBP1 is advisable. NBP5 at the same time is expected to deliver similar if slightly lower benefits, but its distribution of value over different groups in society is more equal: where in NBP1 landowners and large extractors receive reduced benefits compared to current, under NBP5 they too, along with the other stakeholder groups, would benefit or at least would not see a large decrease in benefit. What NBP1 and NBP5 have in common however, is a consideration of local environmental quality. Even if society under NBP5 is resource intensive, it also makes efforts to mitigate environmental impacts of nutrient runoff and biodiversity. The fact that the two scenarios with highest net benefits both include ambitious attempts at local environmental protection suggests this should be a focus of local land management if net benefits are the primary objective.

Optimising for equity of distribution might also be a desired aim, since the estimation results suggest that under all NBPs, differences in distribution among stakeholder groups will increase, potentially increasing the risk of conflicts for land use and management. NBP3 provides the most equal distribution among stakeholder groups, but it also delivers the lowest net benefit. This is, however, a trade-off that can only be partially handled by land management decisions: much of the NBP effects stem from global trends trickling down into these catchments (Rakovic et al. 2020). Another consideration for land management decisions is the spatial distribution of benefits within catchments. As the spatial analysis in Paper III indicates, different bioeconomy pathways can result in different redistribution of benefits over space.

Overall, the findings in this thesis suggest that, depending on what type of policy is prioritised, effects of a green shift will vary among different types of catchments across the Nordics as well as among those groups in society that benefit from their interactions with ecosystems. Increased benefits are likely under a developed bioeconomy, but these will likely also come with increased distributional effects and potential conflict among stakeholders, something found in previous studies as well (Meyer 2017, Priefer et al. 2017, Hafner et al. 2020).

6.3 Contribution to the field

This thesis aims to contribute to the field of ecosystem services in three ways:

1. By the creation of a new framework for ecosystem services estimation that follows an internally consistent definition of ecosystem services, allows for monetary valuation and spatial analysis, and is flexible enough to be applied in an international, comparative context.
2. By the application of this framework across four Nordic countries, allowing for a comparative analysis without the restrictions of comparing varying methodologies.
3. By providing quantitative estimates of the relationship between a green shift and the value of ecosystem services from Nordic catchments.

As chapter 2 illustrates, conceptual frameworks of ecosystem services quantifications abound, as do their practical applications. What is the value then of yet another framework and another application of it? Debate on what an ecosystem service is and how, or even if, it should be quantified have been ongoing for as long as the field exists (Boerema et al. 2017, Costanza et al. 2017). I do not aim for this work to conclude these debates. What I do think my work shows, is that it is possible to create a framework based on the rigorous definitions of final ecosystem services (Boyd and Banzhaf 2007), that allows for the monetary valuation of *all* relevant ecosystem services in multiple geographic entities using publicly available data in a consistent manner, thus allowing for comparative analysis. The concept of final ecosystem services has been described extensively and has been applied to selected ecosystem services (Saarikoski et al. 2015, O'Dea et al. 2017, Lai et al. 2018), but to my knowledge never on the scope of total economic value for multiple study areas. Doing so allows for further discussion on effective ways of measuring our dependency on nature, on both a conceptual and a methodological level.

The overview in chapter 2 showed that ecosystem services provision in the Nordic countries has already been extensively studied, but it also showed that there is still a significant gap in valuation studies covering a complete set of ecosystem services. This thesis adds to the body of knowledge by including an international, comparative analysis of not only the value of such a set of ecosystem services, but also on what causes variability. By using a consistently applied framework across the Nordics, it shows how various attributes of the landscape, and of the groups of society interacting with it, affect these ecosystem service values. By comparing different types of catchments, from rural to peri-urban, in four different countries, this thesis aims to provide deeper understanding of what causes value generated by our interactions with Nordic catchments. This provides grounds for further discussion and refinement of the applied methods, as well as for broader application.

Finally, this thesis aims to provide more understanding of the relationship between the green shift and the generation of ecosystem services. Dietz et al. (2018) performed a literature analysis of 45 studies that link ecosystem services to bioeconomy, and though they found an increasing number studies linking the two concepts, some of the main findings were that papers 'express the need for further and more sophisticated assessments of changes in land use and ecosystem services', and that very few studies draw equally from both concepts. By performing a quantified assessment of the effects of land use and socio-geographic change on the societal value of a complete set of ecosystem services, this thesis has attempted to fill part of this gap.

6.4 Future outlook

Since the green shift to a bioeconomy is a policy priority for the Nordic countries (Gíslason and Bragadóttir 2017), the shape of this transition can be directed by those making decisions on land management and socio-geographic policy. This thesis gives a quantitative assessment of the effects of such decisions on the value of ecosystem services generated in Nordic catchments. It shows that the shape of the bioeconomy will significantly affect the total value of these services, as well as where they are generated and who benefits. However, much about these complex relationships is still unknown, so further research should strive to create better understanding. Some key knowledge gaps that deserve further study are:

1. How are the dynamic processes within catchments affected by bioeconomy-induced land management change?
2. How will the green shift take shape over time, and how do time lag effects impact ecosystem services generation?
3. How will the green shift impact other Nordic catchments?
4. How will climate change affect the bioeconomy and its consequences for ecosystem services generation?

These questions are hard to answer. Projections of effects into the future inherently come with uncertainties, especially when considering the interaction between complex, dynamic human societies and complex, dynamic ecosystems. We cannot predict the future with certainty, but attempting to answer these questions can help societies prepare for the green shift and how it might change their relationship to the natural world.

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Errata

Page	Line	Change from	Change to
4	12	He concludes that instead of focusing on quantifying stocks of resources, we should instead aim…	He concludes that instead of focusing on quantifying stocks of resources, we should aim…
10	Table 1	Separating lines have incorrect thickness	Corrected
27	Table 3	Separating lines have incorrect thickness	Corrected

Paper I Immerzeel, B., Vermaat, J.E., Juutinen, A., Pouta, E. & Artell, J. Appreciation of Nordic landscapes and how the bioeconomy might change that: results from a discrete choice experiment. – Land Use Policy. (Submitted)
Paper II Immerzeel, B., Vermaat, J.E., Riise, G., Juutinen, A. & Futter, M. 2021. Estimating societal benefits from Nordic catchments: An integrative approach using a final ecosystem services framework. – PLoS ONE 16(6): e0252352, 24 pp. DOI: 10.1371/journal.pone.0252352
Paper III Immerzeel, B., Vermaat, J.E., Collentine, D., Juutinen, A., Kronvang, B., Skarbøvik, E. & Vodder Carstensen, M. The value of change: a scenario assessment of the effects of bioeconomy driven land use change on ecosystem service provision. (Manuscript)

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