Norwegian University of Life Sciences
Faculty of Environmental Sciences
and Natural Resource Management

Philosophiae Doctor (PhD)
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# How green is the green shift? The potential effects of a bioeconomy on ecosystem services in Nordic catchments

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Bart Immerzeel

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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.

At the centre of it all, along with me, my three supervisors. Artti Juutinen, who joined as supervisor after I realised how invaluable his knowledge on environmental economics and survey data was for my work. I will never forget the hospitality with which he welcomed me into his home in Oulu and showed me that some of the best pancakes in the world are found there. Gunnhild Riise, who always brought kind words of support or a useful insight when she came knocking at my office door. Jan Vermaat, my main supervisor and partner in this adventure. His invitation to Norway and to this work changed my life in countless good ways. Generous with dinners, board game nights, advice and support from the first day I set foot in this country. Always there when I needed him, even if he was busier than ever, and always trusting enough to let me figure things out for a few weeks when I needed it. A perfect mentor for a person like me, who taught me more about ecosystem services and research in general than anyone else.

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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 20$^{th}$ 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<br>Split services from benefits |
| **NEA** | 2011 | Government of the United Kingdom | National application<br>Spatial analysis |
| **SEEA** | 2012 | United Nations Statistical Commission | Natural capital accounting |
| **CICES** | 2013 | European Environment Agency | Hierarchical structure<br>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<br>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).

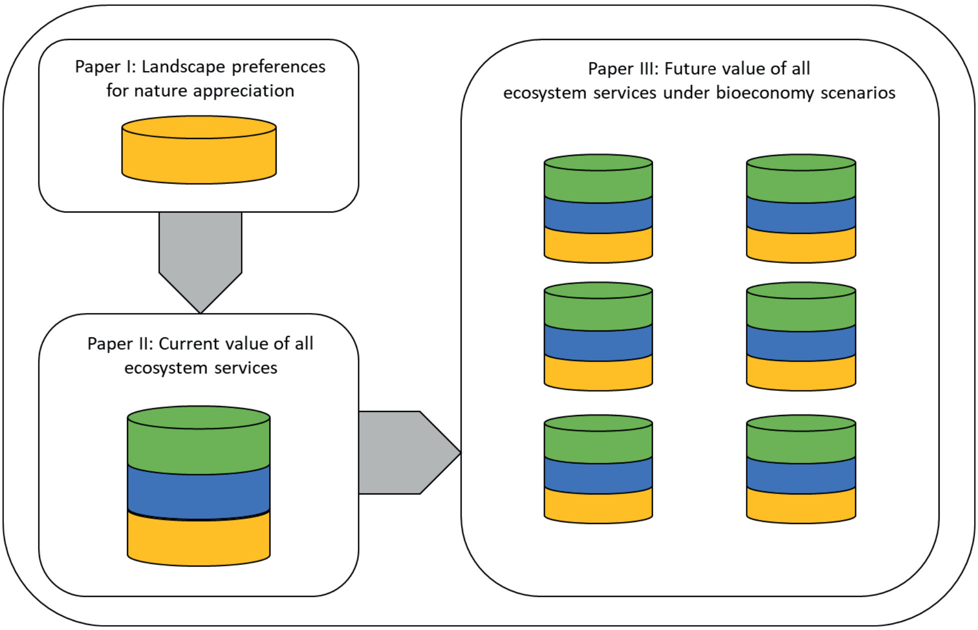


**Figure 1. The thesis structure.** This figure shows how the three papers at the core of this thesis connect to answer the main research questions. Each coloured cylinder represents a category of ecosystem services. Each stack of cylinders represents a complete set of relevant ecosystem services.

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.

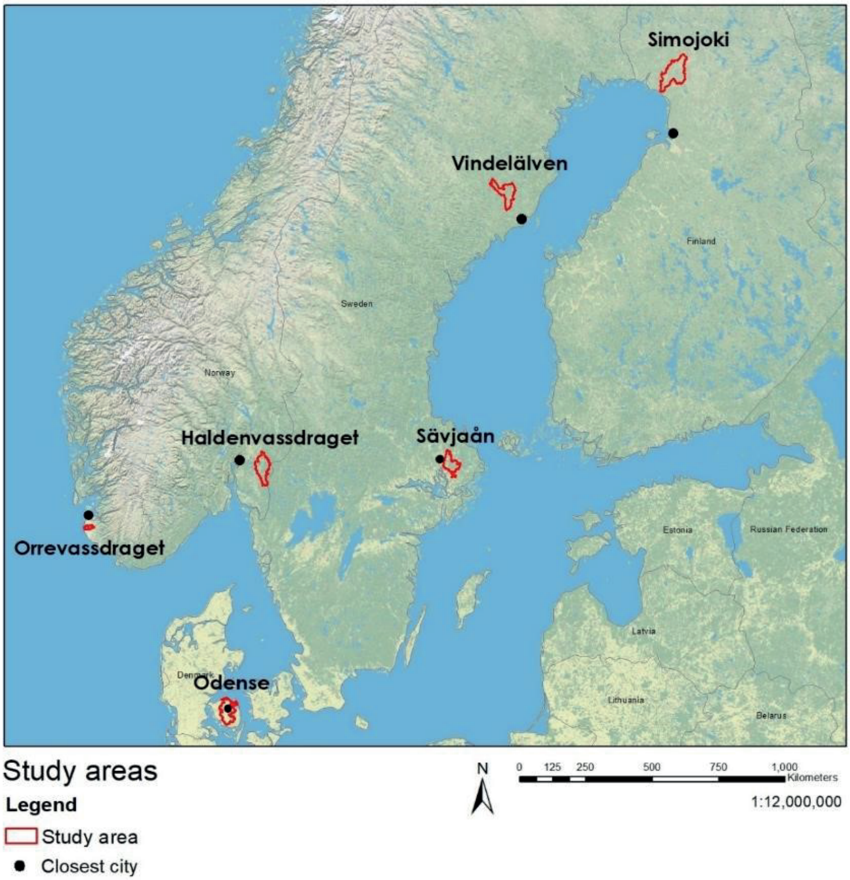


**Figure 2. A map showing the positions of the different catchments across the Nordic countries.** The basemap is provided by ESRI[4]. Study site boundaries are shown in red. Black dots show the city closest to the sub-catchment as described in Table 2. This map illustrates the spatial range of study 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.

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[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.

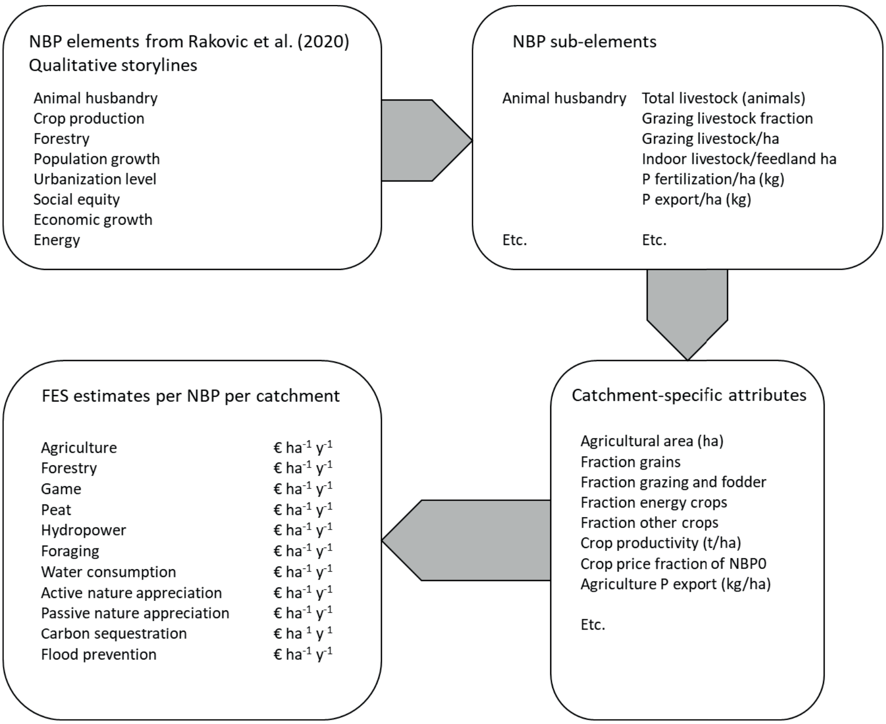


**Figure 3. Flowchart showing how the NBP elements were translated to FES value estimates.** This figure shows how the NBP elements from Rakovic et al. (2020) served as inputs for a set of quantitative variables, the NBP sub-elements. These were then translated into physical attributes in each catchment, 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.

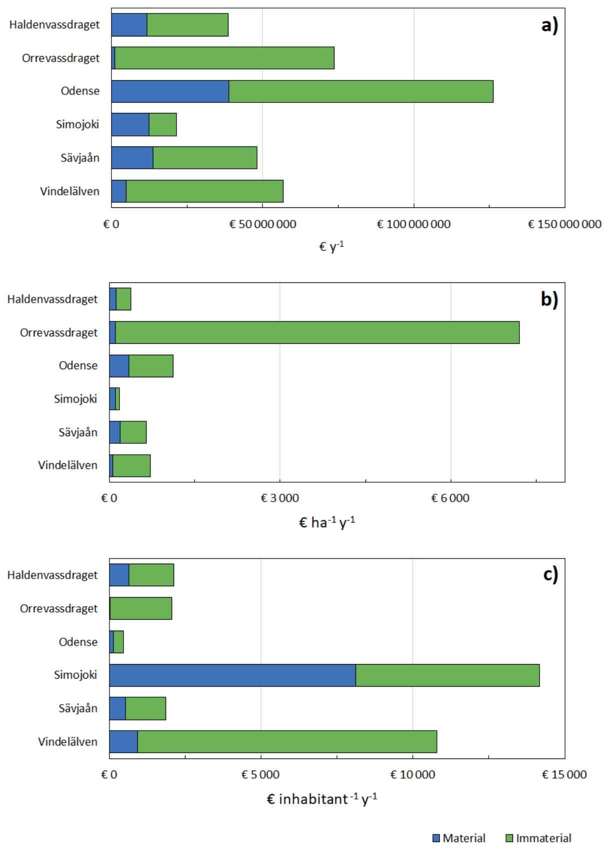


**Figure 4. Total economic value per study site, split out over material and immaterial ecosystem services.** a: The sum of all value consumed from ecosystem services per year in each study site. b: The same values, only divided by study site area in hectares. c: The same values, only divided by study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

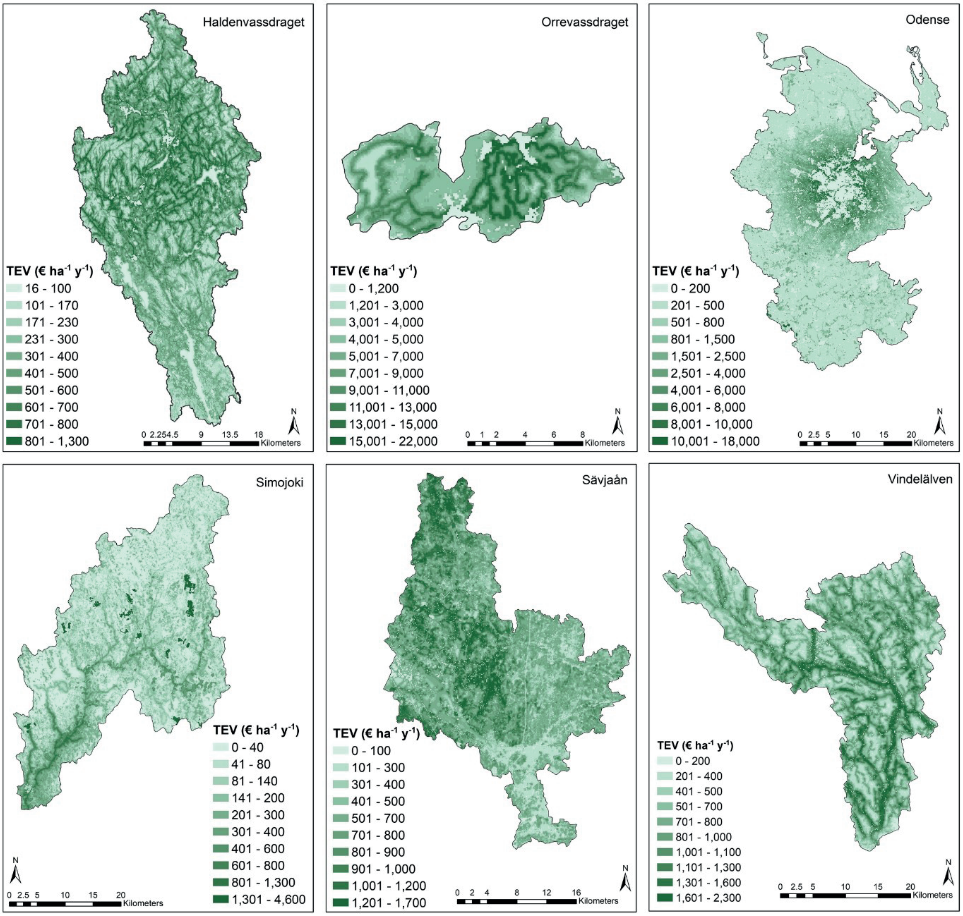


**Figure 5. Total economic value estimates per hectare per year for each study area.** Note the different colour scales. This reduces comparability among study areas, but increases the resolution of values shown within each study area. Figure from Paper II.

## 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).

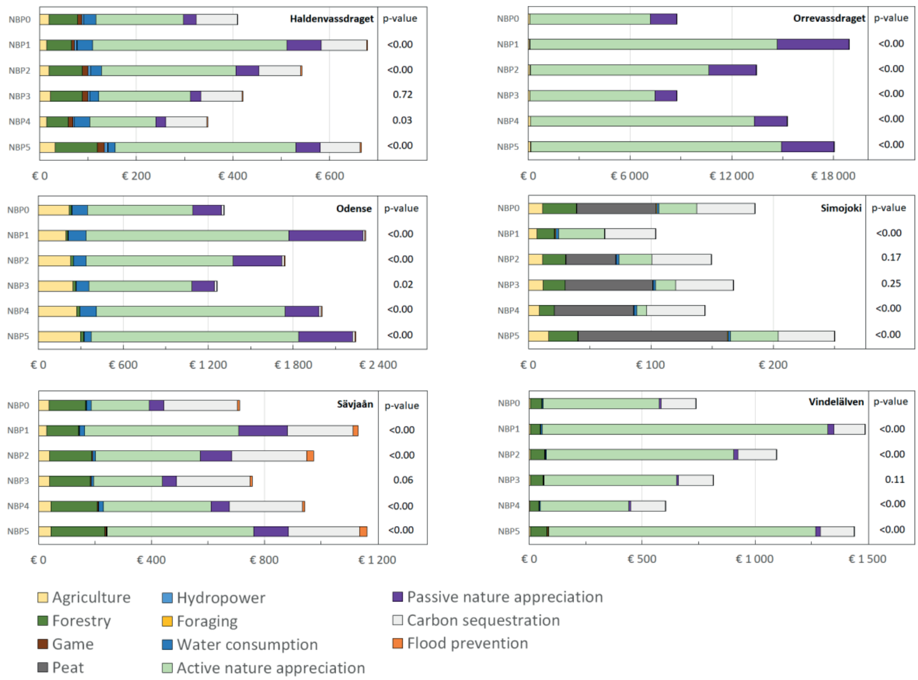


**Figure 6. Economic value of groups of ecosystem services generated in our study areas for each NBP, in € ha$^{-1}$ year$^{-1}$.** This shows per study area the economic value of all estimated ecosystem services. Next to each bar we give a p-value for the chi-square test statistic, indicating whether 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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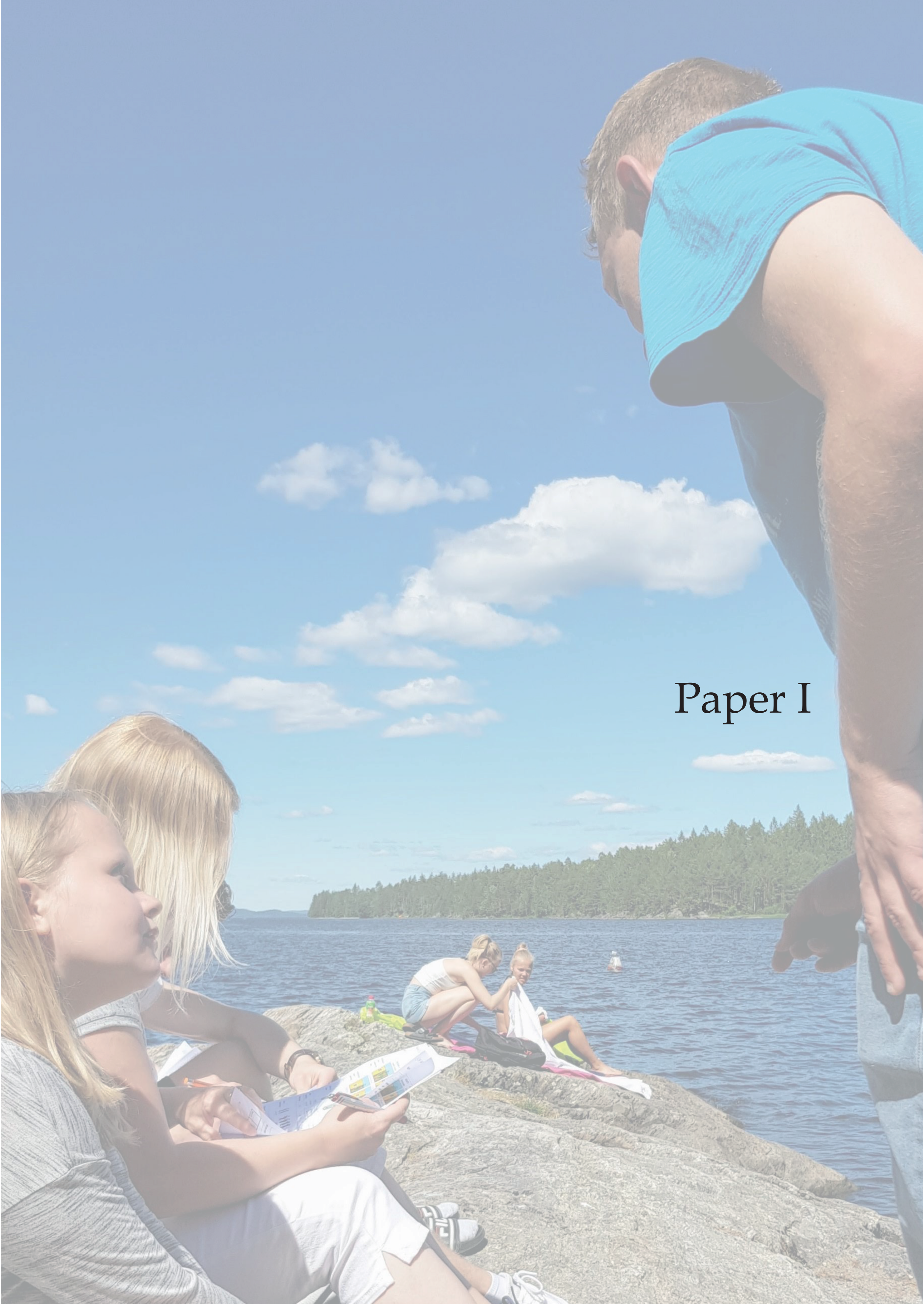
# Errata

| 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

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

Bart Immerzeel, Jan Vermaat, Artti Juutinen, Eija Pouta, Janne Artell

**Abstract**

Surface waters and their catchments provide societal benefits through cultural ecosystem services like recreation and appreciation of nature. The supply of cultural services depends on landscape characteristics like the extent of forested area, water clarity and the intensity of land use. These attributes vary spatially and will likely be influenced by a possible transition to a bioeconomy, i.e. a shift towards more use of renewable, biological resources like forestry products. Using a discrete choice experiment, we quantified survey respondents' preferences and willingness to pay for changing landscape attributes in six Nordic catchments and explored how different characteristics of both the landscape and respondents affect these preferences. Results from a mixed logit (MXL) model analysis show preference for a more equal distribution of agriculture and forest, improved water clarity, increased area used for nature reserves, reduced flood frequency and increased employment from agriculture, forestry and fishery. Variation in preferences between study areas is significant in several of these attributes, and likely linked to respondent characteristics. Since these attributes can be affected by the transition to a bioeconomy, policy makers should take into account the effects of this transition on the supply of cultural services by considering the effects on welfare generated by cultural services when implementing land management policy.

Keywords: Bioeconomy; cultural ecosystem services; catchments; discrete choice experiment; willingness-to-pay.

# 1. Introduction

The ecosystem services framework views ecosystems from an anthropocentric perspective, in which the key variables are the quantified benefits that society derives from the existence of ecosystems. It gives insight into benefits that may not be easily recognized by policy makers and the public, but which can be substantial (MA 2005, Fisher et al. 2009, Grizzetti et al. 2016), making the framework of increasing interest for policy makers (Belling 2017). Cultural services are a subset of ecosystem services that are derived from experiential and intellectual activities through interaction with ecosystems (Daniel et al. 2012, Haines-Young and Potschin 2017). Examples are the possibility to recreate in a lake or river, the enjoyment of being in a natural scene or the knowledge that a certain species exists somewhere in the world. Cultural services are widely recognized as providing a significant contribution of total ecosystem services value (Brander et al. 2006, Daniel et al. 2012). They are also possibly the services that are most widely recognized by the general public (Larson et al. 2016), making them especially important for policy making with an interest in public perception.

Surface waters and their catchments supply a variety of cultural services (Barton et al. 2012, Richnau et al. 2013), from direct use of the water like boating, swimming and fishing, to enjoying the aesthetics of the total landscape. The value of cultural services supplied by catchments depends on a number of factors (Garcia-Martin et al. 2017) which can be grouped into two kinds: attributes of the landscape supplying the potential services, and preferences of the individuals benefitting from these services (Halkos and Matsiori 2014). Analysing the relationship between cultural services and these attributes is complex, and intercorrelations between the different attributes likely exist (Garcia-Martin et al. 2017). Further complicating the matter, this relationship is not one-directional: catchments are subject to pressures from both societal (Lepistö et al. 2014) and climate change (Øygarden et al. 2014), all potentially affecting the supply of cultural services.

One possible change that can affect these attributes within the coming decades is the transition from our current fossil fuel based society to a bioeconomy (Hetemäki and Muys 2017). This transition may involve a range of societal, economic and land use changes, and can play a major role in addressing climate change, food security, health, industrial restructuring and energy security (Issa et al. 2019). What a bioeconomy constitutes is not strictly defined, but key aspects are increased development and use of biotechnology, more widespread and efficient use of biological materials, optimized use of energy and nutrients and promotion of biodiversity and sustainable land management (Bugge et al. 2016). Policy makers have expressed support for and interest in such a transition (Belling 2017). The shift to a bioeconomy can have a substantial effect on land use and land management intensity, for instance due to forestry practices aimed at increasing timber production for use as biofuel or as material input in production processes (Heinonen et al. 2018). This can impact water quality negatively (Forsius et al. 2016), as well as other ecosystem traits linked to recreational value, such as habitat availability and berry yields (Eyvindson et al. 2018). If the demand for biological resources increases, areas used as nature reserves might also be converted to productive areas. Flow regimes and flood frequencies can also be impacted by changes in land management (Komatsu et al. 2011, Collentine and Futter 2018), possibly leading to changes in recreational possibilities. With increasing demand for biological resources, local employment in agriculture and forestry can also be affected, as well as in the recreational sector by changes in demand for recreation.

These links between changes in land management and the potential supply of cultural services are the motivation for this study. It is of interest to decision makers and land use planners to consider how societal changes might affect the value of benefits derived from ecosystems. Though the links between landscape attributes and supply of cultural services have been studied before (Lankia et al. 2015, Queiroz et al. 2015, Brown et al. 2016), and comparative multi-national studies on public preference for ecosystem services have been done (Czajkowski et al. 2015, Dallimer et al. 2015), to our knowledge a valuation study with consistent attributes linked to bioeconomy development on a multi-national scale has not been performed. Doing this allows for analysis of the causes of variation and facilitates an integrative assessment of the effects of this transition on an international scale. Though one can argue that there are intrinsic and communal values to these cultural services that cannot be measured in monetary terms (du Bray et al. 2019), we argue that monetisation offers a way to elicit preferences under scarce resources, facilitates cost-benefit analysis for different scenarios, and allows for quantitative analysis of trade-offs and synergies between different ecosystem services.

The aims of this explorative study are:
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.

# 2. Methods and data

## 2.1 Study area selection

We chose to focus on catchments in the Nordic countries (Denmark, Finland, Norway and Sweden) because these countries have set the common goal of transitioning to a bioeconomy (Belling 2017), and Nordic catchments are often intensively used by sectors that might be impacted by the transition to a bioeconomy, like forestry and agriculture. We selected our study areas to cover the variation in land use, population density and overall geography characterizing the Nordic countries. We sought at least one catchment in each of these countries (Table 1, Figure 1). Key selection criteria were the availability of respondents, defined by a nearby city with at least 50,000 inhabitants, and the availability of data on land use, water quality and water quantity: ample data availability will allow quantification of other provisioning and regulating ecosystem services for estimation of total economic value as well (Immerzeel, in prep.). We aimed to select a mix of catchments representing both agricultural, more densely populated areas and forested, less densely populated areas. When we selected multiple catchments per country, we did so based on maximal contrast in size, land used as forest and agriculture and population density.

**Table 1. Study area descriptions showing 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 2012 CORINE land cover data, a European land cover dataset based on satellite data covering 39 countries (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[1]. We defined cities as having more than 50,000 inhabitants.

| | Halden-vassdraget | Orre-vassdraget | Odense | Simojoki | Sävjaån | Vindelälven |
|---|---|---|---|---|---|---|
| Country | Norway | Norway | Denmark | Finland | Sweden | Sweden |
| Size ($km^2$) | 1,006 | 102 | 1,199 | 1,178 | 733 | 778 |
| Forested area (%) | 67 | 3 | 6 | 76 | 60 | 75 |
| Agricultural (arable and pasture) 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) |

[1] WorldPop (www.worldpop.org - School of Geography and Environmental Science, University of Southampton; Department of Geography and Geosciences, University of Louisville; Departement de Geographie, Universite de Namur) and Center for International Earth Science Information Network (CIESIN), Columbia University (2018).

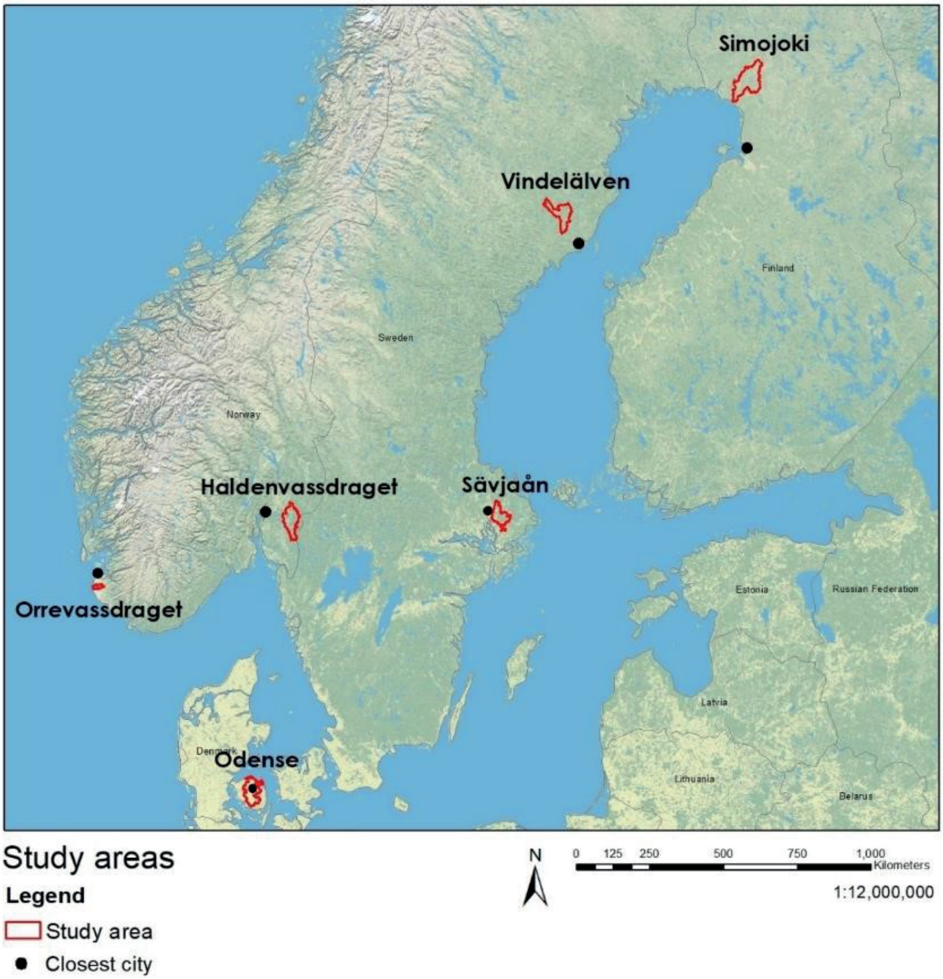



**Figure 1. A map showing the relative positions of the different study areas across the Nordic countries**. Study area boundaries are shown in red. Black dots show the city closest to the catchment as described in Table 1. This map illustrates the spatial range of study areas across the Nordic countries, as well as the range of dominant land use types. Orrevassdraget, Odense and Sävjaån are close to cities and in areas with relatively large areas of agricultural land, while Haldenvassdraget, Vindelälven and Simojoki are further from densely populated areas and contain relatively little agricultural land.

## 2.2 Survey design

We used a discrete choice experiment (DCE) to elicit preference and willingness to pay (WTP) for changes in environmental condition. A DCE is a survey based stated preference method designed for estimating the marginal value of change in separate environmental attributes (Adamowicz et al. 1998, Rakotonarivo et al. 2016) and is therefore often used in scenario studies on environmental change. The respondents are asked to choose their preferred alternative and are assumed to select the option that produces highest personal utility. The alternatives also include a cost to respondents, allowing estimation of willingness to pay for different alternatives or attribute levels. We used the guidelines as presented by Johnston et al. (2017) for designing the experiment as well as for collecting and analyzing the data. In our DCE, we presented respondents with choice cards (see Figure 2 for an example). For each choice card, respondents were asked to make a choice between three scenarios for a situation 30 years in the future. Each scenario consisted of combinations of landscape attribute levels. The combinations were selected for efficiency of analysis and did not necessarily follow a coherent storyline of future development. Key criteria for attribute choice were their expected sensitivity to change from the implementation of a bioeconomy and their understandability to respondents, based on pre-testing. In the final design, we used the attributes as described in Table 2. We did not use the term 'bioeconomy' in the survey text because we assumed it was not commonly understood among respondents, as well as not strictly defined: it might carry different connotations in different countries and between different subgroups of respondents, possibly causing uncontrolled variation.

On each choice card, Option A was a business-as-usual scenario (BAU), where current trends in land use are extended into the future – this served as an opt-out choice without changes in land management. Options B and C were alternative future scenarios, which included an annual environmental tax per household and changed landscape attribute levels. Each attribute had three possible levels, except the tax attribute, which had six levels, based on national household purchasing power and experience from previous choice experiments (Grammatikopoulou et al. 2012, Juutinen et al. 2017, Spegel 2017).[2] The attribute levels were based on the current situation per study area for each attribute. Before filling out the cards, respondents were informed on the

[2] There is a difference in tax levels between the surveys performed in 2018 (Norway and Denmark) and those performed in 2019 (Finland and Sweden). Pre-testing was done step by step in each study area as the surveys were conducted. The original bid vector, adapted to Norway, was also suitable for Denmark, but based on pre-testing the bid levels turned out to be too high for Finland and Sweden: preliminary analysis showed that respondents considered the highest bids too high. We therefore lowered the tax levels for Finland and Sweden.

current state of the various landscape attributes to familiarise them with the attributes. For the specific attribute levels per study area, see Supplement 1.

Thirty choice tasks were constructed with a D-efficient design using NGene (version 1.2.0) software. To minimise the burden on respondents, the choice tasks were divided into six blocks giving each respondent five choice tasks to respond to. The final design has a D-error of 0.001. While a Bayesian efficient design, e.g. Juutinen et al. (2014), would have been preferable, the mode of survey (personal interviews) in different countries did not allow for large scale pilot studies to attain priors. The DCE design is added in Supplement 1.

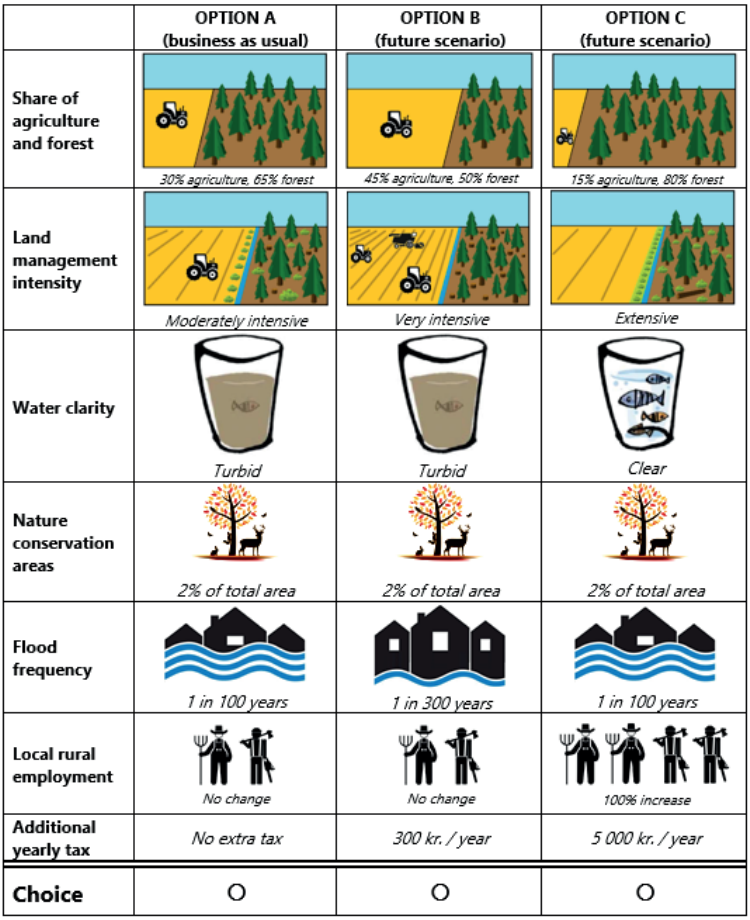



**Figure 2: Example of a choice card from the Sävjaån survey**. This shows three scenarios the respondents were presented with, each with a unique combination of landscape attribute levels and an increase in annual household tax. Each respondent was faced with five different choice cards, each with the same business as usual scenario and two unique future scenarios.

**Table 2. Landscape attributes.** This describes what the different landscape attributes mean, how they were explained to the respondents and what the different levels are.

| Attribute | Description | Levels |
|---|---|---|
| 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. | Dependent on current CORINE land use in the study areas (Buttner et al. 2000), with BAU as intermediate level. |
| 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. | • Extensive<br>• Moderately intensive - BAU<br>• Very intensive |
| 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). | • Clear (Simojoki: Clear)<br>• Moderate (Simojoki: Slightly brown)<br>• Turbid - BAU (Simojoki: Dark brown) |
| Nature conservation | The percentage share of land used as natural conservation area in total land use in the study area. | Dependent on current CORINE land use in the study areas (Buttner et al. 2000), with BAU as intermediate level. |
| 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. | Dependent on the current frequency of flooding in the study area, with BAU as worst-case level. |
| Local rural employment | The percentual change in employment in agriculture, forestry and fishery. | • No change - BAU<br>• 50% increase<br>• 100% increase |

Beyond the DCE, we asked questions on the respondent's current use of the landscape for recreational purposes and their opinion on environmental issues. For this we included the New Ecological Paradigm Scale (NEP-scale), a revised version of the New Environmental Paradigm Scale, which is used to measure the ecological-mindedness of respondents' worldview (Dunlap and Vanliere 1978). This scale has been used in a wide variety of studies to measure concern with environmental quality (Dunlap et al. 2000) and is known for its cross-cultural applicability (Hawcroft and Milfont 2010). In it, respondents are presented a series of statements that either endorse an anthropocentric world view or an ecocentric world view. Respondents must rate their agreement from 'fully disagree' to 'fully agree'. We transformed these responses to a score per respondent on a five-point scale to measure their ecological-mindedness. Hawcroft and Milfont (2010) performed a meta-analysis of studies using NEP-scores on this five-point scale, allowing us to compare our results with a large international dataset. We also asked questions on general demographic information, such as age, gender, education level, occupation and income, to be used in statistical analysis as interaction terms and for validation of sample representativeness. One of the questionnaires is available in Supplement 2.

We performed two qualitative pre-tests of the questionnaire and DCE design, to assess whether the attributes and their levels were realistic to respondents and whether the questions were easy to understand. The first was performed on the survey population in collaboration with Lars Selbekk, manager at the Haldenvassdraget River Basin District (Vannregion). He distributed the questionnaires among colleagues at the municipal office of Marker municipality in the Haldenvassdraget study area in the period 25-31 May 2018. The second pre-test was performed with a small focus group of researchers during the annual Biowater meeting from 1-6 June 2018. Around ten researchers filled out the questionnaire. This group consisted of both economists familiar with designing DCEs, and researchers unfamiliar with the theory and practice of DCEs. Here we focused on applicability of the attributes to our other study areas as well. Because we aimed at consistency between study areas, we did not perform separate qualitative pre-tests for each subsequent study area. Instead for each study area we checked the accuracy of attribute levels with local experts and evaluated on the first day of data collection for each site whether respondents understood and agreed with the design. Though Johnston et al. (2017) advise the use of quantitative pre-testing within the target population, we chose not to do so because the large geographic spread of our study sites made quantitative pre-testing in all sites logistically impossible within the study's timeframe, since we were limited to summer seasons for efficient data collection. We made minor adjustments based on feedback from the pre-testing sessions.

## 2.3 Data collection

We performed the survey on-site. This allowed us to perform face-to-face interviews, minimizing risk of misinterpretation of the questions and giving us opportunity to collect opinions on both the topic and the quality of the survey. Also, as Lindhjem and Navrud (2011) show, there appears to be no significant difference in results between internet or face-to-face interviews, but a higher response rate for face-to-face interviews, potentially limiting self-selection bias in the data. We defined the population for sampling as: all the users of cultural services supplied by the catchment landscape. This included both residents and visitors, so for each study area, we wrote a version of the questionnaire in English and made a translation in the local language. In each area, two or three surveyors visited the area and performed face-to-face interviews with respondents at local recreation hotspots, other public spaces and by going door-to-door. In addition, we set up pick-up and drop-off points for the questionnaire at recreational locations such as cafés, museums and tourist visitor centres, to increase sample size during the surveying period. Though this sampling method is not probabilistic, which might affect representativeness, it does cover a broad range of types of users, including those that would normally not be reached using sampling from population registers or other digital forms of sampling. In each of the study areas, at least one of the surveyors was fluent in the native language, and one of the surveyors was present in all study areas to ensure the interpretation of questions remained uniformly controlled. We collected the data in Norway and Denmark in the summer of 2018, and Finland and Sweden in the summer of 2019. We surveyed in each study site for a period of two to three weeks.

## 3.1 Data description

Sample size varied between the six study areas, as shown in Table 3. We present the sample characteristics that showed significant effects on preference as interaction terms in the DCE. Comparing the sample characteristics to national statistics gives an indication which differences might be caused by population differences, and which might be caused by differences in sampling. For instance, the relatively high percentage of men in the Simojoki sample is not visible in the population statistics and was likely caused by the fact that much of the recreation in the Simojoki area in summer is salmon fishing, which can be considered a male-dominated activity. When considering the most popular recreational activities per study area, walking is the only activity in the top three in all study areas. Other popular activities are swimming and just relaxing. See Supplement 3 for a figure showing the top three activities per study area.

The mean NEP-score per catchment is similar across all areas. The range between 3.47 for Haldenvassdraget and 3.70 for Sävjaån falls within the expected range when compared to Hawcroft and Milfont (2010). In that meta-analysis, 69 studies using the NEP-scale in 36 countries were compared, in which they found a mean NEP-score of 3.75, with a standard deviation of 0.32. This indicates that respondents recreating in our study areas are likely not outliers. They generally place a high value on nature and are concerned about the negative environmental impacts of human activity.

The percentage of respondents choosing only the business-as-usual scenario is relatively low in all six study areas except in Simojoki, suggesting that in general respondents did not judge the future scenarios and accompanying tax levels as unrealistic or unacceptable.

**Table 3. Socio-demographic profiles per study area.** This table summarizes the main characteristics of respondents per study area, with standard error in brackets where appropriate, in the left column. For comparison, the right column shows equivalent national statistics, taken from national central authorities on statistics (*) and the CIA World Factbook (#).

| Study area | Characteristics | Sample | National |
|---|---|---|---|
| Haldenvassdraget N=324 | Median age | 55 (1) | 39# |
| | Men | 42% (3%) | 51%# |
| | University/college degree | 48% (3%) | 36%* |
| | Median monthly gross household income | NOK 25,000 – 34,999 | 54,000* |
| | Non-nationals | 9% (2%) | 17%# |
| | Works in agriculture, forestry or fishery | 10% (0%) | 2%* |
| | Mean NEP-score | 3.47 (0.03) | - |
| | Mean travel distance to recreation area | 96 km (30) | - |
| | Respondents only choosing BAU | 5% (1%) | - |
| Orrevassdraget N=209 | Median age | 49 (1) | 39# |
| | Men | 47% (4%) | 51%# |
| | University/college degree | 77% (3%) | 36%* |
| | Median monthly gross household income | NOK 35,000 – 44,999 | 54,000* |
| | Non-nationals | 21% (3%) | 17%# |
| | Works in agriculture, forestry or fishery | 6% (0%) | 2%* |
| | Mean NEP-score | 3.48 (0.04) | - |
| | Mean travel distance to recreation area | 236 km (41) | - |
| | Respondents only choosing BAU | 5% (2%) | - |
| Odense N=284 | Median age | 30 (1) | 42# |
| | Men | 45% (3%) | 50%# |
| | University/college degree | 38% (3%) | 33%* |
| | Median monthly gross household income | DKK 7,500 – 19,999 | 43,000* |
| | Non-nationals | 18% (2%) | 16%# |
| | Works in agriculture, forestry or fishery | 4% (0%) | 2% |
| | Mean NEP-score | 3.63 (0.03) | - |
| | Mean travel distance to recreation area | 71 km (25) | - |
| | Respondents only choosing BAU | 4% (1%) | - |
| Simojoki N=197 | Median age | 49 (1) | 43# |
| | Men | 67% (3%) | 49%* |
| | University/college degree | 19% (3%) | 31%* |
| | Median monthly gross household income | EUR 3,000 – 3,999 | 2,300* |
| | Non-nationals | 1% (1%) | 5%* |
| | Works in agriculture, forestry or fishery | 7% (1%) | 4%* |
| | Mean NEP-score | 3.50 (0.04) | - |
| | Mean travel distance to recreation area | 170 km (18) | - |
| | Respondents only choosing BAU | 20% (3%) | - |
| Sävjaån N=379 | Median age | 31 (1) | 41# |
| | Men | 44% (3%) | 50%* |
| | University/college degree | 59% (3%) | 42%* |
| | Median monthly gross household income | SEK 25,000 – 29,999 | 40,000* |
| | Non-nationals | 17% (2%) | 19.1%# |
| | Works in agriculture, forestry or fishery | 2% (0%) | 2%* |
| | Mean NEP-score | 3.70 (0.03) | - |
| | Mean travel distance to recreation area | 506 km (267) | - |
| | Respondents only choosing BAU | 5% (1%) | - |
| Vindelälven N=210 | Median age | 44 (1) | 41# |
| | Men | 41% (3%) | 50%* |
| | University/college degree | 47% (4%) | 42* |
| | Median monthly gross household income | SEK 25,000 – 29,999 | 40,000* |
| | Non-nationals | 8% (2%) | 19.1%# |
| | Works in agriculture, forestry or fishery | 6% (0%) | 2%* |
| | Mean NEP-score | 3.68 (0.04) | - |
| | Mean travel distance to recreation area | 203 km (24) | - |
| | Respondents only choosing BAU | 9% (2%) | - |

## 2.4 Econometric model and analysis

We used a mixed logit (MXL) model in preference space to analyse the choice experiment data (Train 2009). The MXL model allows the coefficients to be random according to any distribution, which makes it possible for the model to take into account preference heterogeneity (Hensher et al. 2015). This model is also computationally efficient, making it possible to experiment with different set-ups without excessive time investment (McFadden and Train 2000).

The MXL model assumes that a sampled individual $n$ $(n=1, \ldots, N)$ maximizes their utility through making a choice from $C$ $(c=1, \ldots, C)$ alternatives in every choice situation $S$ $(s=1, \ldots, S)$ described by observed attributes $\boldsymbol{x}_{csn} = \{x_{csn}^{1}, \ldots, x_{csn}^{K}\}$. The utility that individual $n$ derives from choosing $c$ in situation $s$ is specified in equation (1).

$$U_{csn} = \alpha_{cn} + \boldsymbol{\beta}'_{nk}\boldsymbol{x}_{csn} + \varepsilon_{csn} \tag{1}$$

In this specification, $\alpha_{cn}$ is an alternative-specific constant, $\boldsymbol{x}_{csn}$ is a vector of the observed variables capturing the attributes of the alternatives, $\boldsymbol{\beta}'_{nk}$ represents the individual's preference vector for the attributes and $\varepsilon_{csn}$ is the i.i.d. idiosyncratic error. The probability of a respondent making a choice, based on this utility function, then is:

$$\Pr(y_{ns} = c) = \frac{\exp(\alpha_{cn} + \boldsymbol{\beta}'_{nk}\boldsymbol{x}_{csn})}{\sum_{q=1}^{C}\exp(\alpha_{qn} + \boldsymbol{\beta}'_{nk}\boldsymbol{x}_{qsn})} \tag{2}$$

In the MXL model the individual-specific preference parameters $\boldsymbol{\beta}'_{nk}$ and alternative-specific constants $\alpha_{cn}$ are not fixed for all respondents, but vary around means and are modelled as follows:

$$\beta_{nk} = \beta_k + \boldsymbol{\delta}'_k z_n + \vartheta_{nk}, \tag{3}$$

$$\alpha_{cn} = \alpha_c + \boldsymbol{\delta}'_c z_n + \vartheta_{nc}, \tag{4}$$

where $\alpha_c$ is an alternative-specific constant, and $\vartheta_{nc}$ is normally distributed (with zero mean) heterogeneity of the choice-specific constants; $\beta_k$ is the population mean of the k-attribute coefficient and $\vartheta_{nk}$ is the individual specific heterogeneity of a taste parameter. The means of the parameter distributions of $\beta_{nk}$ and $\alpha_{cn}$ are also allowed to be heterogeneous with respondents' individual characteristics $z_n$, which enter the formulas for taste parameters and alternative-specific constants with vectors of weights $\boldsymbol{\delta}_k$and $\boldsymbol{\delta}_c$ respectively. We examined individual characteristics that explain heterogeneous preferences of the alternative-specific constant, a dummy variable which was equal to the business-as-usual alternative.

Notice the observed attributes include a price attribute and non-price attributes. In this study the former was a tax that respondents were willing to pay for improving the quality of landscape. We set the parameters to follow a random distribution across respondents, where the tax attribute varies along a lognormal distribution and the non-price attributes along a normal distribution. Since there is a large number of attribute variables in the model, we chose not to analyse for correlated variables, since this complicates the model significantly and increases the risk of the model not converging. Given the selected form, the parameters can be calculated using a simulated maximum likelihood estimation. We chose to use Halton draws with 500 draws, because using this method, the simulation error is lower than with random draws and the estimation procedure is much faster (Hensher and Greene 2003). We used the software package NLOGIT 6 for the econometric analysis (Greene 2016).

Where attributes (Table 2) were quantitative, as in the fraction of land used for agriculture or the increase in flood frequency, we used a continuous variable in the model. Where attributes were qualitative, as in the clarity of the water or the intensity of land use, we used dummy variables for each attribute level.

We pooled the data from all study areas into one dataset for analysis to improve the explanatory power of the modelling. Because we used local currencies in the DCE, when pooling the data, we transformed the tax attribute to a normalized scale, where 1 is equal to the maximum tax level for each specific study area. Since the continuous variables also have different levels per catchment, we normalized these in the same manner.

After first running a basic MXL model without explanatory variables, we then ran six separate models on the same pooled dataset where for each study area we used a dummy variable as an interaction term one by one (equation 4). This allowed us to see the effect of the survey being from that catchment on the attribute coefficients.

To better understand differences between the study areas, we also analysed correlations between preference for attribute levels and characteristics of the respondents. We chose these characteristics based on their variability across catchments and potential policy relevance in a bioeconomy context. For a description of these characteristics, see Supplement 4. We used these respondent characteristics as interaction terms in the MXL model (equation 4), where we used six different respondent characteristics as interaction terms on the pooled dataset, to examine how characteristics of respondents are associated with preference heterogeneity.

As a final step we ran separate models for each catchment-specific dataset and quantified the marginal WTP for changes to the attribute levels in monetary value. We had to use separate models here because valuation is based on the original tax attribute, which differs per catchment. We used the same model specification as for preference, but using the original attribute values instead of normalized values, and the absolute tax value divided by 1,000 as an attribute to prevent scale issues (Hensher et al. 2015). We also only varied the business-as-usual coefficient according to a normal distribution because the separate datasets for each study area were not large enough to allow a model with more complexity. Since the marginal rate of substitution between two attributes is the ratio of their respective coefficients, we then computed marginal WTP of each attribute as the negative of the coefficient of the attribute divided by the coefficient for the tax variable (Hanemann 1982). To ease comparability, we then transformed these WTP values into euros using the exchange rate on the first day of the field work per study area (xe.com/currencyTables).[3] For the various model specifications in NLOGIT6, see Supplement 5.

[3] Haldenvassdraget: 1 NOK = €0.11
Orrevassdraget: 1 NOK = €0.11
Odense: 1 DKK = €0.13
Simojoki: Already in €
Sävjaån: 1 SEK = €0.09
Vindelälven: 1 SEK = €0.09

# 3. Results

## 3.1 Preferences for landscape change

Preferences for changes to the landscape across all study areas were quantified as coefficients in our basic MXL model for each of the attribute levels (Table 4). The first variable is 'business as usual', which is a dummy variable indicating that the choice is option A, i.e. the business as usual scenario. The negative coefficient suggests a preference for a changed landscape instead of continuing current trends, after taking into account the effects of the landscape attributes. From the landscape attributes the strongest effects appear to be linked to water clarity, though comparison should be made with care since the attributes are measured on different scales. It is also worth pointing out that the coefficient for high water clarity is higher than for medium water clarity, showing a stronger preference for higher clarity as well as a decreasing marginal utility. Both extensive and very intensive land management have negative coefficients, suggesting that respondents on average prefer the current intensity over a change in any direction. An increase in area of nature reserves and an increase in local employment also appear to have a positive effect on probability of choice, while an increase in the frequency of flooding has a negative effect.

**Table 4. MXL attribute preference coefficients.** This shows coefficients of preference for the different attribute levels, with standard errors and stars indicating the level at which the coefficients are statistically significant. The type of variable is also stated. The column 'RP distribution of standard deviation' shows the standard deviation of the distribution of the random parameters in the model specification.

| Variable | Type | Coefficient for preference (SE) | RP distribution of standard deviation |
|---|---|---|---|
| Business as usual | Dummy | -0.88 (0.21) *** | 1.93 *** |
| Increase in agriculture over forest | Continuous | 0.96 (0.15) *** | 1.46 *** |
| Very intensive land management | Dummy | -0.48 (0.13) *** | 0.32 |
| Extensive land management | Dummy | -0.60 (0.14) *** | 1.00 *** |
| Medium water clarity | Dummy | 1.96 (0.22) *** | 1.28 *** |
| High water clarity | Dummy | 2.77 (0.12) *** | 1.41 *** |
| Increase in nature reserve area | Continuous | 1.51 (0.13) *** | 1.32 *** |
| Increase in flood frequency | Continuous | -0.61 (0.11) *** | 0.14 |
| Increase in local employment | Continuous | 0.44 (0.09) *** | 0.08 |
| Tax increase | Continuous | -1.08 (0.29) *** | 3.22 *** |
| N (observations) | | | 6956 |
| McFadden Pseudo R-squared | | | 0.33 |
| Log-likelihood value | | | -5137.22 |

***, **, * ==> Significance at 1%, 5%, 10% level respectively.

## 3.2 Study area effects

For comparing the effects of study area on preferences, we performed mixed logit regressions on the pooled dataset, each time using a dummy variable for one of the study areas as an interaction term. We performed likelihood ratio (LR) tests as in Poe et al. (1994) for each of the study sites to see whether differences in attribute preference varied significantly across study areas, which showed all study areas except Vindelälven varied significantly from the others at the 5% level ($LR > \chi^2_{21,5} = 32.67$).

Some significant differences in attribute preference among the areas appeared (Table 5). The major differences lie in the preference for the ratio of forest and agriculture. In general, respondents in the more agricultural study areas prefer an increase of forest over agriculture, while respondents in the more forested areas prefer increasing agricultural land at the cost of forest. It also appears that preference for improved water clarity, though high everywhere, is significantly higher in the Swedish study areas. Respondents in the Haldenvassdraget study area appear to differ from the others in having a negative preference for a shift to more extensive agriculture, as well as a less strong positive preference for increasing the percentage of land used for nature reserves. In Odense there appears to be a stronger negative preference for increasing flood frequency, while in Sävjaån this negative preference appears weaker.

**Table 5. Effects of study area on attribute preference.** This shows the effect of the survey being performed in a study area on preference for attribute levels in the DCE. Preference in the total sample is greyed out: the values in black represent the difference in preference between the study area subsample and the total sample.

| | Halden-vassdraget | Orre-vassdraget | Odense | Simojoki | Sävjaån | Vindelälven |
|---|---|---|---|---|---|---|
| Variable | | | Coefficient for preference (SE) | | | |
| Business as usual | -1.15 (0.26) *** | -1.40 (0.25) *** | -1.08 (0.26) *** | -0.99 (0.22) *** | -1.05 (0.26) *** | -1.08 (0.23) *** |
| Increase in agriculture over forest | 0.57 (0.17) *** | 0.72 (0.15) *** | 0.86 (0.16) *** | 0.94 (0.16) *** | 1.00 (0.16) *** | 0.77 (0.17) *** |
| Very intensive land management | -0.42 (0.14) *** | -0.50 (0.14) *** | -0.49 (0.14) *** | -0.54 (0.14) *** | -0.47 (0.15) *** | -0.50 (0.14) *** |
| Extensive land management | -0.38 (0.14) *** | -0.46 (0.14) *** | -0.53 (0.15) *** | -0.64 (0.14) *** | -0.61 (0.15) *** | -0.59 (0.15) *** |
| Medium water clarity | 1.87 (0.18) *** | 1.82 (0.19) *** | 1.86 (0.20) *** | 1.91 (0.21) *** | 1.54 (0.19) *** | 1.78 (0.20) *** |
| High water clarity | 2.69 (0.13) *** | 2.60 (0.12) *** | 2.71 (0.13) *** | 2.75 (0.12) *** | 2.75 (0.13) *** | 2.61 (0.12) *** |
| Increase in nature reserve area | 1.47 (0.14) *** | 1.24 (0.14) *** | 1.38 (0.15) *** | 1.46 (0.13) *** | 1.35 (0.13) *** | 1.45 (0.13) *** |
| Increase in flood frequency | -0.49 (0.12) *** | -0.38 (0.13) *** | -0.47 (0.13) *** | -0.59 (0.12) *** | -0.61 (0.12) *** | -0.54 (0.12) *** |
| Increase in local employment | 0.40 (0.10) *** | 0.34 (0.10) *** | 0.39 (0.11) *** | 0.43 (0.09) *** | 0.34 (0.10) *** | 0.39 (0.09) *** |
| Tax increase | -0.36 (0.20) * | -0.34 (0.21) | -0.60 (0.27) ** | -1.15 (0.28) *** | -0.86 (0.25) *** | -0.98 (0.27) *** |
| Variable | | | Coefficient for interaction effect (SE) | | | |
| Business as usual | -0.85 (0.59) | -0.46 (0.64) | 0.73 (0.47) | 0.95 (0.74) | -1.01 (0.53) * | 0.98 (0.72) |
| Increase in agriculture over forest | 0.59 (0.33) * | -1.24 (1.13) | 0.79 (1.15) | -0.05 (0.42) | -1.06 (0.36) *** | 0.61 (0.35) * |
| Very intensive land management | -0.44 (0.37) | -0.16 (0.39) | 0.40 (0.33) | 0.48 (0.41) | -0.18 (0.29) | 0.33 (0.39) |
| Extensive land management | -0.71 (0.36) ** | -0.23 (0.38) | -0.07 (0.35) | 0.43 (0.43) | 0.40 (0.31) | 0.21 (0.38) |
| Medium water clarity | -0.49 (0.36) | -0.18 (0.38) | -0.20 (0.37) | -0.59 (0.42) | 0.93 (0.34) *** | 0.90 (0.52) * |
| High water clarity | -0.13 (0.23) | 0.13 (0.26) | -0.03 (0.24) | -0.15 (0.32) | -0.31 (0.22) | 1.05 (0.30) *** |
| Increase in nature reserve area | -0.74 (0.32) ** | 0.46 (0.37) | 0.34 (0.31) | 0.16 (0.46) | 0.01 (0.32) | 0.06 (0.38) |
| Increase in flood frequency | 0.14 (0.29) | -0.42 (0.33) | -0.67 (0.28) ** | -0.01 (0.37) | 0.66 (0.29) ** | -0.05 (0.33) |
| Increase in local employment | -0.25 (0.24) | -0.16 (0.28) | 0.31 (0.21) | 0.17 (0.32) | 0.03 (0.22) | 0.27 (0.30) |
| Tax increase | -0.78 (0.42) * | -1.47 (0.63) ** | -18.94 (360,425) | -2.25 (0.37) *** | 1.02 (0.24) *** | 0.20 (0.36) |
| N (observations) | 6956 | 6956 | 6956 | 6956 | 6956 | 6956 |
| McFadden Pseudo R-squared | 0.33 | 0.33 | 0.33 | 0.33 | 0.33 | 0.33 |
| Log-likelihood value | -5119.57 | -5129.89 | -5138.91 | -5096.58 | -5121.13 | -5127.83 |

***, **, * ==> Significance at 1%, 5%, 10% level respectively.

## 3.3 Explaining variability

For estimating the effects of respondent characteristics on preference, we once again ran mixed logit regressions on the pooled dataset, using six respondent characteristics as interaction terms in a single model, to account for possible correlations (Table 6). Those travelling more than 25 kilometres (i.e. likely non-residents) appear to have a more positive preference for an increase in tax and for a one-step improvement in water clarity, but a stronger preference for changing the intensity of land management in either direction. Respondents with a higher NEP-score appear to have a stronger preference for improving water clarity and increasing the percentage of land used as nature reserve, but also a more negative preference for an increase in tax. Age appears to have a positive effect on preference for the business-as-usual scenario as well as for having more agricultural area, but a negative effect on preference for a one-step improvement in water clarity and an increase in tax. Respondents with higher education appear to show stronger preference for increasing the area covered by forest and having high water clarity, while the effects of high income are a possible stronger negative preference for very intensive land management. Respondents working in a sector directly linked to natural resources appear to have less positive preference for high water clarity.

We found that respondents that indicated they found the scenarios unrealistic do not have significantly different preference for the different attribute levels. The only significant effect is a weaker negative preference for choosing the business as usual scenario.

**Table 6. Effects of respondent characteristics on attribute preference.** This shows the effect of selected respondent characteristics on preference for attribute levels in the DCE. Preference in the total sample is greyed out: the values in black represent the effects on preference for the respondent characteristics on preference in the total sample.

| | Travels more than 25 km | NEP-score | Age | Higher education | High income | Employed in forestry, agriculture or fishery |
|---|---|---|---|---|---|---|
| Variable | | | Coefficient for preference (SE) | | | |
| Business as usual | -1.86 (2.21) | -1.86 (2.21) | -1.86 (2.21) | -1.86 (2.21) | -1.86 (2.21) | -1.86 (2.21) |
| Increase in agriculture over forest | -0.52 (1.45) | -0.52 (1.45) | -0.52 (1.45) | -0.52 (1.45) | -0.52 (1.45) | -0.52 (1.45) |
| Very intensive land management | -1.47 (1.25) | -1.47 (1.25) | -1.47 (1.25) | -1.47 (1.25) | -1.47 (1.25) | -1.47 (1.25) |
| Extensive land management | -2.20 (1.34) | -2.20 (1.34) | -2.20 (1.34) | -2.20 (1.34) | -2.20 (1.34) | -2.20 (1.34) |
| Medium water clarity | 0.57 (1.34) | 0.57 (1.34) | 0.57 (1.34) | 0.57 (1.34) | 0.57 (1.34) | 0.57 (1.34) |
| High water clarity | 2.09 (0.88) ** | 2.09 (0.88) ** | 2.09 (0.88) ** | 2.09 (0.88) ** | 2.09 (0.88) ** | 2.09 (0.88) ** |
| Increase in nature reserve area | -2.44 (1.17) ** | -2.44 (1.17) ** | -2.44 (1.17) ** | -2.44 (1.17) ** | -2.44 (1.17) ** | -2.44 (1.17) ** |
| Increase in flood frequency | 1.44 (1.14) | 1.44 (1.14) | 1.44 (1.14) | 1.44 (1.14) | 1.44 (1.14) | 1.44 (1.14) |
| Increase in local employment | -0.75 (0.83) | -0.75 (0.83) | -0.75 (0.83) | -0.75 (0.83) | -0.75 (0.83) | -0.75 (0.83) |
| Tax increase | 4.14 (1.05) *** | 4.14 (1.05) *** | 4.14 (1.05) *** | 4.14 (1.05) *** | 4.14 (1.05) *** | 4.14 (1.05) *** |
| Variable | | | Coefficient for interaction effect (SE) | | | |
| Business as usual | -0.08 (0.57) | -0.21 (0.59) | 0.04 (0.02) ** | -0.58 (0.56) | -0.16 (0.59) | -2.06 (1.33) |
| Increase in agriculture over forest | -0.05 (0.37) | -0.05 (0.36) | 0.03 (0.01) *** | -0.62 (0.36) * | 0.05 (0.38) | -0.15 (0.87) |
| Very intensive land management | 0.58 (0.33) * | 0.15 (0.33) | 0.01 (0.01) | -0.21 (0.34) | -0.58 (0.33) * | -0.61 (0.67) |
| Extensive land management | 0.58 (0.34) * | 0.45 (0.35) | -0.00 (0.01) | 0.20 (0.34) | -0.19 (0.36) | -0.20 (0.66) |
| Medium water clarity | -0.76 (0.35) ** | 0.66 (0.36) * | -0.02 (0.01) ** | 0.07 (0.36) | 0.55 (0.38) | -1.05 (0.76) |
| High water clarity | 0.39 (0.22) * | 0.07 (0.23) | 0.00 (0.00) | 0.37 (0.22) * | -0.09 (0.22) | -0.93 (0.56) * |
| Increase in nature reserve area | 0.21 (0.31) | 0.98 (0.30) *** | 0.00 (0.01) | 0.43 (0.29) | 0.01 (0.32) | -1.05 (0.68) |
| Increase in flood frequency | 0.00 (0.28) | -0.44 (0.30) | -0.01 (0.01) | 0.24 (0.27) | 0.24 (0.30) | 0.97 (0.75) |
| Increase in local employment | 0.05 (0.24) | 0.28 (0.22) | 0.01 (0.01) | -0.33 (0.22) | -0.09 (0.25) | -0.91 (0.61) |
| Tax increase | 0.84 (0.28) *** | -0.94 (0.30) *** | -0.03 (0.01) *** | -0.25 (0.28) | 0.39 (0.30) | 0.36 (0.60) |
| N (observations) | 4958 | 4958 | 4958 | 4958 | 4958 | 4958 |
| McFadden Pseudo R-squared | 0.36 | 0.36 | 0.36 | 0.36 | 0.36 | 0.36 |
| Log-likelihood value | -3511.29 | -3511.29 | -3511.29 | -3511.29 | -3511.29 | -3511.29 |

***, **, * ==> Significance at 1%, 5%, 10% level respectively.

## 3.4 Willingness to pay for landscape change

Looking at differences in willingness to pay for landscape change, there is a clear distinction between the first three study areas and the second three. WTP in the Finnish and two Swedish areas is substantially lower than in the Danish and the two Norwegian areas (Table 7). To some extent this can be explained by the difference in tax levels between the surveys performed in 2018 (Norway and Denmark) and in 2019 (Finland and Sweden). Since there is a relatively low percentage of voters only choosing BAU as shown in Table 4, this suggests that most respondents felt the tax levels were acceptable. Taking the difference between the two groups of study areas into account by looking at the relative differences between attributes per study area, there is still a consistently high WTP for improving water clarity across all study sites compared to the other landscape attributes. See Supplement 6 for estimation results in preference space for each study area.

**Table 7. WTP estimations for attribute levels per study area, based on area specific MXL models in preference space, in euros per household per year.** WTP is measured as an increase in a new household tax for environmental improvement. The attribute preference outputs of the area-specific MXL models can be found in Supplement 6.

| | Haldenvassdraget | Orrevassdraget | Odense | Simojoki | Sävjaån | Vindelälven |
|---|---|---|---|---|---|---|
| Variable | WTP (95% Confidence interval) | | | | | |
| 1% of area from forest to agriculture | € 28.58 **<br>(4.09 - 53.07) | -€ 59.35 ***<br>(-92.11 - -26.6) | -€ 58.01 ***<br>(-89.92 - -26.11) | -€ 2.42<br>(-14.78 - 9.95) | € 1.58<br>(-3.85 - 7.02) | € 21.46<br>(-5.47 - 48.39) |
| Very intensive land management | -€ 1 479.32 ***<br>(-2 562.12 - -396.53) | -€ 1 217.49 ***<br>(-2 113.17 - -321.83) | -€ 663.91<br>(-1 764.71 - 436.89) | -€ 28.73<br>(-134.07 - 76.61) | -€ 105.49<br>(-262.22 - 51.25) | -€ 131.87<br>(-462.34 - 198.6) |
| Extensive land management | -€ 1 283.17 **<br>(-2 386.71 - -179.64) | -€ 580.26<br>(-1 599.44 - 438.92) | -€ 431.25<br>(-1 630.2 - 767.7) | € 91.93<br>(-21.16 - 205.02) | -€ 28.55<br>(-203.1 - 146.01) | -€ 201.20<br>(-620.22 - 217.82) |
| Medium water clarity | € 1 261.59 **<br>(253.45 – 2 269.72) | € 1 389.12 **<br>(208.19 – 2 570.06) | € 2 021.70 **<br>(412.76 – 3 630.64) | € 86.41<br>(-32.61 - 205.44) | € 596.99 ***<br>(300.04 - 893.94) | € 782.70 **<br>(104.55 – 1 460.85) |
| High water clarity | € 2 859.38 ***<br>(1 844 – 3 874.76) | € 2 373.28 ***<br>(744.06 – 4 002.49) | € 3 048.42 ***<br>(1 101.28 – 4 995.55) | € 245.37 ***<br>(101.03 - 389.71) | € 618.50 ***<br>(310.89 - 926.12) | € 1 094.82 **<br>(225.3 – 1 964.35) |
| 1% of area to nature reserve | € 65.25<br>(-29.82 - 160.32) | € 42.62<br>(-24.74 - 109.98) | € 360.12 *<br>(-39.92 - 760.16) | € 0.54<br>(-5.11 - 6.18) | € 79.70 **<br>(15.12 - 144.29) | € 95.58<br>(-23.63 - 214.78) |
| 1-year increase in flood frequency | -€ 13.08<br>(-64.6 - 38.44) | -€ 7.10<br>(-86.51 - 100.71) | -€ 54.59<br>(-56.62 - 165.8) | € 1.06<br>(-3.48 - 1.36) | -€ 0.28<br>(-0.42 - 0.97) | -€ 2.26<br>(-1.83 - 6.36) |
| 1% increase in local employment | -€ 2.50<br>(-6.32 - 1.32) | -€ 3.85<br>(-9.18 - 1.48) | € 1.08<br>(-5.88 - 8.04) | -€ 0.35<br>(-1.36 - 0.65) | € 1.37<br>(-0.29 - 3.03) | € 2.45<br>(-1.26 - 6.16) |
| N (observations) | 1274 | 908 | 1123 | 876 | 1797 | 1324 |
| McFadden Pseudo R-squared | 0.29 | 0.31 | 0.32 | 0.34 | 0.31 | 0.39 |
| Log-likelihood value | -996.51 | -685.17 | -840.61 | -630.54 | -1371.29 | -651.31 |

***, **, * ==> Significance at 1%, 5%, 10% level respectively.

# 4. Discussion

Our findings show that respondents in our selected Nordic catchments have statistically significant preferences for landscape changes associated with the transition to bioeconomy. Improving water clarity is a strongly preferred change in all study areas. An increase in area used for both agriculture and nature reserves is also preferred, as well as reducing the frequency of floods and the number of jobs provided by forestry, agriculture and fishery, sectors closely linked to the development of a bioeconomy. This suggests that when land management practices change due to the development of a bioeconomy, this may affect the appreciation of cultural services supplied in these areas. For instance, the transition to a bioeconomy can cause both an increase in land used for forestry, as well as an increase in management intensity for these forests, which both can impact water quality negatively (Forsius et al. 2016). Since preference among respondents for increasing water clarity is positive and for increasing the intensity of land management is negative, our findings suggest that both increasing the area used for forestry and increasing management intensity would decrease the value of these landscapes for respondents visiting the area for recreation and non-use services. Of course, land use policies are rarely implemented on catchment scale but often on national scale, so preference of the respondents found in our study areas are likely not the only targets for policy changes. For a social optimum on a national scale, other stakeholders, as well as implementation costs, need to be considered as well.

We also found differences between study areas that bring nuance to the results from analysing the total dataset, and that can have implications for possible future land use changes. Respondents from the Swedish study areas show a significantly stronger preference for improving water clarity and the general preference for shifting from forest to agriculture does not hold in all areas, suggesting that the change in value from cultural services depends on the location of changes in environmental and landscape attributes. Preference for increasing nature reserves also varies across catchments. Since increasing landscape productivity for the bioeconomy might reduce land available for nature reserves, this suggests location selection for land use change should take these variations in preference into account if the supply of cultural services is a consideration. The average preference for an increase in agricultural land over forested land was stronger in catchments that already have a relatively large share of forest like Haldenvassdraget and Vindelälven. In a more agricultural and densely populated catchment like Sävjaån, an increase in forest over agriculture seems in fact preferred. This suggests that where to increase the land used for forestry in a bioeconomy matters when considering the value of the landscape for the supply of cultural services. Respondents appear to favour a mixed landscape, irrespective of being resident or visitor. This corresponds with previous findings in a study on German forest landscapes by Elsasser et al. (2010), where results from a DCE show a positive preference for an increase in landscape diversity as well.

Our analyses indicate that respondent characteristics affect preference for landscape change. Respondents travelling from further away appear to have a less negative preference for changing land management intensity, as well as a lower positive preference for a medium improvement in water clarity. This suggests that respondents feel more strongly about changing the landscape when they live closer to it, an intuitively logical interpretation. Since there are significant differences in travel distance between the study areas (Table 3), this can help explain variation. This also suggests that in areas with a higher population density, the aggregate effects of changing the landscape on the value of cultural services can be higher because more people live close to the area in which they recreate. Respondents with higher NEP-scores also appear to have a stronger positive preference for improving water clarity and increasing the area used as nature reserve. Since there are indications that average NEP-scores are increasing over time (Dunlap et al. 2000, Inglehart and Baker 2000), this effect needs to be taken into account when studying future scenarios where societal change is a factor. Age also has significant effects on preference: with higher age, respondents seem more likely to choose the business as usual scenario and have stronger negative preference for increased tax, as well as a stronger positive preference for increasing agriculture at the cost of forest and a weaker positive preference for increasing water clarity. This indicates that in the future, population preference might shift towards stronger preference for increasing forest area and improving water clarity, and a higher willingness to pay for that.

When analysing the preference data, we worked under several assumptions. In our MXL model, we transformed discrete attribute levels in the choice cards into continuous scales where possible, for instance in the percentage of land used as nature reserve. Estimating the marginal effect of increasing an attribute level based on discrete levels in the DCE in this way assumes constant marginal benefit. This is not necessarily true (Bateman et al. 2011). An increase from 1% to 2% might be much more preferable to respondents than an increase from 30% to 31%. Since the baseline levels varied per study area, this is an issue to keep in mind. We also did not include correlation effects between attributes in the MXL model, even though preference for these attributes might be correlated. We chose not to include these to prevent estimation issues caused by the complexity of the model (Greene 2016). When using socio-demographic characteristics as interaction terms in the choice models, we did not take into account that these might be latently dependent on other observed or unobserved variables (Sheremet et al. 2018). This may lead to endogeneity issues in the model estimation. For further analysis on this issue, we suggest further study by performing a hybrid MXL model analysis on the data, as described by (Czajkowski et al. 2017). Another interesting avenue for future research is a latent class model analysis on the data to identify different segments of respondents similarly as in Hess and Train (2017) and Hensher et al. (2015).

The WTP estimations show statistically significant WTP values for changing the landscape in each of the study areas. This will be valuable information when analysing possible trade-offs in scenarios for land use change, especially when taking into account the monetary values of other ecosystem services, such as crop and timber production. The WTP values from our DCE can be included in an ecosystem services framework that uses monetisation as a standardisation method, as for instance in Vermaat et al. (2016). This allows for comparison of the effects of scenarios on total ecosystem services provision, including how respondents from this study value the landscape for recreation and non-use benefits. It also allows for the comparison of distributions of benefits across different societal stakeholder groups, where respondents recreating in the area are one of those groups. However, care must be taken when interpreting the WTP values. There is inherent uncertainty in the WTP values derived from DCEs because there is a risk that respondents do not think the payment vehicle is realistic, possibly causing them to overstate how much they would be willing to pay (Johnston et al. 2017). However, we did not find evidence that respondents experienced the scenarios as unrealistic in their preference for attribute levels. Care should also be taken in comparing the WTP estimates of the different study areas. Since we changed the tax levels in the DCE for the second year of surveying, comparing the WTP estimates from Norway and Denmark with those from Sweden and Finland should take this change into account. However, since there were relatively few respondents that only chose the business as usual scenario (Table 4), we assume that respondents judged the tax levels in our DCE to be realistic and acceptable.

## Concluding remarks

This study draws on extensive data from four different countries and shows that across our Nordic study sites, respondents benefitting from cultural ecosystem services have clear preference for a more equal distribution of agriculture and forest, improved water clarity, increased area used for nature reserves, reduced flood risk and increased employment from agriculture, forestry and fishery. There is significant variation in preferences between study areas, which appear linked to characteristics of our respondents. The preferences for landscape change and the variation in these preferences carry implications for future policy decisions. If Nordic societies transition toward a bioeconomy, this can affect the landscape attributes that we studied and that contribute to the supply of cultural ecosystem services. Our results indicate that how and where land use changes can impact the total value of cultural ecosystem services delivered by Nordic catchments. As Raudsepp-Hearne et al. (2010) showed, increasing the output of provisioning services in a growing bioeconomy can lead to trade-offs with the supply of cultural services like recreational opportunities and appreciation of nature. Policy aimed at minimizing these trade-offs should consider local differences in preference: for instance, our results indicate increasing forested area is most beneficial to cultural services supply in agriculturally dominated areas. Of course, for a socially optimal solution, other ecosystem services and costs of implementation also have to be considered. Nonetheless, our WTP estimates can be useful for integrated assessment, to make comparison of producing different bundles of ecosystem services possible.

We suggest further study in two directions. First, we think that a further analysis of the determinants of preference and WTP is needed to explain how much different societal groups benefit from the cultural services supplied by Nordic catchments. Subgroups of beneficiaries can be determined, and larger study sites can be added, including more detailed spatial analysis, to better understand how characteristics of the catchment affect the value of the cultural services they provide. Secondly, we suggest quantifying the impact of the transition to a bioeconomy on total ecosystem services provision from Nordic catchments. This can be done by integrating the results from this work into a quantification of the total economic value of ecosystem services provision. Our WTP estimates make their inclusion in an integrative ecosystem services framework possible. Doing so elucidates the relative importance of cultural services and the trade-offs between different ecosystem services and their beneficiaries caused by land use change.

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# Supplementary materials

1. Attribute levels per study area and DCE design
2. Sävjaån questionnaire (English, DCE version 1)
3. Top three recreational activities per study area
4. Characteristics of respondents and the landscape used to explain variation of preference between study areas
5. Model set up for the pooled mixed logit estimation and the WTP estimation
6. Full model outputs

Until publication the supplementary materials are available upon request from the first author.

# Paper II

PLOS ONE

RESEARCH ARTICLE

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

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## Abstract

Nordic catchments provide a variety of ecosystem services, from harvestable goods to mitigation of climate change and recreational possibilities. Flows of supplied ecosystem services depend on a broad range of factors, including climate, hydrology, land management and human population density. The aims of this study were: 1) to quantify the total economic value (TEV) of consumed ecosystem services across Nordic catchments, 2) to explain variation in ecosystem service value using socio-geographic and natural factors as explanatory variables in multiple linear regression, and 3) to determine which societal groups benefit from these ecosystem services. Furthermore, we tested the scientific rigour of our framework based on the concept of final ecosystem services (FES). We used a spatially explicit, integrative framework for ecosystem services quantification to compile data on final ecosystem services provision from six catchments across Denmark, Finland, Norway and Sweden. Our estimates showed a broad variation in TEV and in the proportion contributed by separate services, with the highest TEV of €7,199 ± 4,561 $ha^{-1}$ $y^{-1}$ (mean ± standard deviation) in the Norwegian Orrevassdraget catchment, and the lowest TEV of €183 ± 517 $ha^{-1}$ $y^{-1}$ in the Finnish Simojoki catchment. The value of material services was dependent on both geographic factors and land management practices, while the value of immaterial services was strongly dependent on population density and the availability of water. Using spatial data on land use, forest productivity and population density in a GIS analysis showed where hotspots of ecosystem services supply are located, and where specific stakeholder groups benefit most. We show that our framework is applicable to a broad variety of data sources and across countries, making international comparative analyses possible.

## 1 Introduction

Society depends on ecosystems in a multitude of ways: these can be easily visible and quantifiable processes as well as more concealed ones. The focus in land management decisions has historically been on maximising the production of material goods such as agricultural products

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and forestry goods [1–3]. This can lead to societally sub-optimal results, since more concealed benefits received from ecosystems can be negatively impacted by a focus on marketable goods [1]. Researchers use ecosystem services frameworks to assess all the benefits society receives from its interaction with ecosystems, and to avoid the limited focus on those already quantified in standard economic analysis, like agricultural production [4]. To approximate a societal optimum in the benefits received from ecosystems, a complete overview of the net value of all relevant ecosystem services is useful. For instance, a landscape can create value both through agricultural production, as well as through recreational possibilities. The former can be quantified with established methods using statistics on agricultural production, the latter is typically not included in standard economic analysis. To reach a societal optimum, optimisation has to take both into account. Additionally, a framework that accounts for the manner in which underlying landscape and ecosystem characteristics influence these values will allow for an analysis of the effects of focused policy measures for different beneficiary groups in society, and can show possible synergies and trade-offs between ecosystem services.

However, applying the concept of ecosystem services raises questions: how do we actually benefit from ecosystems? Which processes are of value to us, and how do these benefits flow into societies [5]? Why are some services often ignored in policy decisions while others are not [3]? The open-ended nature of these questions has led to ongoing debate among researchers [6], including calls for 'a clear and robust definition' of what an ecosystem service is [7]. Currently, a variety of frameworks is available with definitions of variable precision [7–10]. Such heterogeneity hinders comparability across different studies, countries and likely also spatial scales [11,12], and it prevents well-informed generalisations for decision making and policy implementation [13].

In an effort to standardise ecosystem services accounts, the concept of final ecosystem services (FES) was introduced by Boyd and Banzhaf [8], and further elaborated on by Wallace [5]. Their definition of FES is: 'components of nature, directly enjoyed, consumed, or used to yield human well-being', in which the key term in our view is 'directly'. A FES requires a direct link to a component of nature and a human beneficiary, which differentiates this definition from other frameworks in general use, such as the Common International Classification of Ecosystem Services (CICES) [9] and Nature's Contributions to People by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services [14], where quantified services can include outputs from other processes which would be intermediate steps under the FES definition. The concept of FES was also specifically designed to allow monetary valuation of ecosystem services to support environmental accounting [8]. This means that hard to quantify ecosystem services, such as heritage landscape value, are easily excluded from FES-based frameworks [7]. We, however, choose to use it because of its methodological rigour in definitions and its ability to quantitively compare effects for different groups of society and across different spatial scales.

Whereas Boyd and Banzhaf [8] argue for 'final' services as the ultimate flows that are truly used by society (see also Bateman, Brouwer [15]), Boerema, Rebelo [12] present six recommendations for ecosystem service assessments. In the current study we developed a framework that combines the rigor of the FES approach with locally available land use and statistical data and closely follows the recommendations from Boerema, Rebelo [12]. Its conceptual basis in literature, structure and content is presented in the methods section below. We use estimated monetary value as an indicator of the importance of the societal benefit of each service. We do so for comparative reasons, as monetary value has strong communicative power [16].

We developed this framework because of our interest in the importance of those ecosystem services that are overseen in spatial planning decisions and that may favour particular user groups or sectors [17], and to supply decision makers with enough information to take these services into account. Additionally, it may stimulate further scientific debate on the definition of ecosystem services and the usefulness of ecosystem services frameworks. We populate our framework with empirical, locally and publicly available, statistical data from six catchments across four Nordic countries (Denmark, Finland, Norway and Sweden). These six catchments cover a considerable range in land use intensity, land use type, landscape and climate, and allow us a comparative approach beyond the scope of a single case study area (as in Queiroz, Meacham [18] or Zhou, Vermaat [19]), whilst we maintain the rigor of one common methodology, and apply sufficient resolution to analyse underlying explanatory factors and spatial relationships. Our interest is in the relative importance of different ecosystem services and we ask ourselves 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?

To structure our inquiry, we formulated the following hypotheses based on literature [18,20–22]:

1. Where primary sectors (forestry, agriculture) dominate land use, material services provide the most societal value;
2. Where population density is high, immaterial (e.g. recreational) services are of the highest value;
3. Recreational value is strongly linked to the presence of water.

Additionally, we aim to assess to what degree the framework meets the criteria for a scientifically rigorous method for ecosystem services quantification using criteria proposed by Boerema, Rebelo [12].

## 2 Method

### 2.1 Study site selection

We define our study sites using catchments, or river basins, since these form a naturally bounded system based on hydrology, which is a key factor in the supply of many ecosystem services [23] and are thus more suitable to ecosystem service estimation than administrative boundaries.

We selected our study sites to cover the variation in land use, population density and overall geography characterizing the Nordic countries. We sought at least one catchment in each of these countries (Table 1, Fig 1). We aimed to select a mix of catchments representing both agricultural, more densely populated areas and forested, less densely populated areas. When we selected multiple catchments per country, we did so based on maximal contrast in size, land covered by forest and agriculture, and population density. Haldenvassdraget, Vindelälven and Simojoki here represent sparsely populated, forest-dominated catchments in different geographic regions. Odense, Orrevassdraget and Sävjaån represent more densely populated catchments dominated by agriculture and urban areas.

### 2.2 Defining final ecosystem services

Our framework of FES builds on the 'Mononen-cascade' as applied in Boerema, Rebelo [12] and Vermaat, Immerzeel [21], and is based on the cascade perspective as described in

**Table 1. Study site descriptions showing 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.**

| | Halden-vassdraget | Orre-vassdraget | Odense | Simojoki | 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) |

We took land use values for forest, agriculture, water bodies and urban area from 2016 CORINE land cover data [24]. 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.

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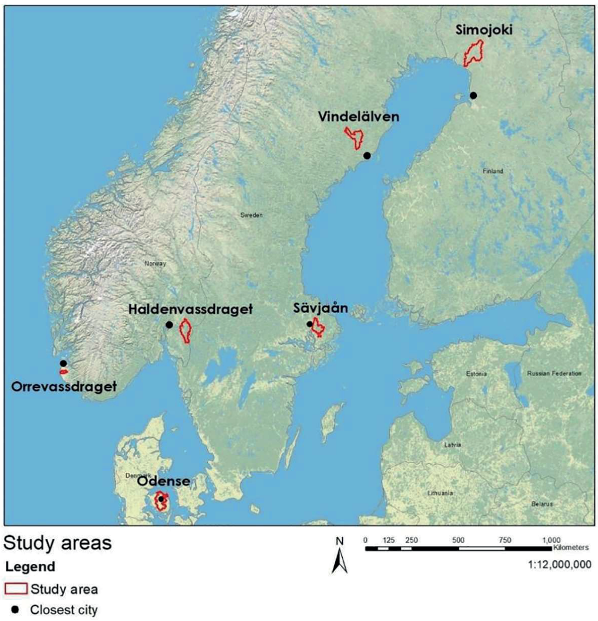


**Fig 1. A map showing the relative positions of the different study sites across the Nordic countries.** The basemap is provided by ESRI[a]. Study site boundaries are shown in red. Black dots show the city closest to the catchment as described in Table 1. This map illustrates the spatial range of study 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 areas of agricultural land, while Haldenvassdraget, Vindelälven and Simojoki are further from densely populated areas and contain relatively little agricultural land. [a] Esri. "World Topo Base". February 5, 2020. https://www.arcgis.com/home/item.html?id=3a75a3ee1d1040838f382cbefce99125. (September 14, 2020).

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Mononen, Auvinen [25]. A key aspect of this framework is the stepwise flow from ecosystem structure and processes to the valuation of societal benefits. We define the FES as the point where, through a beneficiary, the ecosystem functioning flows into a societal benefit. We define FES based on the conditions set by Johnston and Russell [26], so that a biophysical outcome is a final ecosystem service if a beneficiary's welfare is influenced by it directly and with all other ecosystem outputs held constant, and an ecological process has served as an input to the biophysical outcome.

## 2.3 Quantifying FES consumption and TEV

We quantify consumption of FES using the concept of total economic value (TEV). The method of TEV quantification has been described in Pearce and Turner [27], TEEB [28] and de Groot, Alkemade [13], and has been used in a wide range of ecosystem services accounting studies including Costanza, dArge [29] and Wustemann, Meyerhoff [30]. This method monetizes the value of ecosystem services, including those services that are typically not included in standard economic analysis. By aggregating the values of ecosystem services into TEV in monetary terms, we allow for analysis of relative value of different services in a transparent way.

Using 1) the Mononen-cascade, 2) the CICES framework, 3) the above definition of FES and 4) established ecosystem services frameworks and applications in similar study sites [9,21,31,32], we constructed a list of FES consumed in our six study sites with their corresponding beneficiaries (Table 2). The list of beneficiaries aims to show every group in society directly benefitting from the ecosystem.

FES consumption is a flow over a given time period, where interaction with the ecosystem by a beneficiary yields a benefit. We chose to estimate the consumption of FES at an annual time step, allowing for the use of annual statistics as input data for quantification and valuation. This yields a TEV $yr^{-1}$ for each FES in each study site. Since we compare areas of varying size, we also estimated these flows in TEV $ha^{-1}$ $yr^{-1}$ for each FES. As the study sites also show large variation in population (Table 1), we also estimate TEV $inhabitant^{-1}$.

We base our quantification of ecosystem services on existing datasets, e.g. regional statistics on agricultural production and water extraction. This allows for assessment of actually consumed ecosystem services using multi-year means. An alternative method would have been modelling of the supply side processes that generate ecosystem services together with economic modelling of demand to generate estimated quantities of consumed ecosystem services. While modelling allows for more flexibility, we believe using real world statistics increases reliability and transparency. For an overview of all input data for each FES and how they link to final quantification, see S1.

We group FES into either material or immaterial ecosystem services, harkening back to the concept of ecosystem goods and services in Costanza, dArge [29] and Daily [4]. Material FES are tangible goods and energy, extracted from the ecosystem. Immaterial FES are intangible benefits, such as the enjoyment of recreating in nature, the prevention of flooding of property and the mitigation of climate change due to carbon sequestration. We choose this categorisation because it is easy to understand and is clearly linked to different beneficiaries as well as quantification and valuation methods (see Table 2).

For material FES, we quantify the mass of consumed goods. For immaterial FES, quantification is based on the amount of interaction: for recreation that is free to enjoy, we estimate the annual number of recreational trips based on survey data collected for Immerzeel et al. (in review), for carbon we quantify the amount sequestered annually, while for flood reduction we estimate the land area that is annually prevented from flooding due to water retention within the catchment [33].

**Table 2. List of final ecosystem services to quantify.**

| Type | Final ecosystem service | Beneficiary | What to quantify (per year) | Valuation method |
|---|---|---|---|---|
| Material | Supporting environment for crop production | Crop producers | Grains produced | Producer prices with ecosystem contribution coefficients |
| | | | Grass and fodder produced | Producer prices with ecosystem contribution coefficients |
| | | | Other crops produced | Producer prices with ecosystem contribution coefficients |
| Material | Supporting environment for forestry | Foresters | Roundwood removed | Producer prices with ecosystem contribution coefficients |
| Material | Availability of game | Hunters | Hunted big game | Producer prices |
| | | | Hunted small game | Producer prices |
| Material | Availability of peat | Peat extractors | Peat extracted | Producer prices with ecosystem contribution coefficients |
| Material | Potential for hydropower generation | Electricity generators | Electricity generated | Producer prices |
| Material | Availability of berries and mushrooms | Foragers | Berries gathered | Producer prices |
| | | | Mushrooms gathered | Producer prices |
| Material | Availability of water for drinking and processing | Water extractors | Water extracted | Producer prices |
| Immaterial | Recreational possibilities | Recreating visitors | Hunting licenses sold | License prices |
| | | | Fishing licenses sold | License prices |
| | | | Days of inhabitant recreation | Travel cost (Juutinen et al., in prep) |
| | | | Days of national visitor recreation | Travel cost |
| | | | Days of international visitor recreation | Travel cost |
| Immaterial | Mitigated climate change | Global society | Carbon sequestered in biomass | Social cost of carbon [36] |
| | | | Carbon sequestered in lake beds | Social cost of carbon |
| Immaterial | Prevented flood damage | Downstream property owners | Downstream area prevented from flooding | Land values and damage curves [37] |

This table shows for each ecosystem service whether it is a material service (benefit in terms of physical material and energy) or an immaterial service (related to experience or wellbeing), what we quantified and how we valued these quantified services. For detailed quantification methods and sources, see S1 File.

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The final step in quantifying TEV is to convert quantities of consumed FES into monetary value. We use a variety of sources for this, depending on the type of FES. For marketable goods, we base our valuation on mean prices from official statistics for the period 2013–2017 or its closest available equivalent. Here Bateman, Mace [34] recommend to extract from the price all non-ecosystem sources of value in the value chain to end up with the contribution of the ecosystem in the price. We use ecosystem contribution coefficients for this, as described in Vallecillo, La Notte [35]. For agriculture and forestry, we use coefficients per EU country, using Swedish values for Norway as a proxy. For peat production in Finland, we use the same coefficient as for forestry in Finland. For FES that are produced without further human input and only require harvesting, like berries, game and mushrooms, we argue that the price the harvester receives for the good is a close approximation of the economic value of the FES. For non-material FES we estimate recreational value by taking travel cost values as estimated in Juutinen et al (in prep). This is data taken from a survey performed in the same six study sites, where respondents were asked how far they travel from their home to where they recreate most often within the area, and what mode of travel they typically use. This data was used to estimate the number of trips taken, as well as the willingness-to-pay for a recreational trip. For climate change mitigation we use the social cost of carbon [36] and for flooding we use land values and damage curves as described in de Moel and Aerts [37].

The source data underlying our TEV quantification includes spatial datasets (S2). This consists of data on land cover, population density, slope, soil type, stream networks, road networks and biomass productivity. This means we can convert our estimates of TEV into a spatial dataset and link this to these underlying landscape attributes. This allows us to visually analyse how the consumption of ecosystem services varies spatially and how these landscape attributes affect the spatial distribution of TEV. For this purpose, we created a vector data file in ArcMap 10.6, containing square polygons of 1 ha for each of our study sites. Using a set of if-statements (S3) to link our landscape attributes as well as our FES consumption to each separate cell, we distributed our TEV estimates over the study site per hectare.

## 2.4 Explaining variation

By dividing our study sites into subcatchments based on Strahler stream order, we were able to create a dataset of 223 nested spatial units with values on average consumption of specific FES and TEV per hectare as well as average values for various landscape and socio-geographic characteristics. We used linear regression in R (R Stats Package) to estimate correlations within and among catchments between FES and subcatchment characteristics.

We set up four models, each with a separate dependent variable: TEV, value from the supporting environment for crop production, value from the supporting environment for forestry, and value from recreational opportunities. We ran a multiple linear regression model for each predictand, using five landscape and socio-geographic characteristics as explanatory variables: average percentage of clay soil, average slope, average landscape diversity (using the Shannon Diversity Index or SDI on land cover data), average population density and fraction of open water area as part of total land cover. We chose these variables because they cover a wide variety of landscape characteristics: geophysical characteristics (soil and slope), characteristics directly affected by land management (SDI), societal characteristics (population density) and hydrology (surface water).

## 2.5 Stakeholder and conflict analysis

To analyse which groups in society benefit in which study sites, we created four stakeholder types: visitors, landowners, large extractors and global society. We then grouped the different beneficiaries linked to FES into these types (Table 3). This allowed us to analyse how the benefits for each group vary among and within study sites, by defining the main stakeholder group for each hectare cell, i.e. the group benefitting most in monetary terms.

Because different stakeholder groups receive different benefits for different land use types, conflicts may arise when land use changes from one type to the other. To show how possible land use change might impact stakeholder groups, we implemented two basic scenarios in each study site: in the first, all forest within 500m distance of agriculture transforms to agriculture, as long as the soil is not bedrock or moraine. In the second, all agriculture within 500m of forest transforms to forest. For Simojoki, we created two additional scenarios along the same

**Table 3. Stakeholder groups.** This shows per stakeholder group which beneficiary it contains, including the FES connected to that beneficiary.

| Visitors | Landowners | Large extractors | Global society |
|---|---|---|---|
| Hunters Availability of game | Crop producers Supporting environment for crop production | Water extractors Availability of water | Global society Mitigated climate change |
| Foragers Availability of berries and mushrooms | Foresters Supporting environment for forestry | Electricity generators Potential for hydropower generation | |
| Recreating visitors Recreational possibilities | Downstream landowners Prevented flood damage | Peat extractors Availability of peat | |

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principle, but used a shift between forest and peat production areas. We then analysed for each of these scenarios what the impact is on annual TEV for each specific stakeholder group, by switching the average value of the original land use with the average value of the new land use type close to the original land use.

### 2.6 Testing the framework

Boerema, Rebelo [12] suggest the following six criteria for a successful ecosystem services framework (paraphrased):

1. Understand and explain the difference between the supply and demand side of ecosystem services, and be explicit of what you quantify.
2. Take into account the relationships between ecosystem services.
3. Use clear and consistent definitions for ecosystem services.
4. Measure all components that need to be measured for ecosystem service quantification.
5. Use scientifically rigorous and practically applicable measures and indicators.
6. Use scientific rigour in quality control, such as transparency, validity and uncertainty.

In addition, Hanna, Tomscha [2] give recommendations on quantification of riverine ecosystem services, stressing the need for complete quantification of all relevant ecosystem services, their interactions, and their spatial and temporal extent.

Since the above list is mostly qualitative rather than quantitative, we cannot test our data along these criteria statistically. Rather, we choose to discuss our results using previous literature to assess to what extent we comply with the criteria, and where shortcomings might have arisen. To judge our results on criteria 6, we compared our estimates to previous work on similar quantifications, and we performed a sensitivity analysis, showing how our results vary when changing our most uncertain source data values. We applied the following parameter changes:

- A 50% increase of the contribution of the ecosystem to crop production, to compensate for possible underestimation of the ecosystem contribution.
- A 50% reduction of value for carbon, to compensate for possible overestimation of the societal value of mitigating climate change.
- A 50% reduction in travel cost value for recreation, to compensate for possible overestimation of the value of recreational trips.

## 3. Results

### 3.1 Total economic value

Results of TEV estimation show large variation among the six study sites (Fig 2). TEV per study site is highest in Odense, with a total net value of more than 100 million euros per year. The area with the lowest net value estimate is Simojoki, with a TEV of around 20 million euros per year. This is partly caused by low population density and limited agricultural production, but also by low biomass growth rates, limiting the value of carbon sequestration (Table 4). The distribution of value between material and immaterial ES also varies across study sites. In Orrevassdraget by far most of the value is derived from recreational benefits, caused by high population density and high frequency of recreation. In Simojoki by contrast, more than half

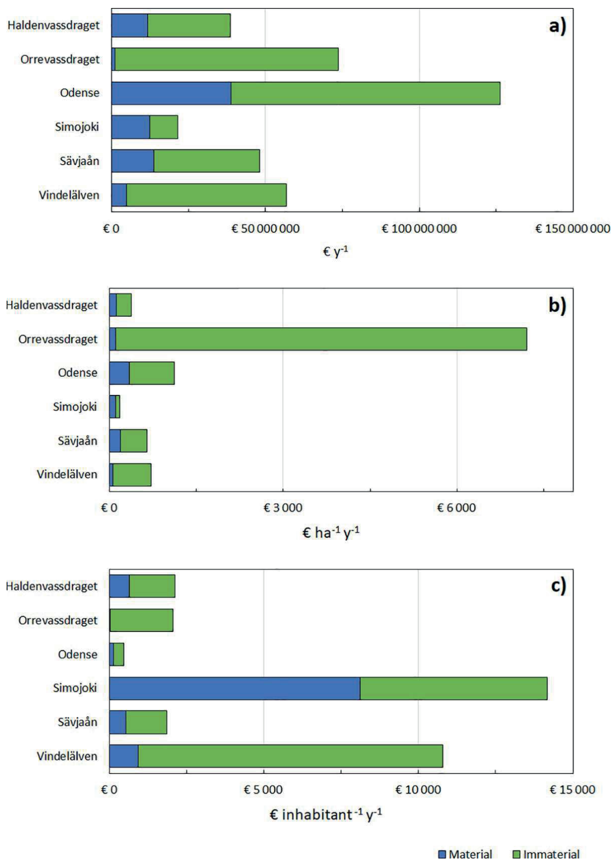


**Fig 2. Total economic value per study site, split out over material and immaterial ecosystem services.** a: The sum of all value consumed from ecosystem services per year in each study site. b: The same values, only divided by study site area in hectares. c: The same values, only divided by study site population.

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of the net value is derived from material benefits. This is caused partly because of low population density, dampening the value of immaterial benefits, but also because of peat production, which generates some of the highest economic benefits per productive area (S1).

When standardising over area, a different picture appears. Orrevassdraget, the smallest catchment of the set but with high population density and a high share of land used for agriculture, yields by far the most value per hectare. This is mostly due to recreation enjoyed by a large number of inhabitants and visitors. Simojoki has the lowest areal net value, with some derived from mostly peat extraction, forestry and recreation along the river, but due to the very low population density and low carbon sequestration, there are few that benefit from the study site.

The negative effect of population is more strongly visible when looking at TEV per inhabitant of the area. Here the least densely populated areas stand out, signifying that there are decreasing marginal benefits over population. Simojoki and Vindelälven, mostly forested areas

**Table 4. Quantified ecosystem services, including corresponding CICES code [9] for reference, and their estimated monetary annual values in € ha$^{-1}$ year$^{-1}$ in each study site.**

| | Haldenvass-draget | Orrevass-draget | Odense | Simojoki | Sävjaån | Vindelälven |
|---|---|---|---|---|---|---|
| M–Agriculture (CICES 1.1.1.1/1.1.3) | 19 (58) | 99 (76) | 217 (333) | 11 (58) | 36 (62) | 6 (29) |
| Grains | 11 | 8 | 80 | 0.12 | 19 | 1 |
| Grazing and fodder | 7 | 87 | 99 | 11 | 13 | 4 |
| Other crops | 0.11 | 4 | 38 | 0.13 | 3 | 2 |
| M—Forestry (CICES 1.1.1.2) | 60 (49) | 9 (34) | 17 (465) | 28 (43) | 130 (107) | 50 (29) |
| Roundwood removal | 60 | 9 | 17 | 28 | 130 | 50 |
| M—Game (CICES 1.1.6.1) | 8 (3) | 0 (0) | 1 (1) | 0 (0) | 2 (1) | 1 (0) |
| Hunted big game | 8 | 0.22 | 1 | 0.21 | 2 | 1 |
| Hunted small game | 0.01 | 0.07 | 1 | 0.14 | 0.01 | 0.00 |
| M—Peat (CICES 1.1.5.2) | 0 (0) | 0 (0) | 0 (0) | 65 (513) | 0 (0) | 0 (0) |
| Milled peat | 0.00 | 0.00 | 0.00 | 65 | 0.00 | 0.00 |
| M—Hydropower (CICES 4.2.1.3) | 4 (0) | 0 (0) | 0 (0) | 0 (0) | 0 (0) | 0 (0) |
| Electricity generated | 4 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| M—Foraging (CICES 1.1.5.1) | 0 (0) | 1 (3) | 0 (0) | 1 (0) | 1 (0) | 2 (1) |
| Berries gathered | 0.15 | 1 | 0.00 | 1 | 0.35 | 1 |
| Mushrooms gathered | 0.22 | 1 | 0.00 | 0.08 | 0.22 | 0.11 |
| M—Water consumption (CICES 4.2.1.1/4.2.2.1) | 26 (1) | 0 (0) | 109 (13) | 1 (0) | 17 (2) | 4 (0) |
| Water from catchment | 26 | 0.00 | 109 | 1 | 17 | 4 |
| I—Recreation (CICES 3.1.1) | 181 (171) | 7 080 (4 544) | 745 (1 748) | 31 (76) | 205 (210) | 512 (345) |
| Value of hunting | 1 | 1 | 5 | 0.06 | 0.23 | 0.13 |
| Value of fishing | 0.26 | 0.00 | 19 | 0.38 | 0.34 | 0.16 |
| Value of recreational trips–inhabitants | 162 | 5 775 | 677 | 21 | 168 | 373 |
| Value of recreational trips–national visitors | 17 | 1 099 | 39 | 9 | 34 | 133 |
| Value of recreational trips–international visitors | 1 | 205 | 6 | 0.00 | 3 | 5 |
| I—Carbon sequestration (CICES 2.2.6.1) | 85 (59) | 2 (4) | 16 (42) | 47 (29) | 260 (174) | 153 (99) |
| Carbon stored in biomass | 82 | 1 | 15 | 47 | 260 | 142 |
| Carbon stored in lakes | 3 | 1 | 1 | 0.33 | 1 | 11 |
| I—Flood prevention (CICES 2.2.1.3) | 1 (7) | 7 (52) | 15 (101) | 0 (1) | 4 (18) | 0 (2) |
| Water prevented from flooding land | 1 | 7 | 15 | 0.04 | 4 | 0.05 |
| **Total** | **384 (210)** | **7 199 (4 561)** | **1 121 (1 810)** | **183 (517)** | **656 (286)** | **728 (375)** |

Standard deviations are given in parentheses behind the main category mean. 'M' stands for material FES, 'I' stands for immaterial FES. Standard deviations are calculated using values per hectare cell. Note that these are means over total catchment area, not over area where the FES is consumed.

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with low population density, shows the most value per inhabitant. This is due to the mix of services consumed: some agriculture, some forest with high consumption of timber, recreation of inhabitants and a relatively large number of visitors from outside the area (Table 4).

A spatial analysis of where the value is generated (Fig 3) shows that on first glance, landscape characteristics, especially river courses, along with population density, seem to have a strong impact on where value is generated. Looking at Haldenvassdraget, the central river valley contains the highest values per area. This corresponds to where agricultural land is located. A network of high value areas is also visible close to water edges, roads and high landscape diversity, since this is where people are most likely to recreate (Immerzeel et al., in prep). This effect is especially visible in Orrevassdraget, where the majority of value is derived from recreational trips. In Odense recreational value derived from the densely populated urban area can be seen radiating out from Odense city, with much of the value generated outside this core

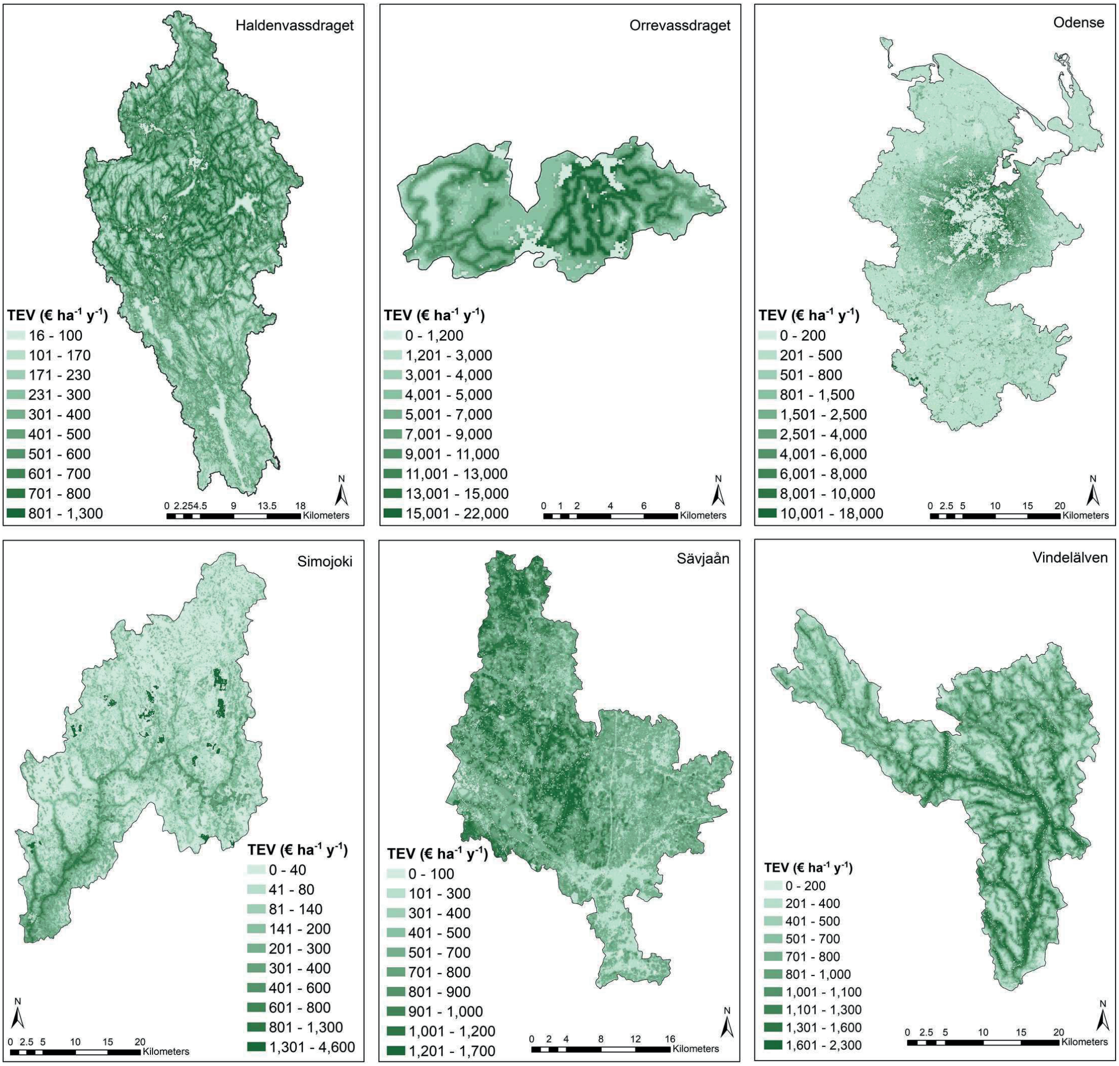


**Fig 3. Total economic value estimates per hectare per year for each study area.** Note the different colour scales. This reduces comparability among study areas, but increases the resolution of values shown within each study area.

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coming from agriculture. This visible radius around the city is also caused by relatively short travel distances in this study site, due to a large proportion of relatively young city inhabitants going on shorter trips, often by bicycle (Immerzeel et al., in prep). In Simojoki two things stand out: the high value of the central river corridor, caused by agriculture along the stream and recreational salmon fishing, and the dark green areas of high value, where peat is extracted. In Sävjaån value is relatively uniformly spread, since forest productivity is high here,

**Table 5. Results of multiple linear regression models on subcatchment level.** Different FES values (top row) are dependent variables, and five study site characteristics (percentage clay soil, average terrain slope, average landscape diversity (SDI), average population in a 5 km radius around the cell and the fraction of water of total land cover in the subcatchment) are independent variables.

| | TEV | Agricultural value | Forestry value | Recreational value |
|---|---|---|---|---|
| | *Coefficient (standard error)* | | | |
| Intercept | 736.33 (216.45)*** | 77.35 (17.08)*** | 13.85 (13.00) | 615.30 (217.80) *** |
| Clay | -344.12 (293.32) | 131.04 (23.15)*** | -6.61 (17.62) | -474.67(295.16) |
| Slope | -4.41 (2.16)** | -1.11(0.17)*** | 0.08 (0.13) | -3.76 (2.18)* |
| SDI | - 323.58 (299.67) | -47.57(23.65)** | 60.30 (18.00)*** | -516.02 (301.54)* |
| Population | 384.32 (56.43)*** | 17.53 (4.45)*** | -7.06 (3.39)** | 372.61(56.78) *** |
| Water fraction | 2653.73(905.12)*** | -284.63(71.43)*** | 93.27 (54.38)* | 2626.63 (910.80)*** |
| N | 223 | 223 | 223 | 223 |
| Adj. $R^2$ | 0.22 | 0.36 | 0.08 | 0.20 |
| F-statistic | 13.36 | 26.23 | 5.08 | 12.09 |
| DoF | 217 | 217 | 217 | 217 |

***, **, * = = > Significance at 1%, 5%, 10% level respectively.

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which increases value from both forestry and carbon sequestration. The western half of the area has a higher average value due to the proximity of Uppsala, increasing the recreational benefits received closer to the city. In Vindelälven the relative weight of recreational value is clearly visible in the contours of water edges and areas close to roads.

### 3.2 Explaining variation

Multiple linear regression using study site characteristics as explanatory variables for variation in the consumption of FES showed a number of significant correlations (Table 5). When looking at total economic value, we found that population density ($p<0.00$) and the fraction of surface water ($p<0.00$) show positive correlations, while the average terrain slope ($p<0.05$) shows a negative correlation. Zooming in on value from supporting environment for growing crops, availability of clay soils ($p<0.00$) and population density ($p<0.00$) show positive correlations to supplied value, while average slope ($p<0.00$), landscape diversity ($p = 0.05$) and the fraction of surface water ($p<0.00$) show negative relationships. Landscape diversity ($p<0.00$) and the fraction of surface water ($p = 0.09$) have a positive correlation to the value of the supporting environment for forestry, while population density ($p = 0.04$) has a negative correlation. For recreation, we found positive correlations with population density ($p<0.00$) and the fraction of water ($p<0.00$) in the subcatchments ($p<0.00$), and negative correlations with average slope ($p = 0.09$) and landscape diversity ($p = 0.09$). None of these models explain the majority of variance though, with a highest $R^2$ of 0.36 for value of the supporting environment for growing crops.

### 3.3 Stakeholder and conflict analysis

When taking the spatial variation of TEV consumption and zooming in on the distribution among the main stakeholder groups, some spatial patterns emerge, both within and among study sites (Fig 4). In Haldenvassdraget, Simojoki and Sävjaån, landscape characteristics are clearly visible: the main river valley with its fertile soil and low slope gradient appear as areas where landowners are the dominant stakeholder group. Additionally, in Simojoki, large areas along the main river have recreating visitors as the main stakeholder, since salmon fishing is one of the main recreational attractions in the area.

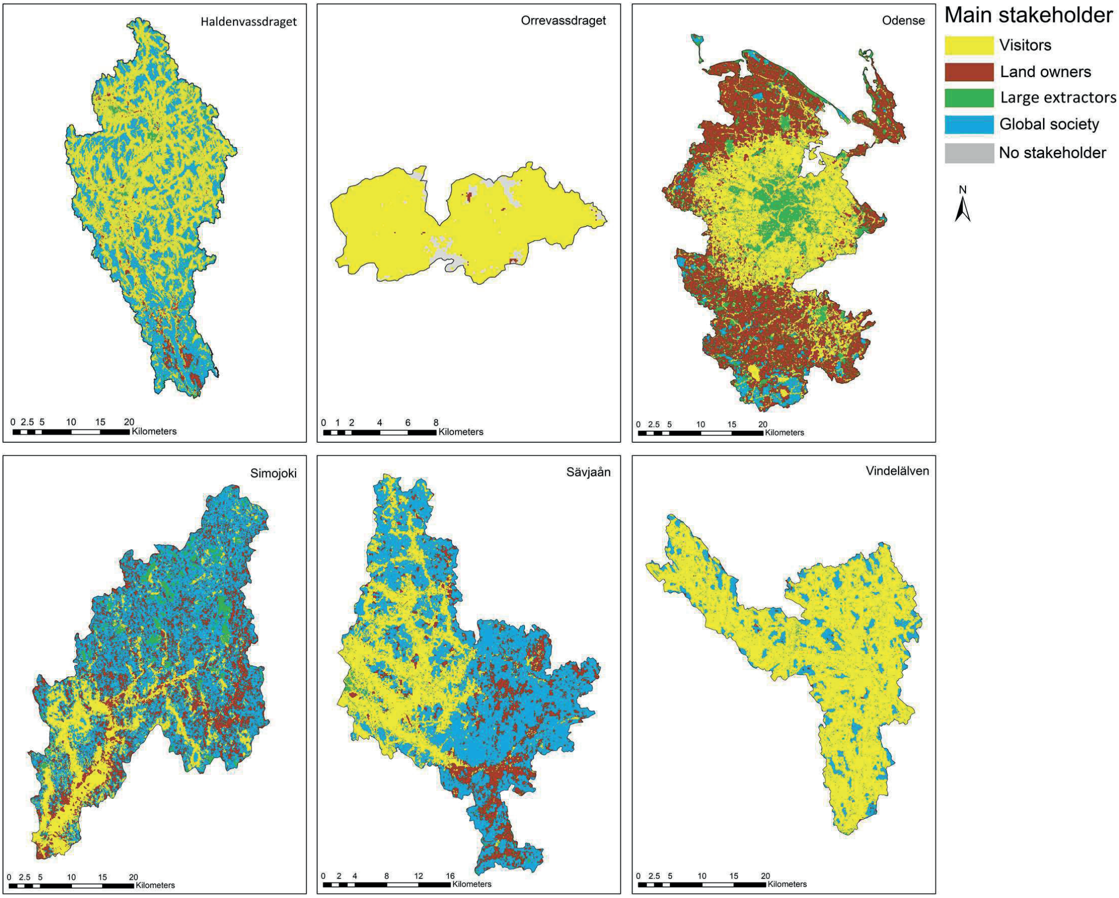


**Fig 4. Main stakeholder per hectare cell.** This shows per study area how FES consumption is spatially distributed by showing the stakeholder group with the highest TEV per cell.

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Besides landscape characteristics, population density also shows clear effects on the spatial distribution of dominant stakeholders. In Odense, a clear radius can be seen around the city, illustrating that recreational visitors are dominant close to the city, while further away the weight shifts to landowners benefiting from the supporting environment for crop production. Large extractors dominate only where peat extraction occurs, or in urban areas where drinking water extraction is the main FES. Global society is the main stakeholder in the more remote areas in each study site. This also means that in densely populated areas like Orrevassdraget or Odense, global society is the main stakeholder in only a few small areas.

Moving on to possible conflicts, the effects of land use change on groups of stakeholders vary among study sites (Fig 5). When forested areas are transformed into agricultural land, the net effect varies among study sites. Global society loses everywhere, though in the northernmost sites, Simojoki and Vindelälven, this effect is negated by gains from other groups,

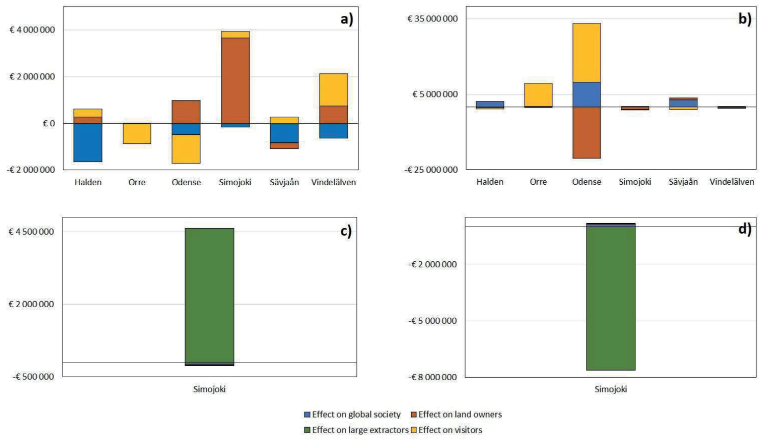


**Fig 5. Effects of land use change on stakeholder TEV.** a: shift from forest to agriculture. b: shift of agriculture to forest. c: shift from forest to peat extraction. d: shift from peat extraction to forest (see Methods section for details). For each of the scenarios, the total study site effect for each stakeholder group is presented as TEV in € $ha^{-1}$ $year^{-1}$.

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since carbon sequestration is lower in these sites. Additionally, in areas already dominated by agriculture, the marginal value of forest for recreation is higher than in more mixed landscapes (Immerzeel et al., in prep), so there the value for visitors is also decreased. In Orrevassdraget and Odense this even leads to a net negative effect on TEV.

When moving in the opposite direction, converting agriculture to forest, the opposite effect on recreational value is apparent. Odense stands out here with a strong negative effect for landowners, though this is offset by gains for both recreating visitors and global society. This is caused by the large number of scattered forest areas in the area: when all these expand, an area of around 73,000 hectares is transformed to forest, greatly reducing agricultural production in the catchment.

The results of this exercise also suggest that eliminating peat extraction in Simojoki will create a net loss to society of close to 8 million euros per year, due to the loss of value from extracted peat for large extractors. However, visitors, landowners and global society would all benefit.

## 3.4 Validity and partial sensitivity analyses

When comparing our results to previous studies with comparable study sites and methods, we find comparable results. Vermaat, Wagtendonk [38] use a similar method, estimating TEV in € $ha^{-1}$ $year^{-1}$ in six European river corridors before and after river restoration, based on market prices and stated preference studies. They estimate a mean TEV of €500 $ha^{-1}$ $year^{-1}$ before restoration for their Finnish site, mostly from crop production, and the southern Scandinavian sites closer to €1,000 $ha^{-1}$ $year^{-1}$, from a broader mix of FES. Remme, Edens [39] estimated value in € $ha^{-1}$ $year^{-1}$ for a province in the Netherlands for agriculture, drinking water, air quality regulation, carbon sequestration and recreation, finding a TEV of around €500 $ha^{-1}$ $year^{-1}$, mostly coming from agriculture and recreation. Lankia, Kopperoinen [40] estimated the value of recreation in various Finnish regions using survey data. The Simojoki area, for which we estimated a mean recreational value of €37 $ha^{-1}$ $year^{-1}$, is on the border of Lapland and

Northern Ostrobothnia, which Lankia, Kopperoinen [40] estimated to deliver recreational value of €15 and €58 ha$^{-1}$ year$^{-1}$ respectively. When looking at the highly productive agriculture in Odense, Lehmann, Smith [41] focused on the value of ecosystem services from agroforestry systems and found an average gross margin for agricultural production in Denmark of €1,067 ha$^{-1}$ year$^{-1}$, compared to our €217. They do not split out the ecosystem contribution to the gross margin, however. Our estimate for the ecosystem contribution is based on average producer prices of €930, close to their estimate. Nikodinoska, Paletto [42] estimated the value of various ES from the region around Uppsala, which also contains our Sävjaån area, using market prices, carbon permit prices and survey data. They found mean TEVs of around €1,200 ha$^{-1}$ year$^{-1}$ from forest areas and €600 from agricultural areas, compared to our combined average of around €650, based on an average producer price of around €750.

Sensitivity analyses on three of the more uncertain underlying variables show that doubling the value of the ecosystem contribution to crop production, and halving the value of carbon sequestration or recreational trips does not change the ranking of catchment TEV (Fig 6). However, effects of the changes vary among catchments. An increase of the contribution to crop production shows a particularly strong effect in Odense, increasing TEV by almost 20% due to its dependence on agriculture. Halving the value of carbon sequestration (presuming an overestimation of the societal value of mitigating climate change) mostly affects Sävjaån, with a reduction of 20% of TEV, due to the relatively high biomass productivity in that catchment. Reducing the value of recreational trips (presuming an overestimation of travel cost value) has a particularly strong effect in Orrevassdraget, with a reduction in TEV of 49%.

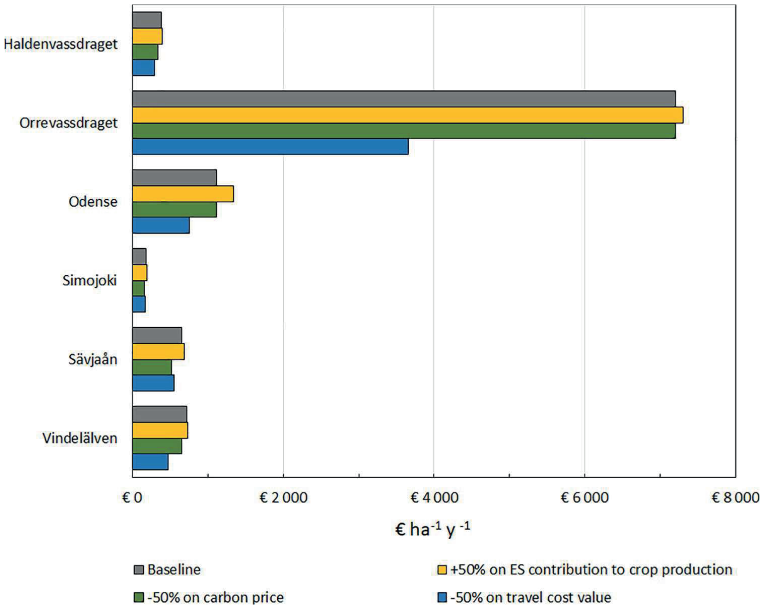


**Fig 6. Sensitivity analysis on three variables.** For each study site, TEV in € ha$^{-1}$ year$^{-1}$ is shown for the baseline, as well as for a change in three underlying variables.

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# 4 Discussion

## 4.1 Interpretation of results

Our estimates show that catchments in Northern Europe provide society with value from a diverse source of ecosystem services. One of the main sources of value in our study sites is recreational value, being the main FES in Orrevassdraget, the area with the highest average value per hectare, as well as in Vindelälven, where that value is especially high due to the high travel cost per trip (Juutinen et al., in prep.). In all other study sites, it is also one of the FES with highest value. Since most of the recreation is enjoyed by local visitors from within the study site, one would assume a positive correlation between population density and value of recreation, and that is indeed what we found in our linear regression. The relationship between population density and ecosystem service value is in line with findings in other studies, for instance Vermaat, Wagtendonk [38] and Brander, Wagtendonk [43]. We also found a positive correlation between the fraction of surface water in an area and recreational value, supporting previous findings that water provides significant recreational value, for instance in Grizzetti, Liquete [44]. This also suggests possible conflicts in future land management change: agriculture, forestry and peat extraction can have a significant negative impact on water quality [45,46], and surveys in our study sites have shown that preference for recreation in these areas depends strongly on good water quality (Immerzeel et al., in prep.). Future studies on land use change should take these interactions into account. In Europe the Water Framework Directive, the EU directive that commits member states to achieve good qualitative and quantitative water state, also places demands on water management that need to be considered [47]. The negative relationship between terrain slope and recreational value likely has to do with access: roads and rivers, where recreation concentrates, tend to lie in relatively flat areas. Another major ecosystem service is the ability of the ecosystem to support agricultural production. Again, there is strong variation among our study sites, as well as within them. The availability of clay soil had a strong positive effect on value produced by agriculture. This reaffirms the notion that a combination of natural and societal characteristics affects TEV, emphasizing that value is created in the interaction. Besides soil and population, we found a significant negative effect of landscape diversity and slope on agricultural value. This is supported by previous work [48,49], showing that most agricultural value is created in relatively flat areas with uniform land use. For forestry we found few correlations between our selected study site characteristics and supplied value. This suggests that forestry is located where other land use is unprofitable or physically impossible, a 'leftover' land use [50].

The analysis of TEV and its variability additionally suggests that the effects of land management on the value of supplied ecosystem services might vary significantly, depending on the characteristics of the study site. This is also illustrated in our analysis of stakeholders and conflict. Changing land use in a similar manner can have a profoundly different effect on the benefits that society receives from ecosystems, depending on where this land use change takes place. For instance, landscape diversity appears to have a significant positive effect on recreational value. When changing from agriculture to forest, our analysis suggests that the effect on recreational value is positive in an agricultural area, but negative in an area that is already mostly forested. Alternatively, when transforming forest to agriculture, the benefits to land owners can be very high in an area where forest grows slowly, as in the northern Simojoki study site, but negative in an area like Sävjaån, where forest productivity is much higher and the value of FES consumed through forestry are higher than through agriculture.

These differences, along with large differences in share of benefit received by different stakeholder groups, indicate that conflict due to land use change is likely to occur when land management does not take them into account. Lee, Markowitz [51] have shown that perception of local temperature change and understanding of anthropogenic effects on climate are strong predictors of concern for climate change, making it likely that public concern for climate change will increase over time. This can create conflict in study areas where global society would benefit from a transformation to forested area, whereas landowners would benefit from the opposite. Since our method only takes into account benefits from processes within the ecosystem, we do not include the effects of what happens with goods after extraction from the ecosystem. Therefore, we do not quantify carbon emissions from burning firewood or peat, or any change along the value chain towards consumers. However, with increasing public perception, pressure can increase for ceasing peat extraction altogether, and indeed, in Finland the public debate on what to do with this industry is increasingly volatile due to environmental concerns [52]. In the more densely populated areas, visitors and landowners also have competing interests, where visitors (and businesses dependent on them) appear to prefer more forest, while landowners are receiving benefits from maintaining the area for agricultural production.

## 4.2 Testing our hypotheses

Based on our findings, we partially dismiss the hypothesis that dominance of primary sectors in land use translates into dominance of FES delivered by these sectors. Forestry dominated catchments such as Haldenvassdraget and Vindelälven generate most of their value from the non-material FES recreation and carbon sequestration. In fact, material benefits only dominate in Simojoki, an area with very low population density. In areas with higher population density, such as Orrevassdraget, immaterial FES dominate, even in areas that are mostly covered by agricultural land use. This reinforces the notion of farmers serving as landscape stewards; the management of their lands serving not only their private interests, but a broader value of the landscape, as reflected in for instance the European Union's Common Agricultural Policy [53] and Norwegian subsidies to farmers for buffer strips along streams [54]. The spatial shift in dominance from material to immaterial services linked to population density can be most clearly seen in Odense (Fig 4).

This ties into the second hypothesis: where population density is low, immaterial FES tend to make a smaller contribution. Previous studies have found a positive relationship between population density and consumption of immaterial FES [38,55]. We also found that high population density in the vicinity of a cell increases recreational value (Table 4).

The third hypothesis, that recreational value is strongly linked to the availability of water, is clearly supported by our data. We found a significant positive relationship between the fraction of surface water in a subcatchment and the value of recreation generated in that subcatchment (Table 4).

## 4.3 Methodological limitations, uncertainty, and their implications

Using TEV has limitations, and there are alternative estimators of value. For example, TEV needs ecosystem services to be both quantifiable and of measurable value. Turner, Paavola [56] claim that TEV does not necessarily equate to a total system value, since it excludes the value of the system working as a whole, which they claim is more than the sum of its parts but cannot be measured in economic terms. There are also opponents of economic valuation of nature on principal, arguing that there is no objective measure of the value of an ecosystem, and that economic terminology is unsuitable for describing our relationship to nature [57]. While acknowledging these criticisms, we use TEV because it provides a transparent way of making quantitative estimates. This transparency gives unique power because it allows for comparative analysis, and monetary value works as a clear communicative method because it is easily understood and can be compared to costs and benefits of other societal activities.

The results of this study are based on a broad variety of sources. For quantification of FES consumption, we prioritised data with high detail and accuracy, which were national and regional statistics. This implies that we used separate sources for our study sites when crossing administrative boundaries, with different categorisations and data collection methods. When converting quantities to values, we again relied on a broad variety of sources to maximise precision (Table 2). Each of these however comes with its own range of uncertainty, and as Brander, Florax [58] and Schild, Vermaat [59] show, choice of valuation method can have a significant impact on value estimations. Therefore, compiling values based on different sources increases uncertainty in aggregate TEV estimates.

Another source of uncertainty is in the quantification method of marketed goods. The value of the FES is not in the value of the product, but in the part of that value generated by the ecosystem. We use ecosystem contribution coefficients from Vallecillo, La Notte [35]. However, these are country wide averages and not specified to our study sites. Additionally, Norway was not included in their study, nor was peat extraction, which we quantified by transferring their estimates from other values.

A third source of uncertainty is in the spatial analysis. Typically, spatial analyses of ecosystem service supply or demand use data from a single source for each variable. For land cover data for instance, CORINE data is often used, which allows for consistent comparison of land cover across European study sites [22]. However, these international datasets are typically generalised and of relatively low resolution. Due to the resolution we needed for our analyses on underlying drivers and stakeholders, we decided to use local datasets containing more detail and higher spatial resolution. This meant we compared outputs from datasets with different underlying methodologies and varying spatial resolution, which potentially increases uncertainty when comparing among different study sites. However, we argue this choice is worthwhile because it allows us the spatial resolution necessary to identify patterns, without claiming to know on a less than hectare level resolution what quantities of FES are supplied where.

In following the recommendations as given in Boerema, Rebelo [12], we argue we have succeeded or partially succeeded on all six recommendations (Table 6). When comparing our estimates to previous studies using similar methods in similar study sites, we find comparable results, which strengthens the argument that our estimates have sufficient validity according to recommendation six. However, since a full meta-analysis is beyond the scope of this study, we limited ourselves to five recent studies in North-western Europe. A basic sensitivity analysis on three of the underlying variables shows that sensitivity is low in general, but higher in study sites depending highly on a single FES, such as the large share of recreation in TEV in Odense. However, the general trends do not change, even when halving the strength of these variables.

## 4.4 Further research

One avenue of further research is to streamline the method we used for wider application. The current set-up is based on a broad variety of sources, making data collection labour intensive. A version based on international datasets, with a more user-friendly interface and quantification method, could possibly allow for large scale international comparisons that are relatively easy to implement and analyse.

A second direction for future research is to implement the dynamics of ecosystem processes in more detail. FES depend on a broad variety of ecological and environmental processes that are not currently included in our method, but are of importance because of interactions between human activity and these processes [60]. Nutrient retention and carbon cycling for instance affect multiple FES, but human activity conversely affects these processes as well [61].

**Table 6. Criteria for a framework of ecosystem services quantification based on Boerema, Rebelo [12], comments on the performance and success level in following the criteria for our framework.**

| Criterion | Comment | Success |
|---|---|---|
| Understand and explain the difference between the supply and demand side of ES, and be explicit of what you quantify. | We focused on the benefits society receives from interacting with ecosystems and we have explicitly tailored our framework to this, by incorporating a valuation step in monetary terms. By using FES, which focuses specifically on the point of interaction between ecosystem and beneficiary, we also choose to leave out the complexity of how ecosystem processes lead to a benefit. | Succeeded |
| Take into account the relationship between ES. | Since FES only quantify the value of the interaction between a specific stakeholder and the ecosystem, relationships between ES are minimised. Underlying processes within an ecosystem might impact several different FES and in that sense they are related, but because FES are focused on the process end point on the ecosystem side, we argue double counting is minimised: because every FES is explicitly linked to a stakeholder, there is no direct interaction. The only interaction is in changing the underlying base that supplies the FES, for instance when changing land use, and this we have taken into account in our stakeholder and conflict analysis. | Succeeded |
| Use clear and consistent definitions for ES. | In paragraph 2.2 we gave a definition of FES based on previous literature we believe to be clear and consistent, and that we adhered to throughout our data collection and analysis. | Succeeded |
| Measure all components that need to be measured for ES quantification. | It is likely that for some FES, such as the supporting environment for crop production, we measure more than the actual FES contribution. For other FES, such as climate change mitigation, we only take into account carbon sequestered into biomass and aquatic sediments, not those sequestered into soils. Since the majority of carbon fluxes is in biomass and not in soils [61], we argue this does not have a strong impact on our results. | Partially succeeded |
| Use scientifically rigorous and practically applicable measures and indicators. | Our measures and indicators are grounded in the FES definition and our quantification shows that they are practically applicable. Some of the indicators are proxies, but these are incidental, such as the contribution of the ecosystem to the producer price for extracted peat. | Partially succeeded |
| Use scientific rigour in quality control, such as transparency, validity and uncertainty. | We documented all sources and steps in the quantification process. These are available in S1. We compared our estimates to estimates from previous research using similar methodology on similar study sites, and found them to be within the same range of values previously reported. A basic sensitivity analysis showed how changes in value for our more uncertain inputs can affect the results. | Partially succeeded |

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This affects immaterial value from recreation as well: Milner, van Beest [62] for instance show that the type of forest management impacts moose populations, which in the Nordic countries can have a strong effect on recreational value from hunting as well as the provision of game meat.

A third topic of further framework development is to apply scenario analysis. This builds upon the addition of interaction between ecosystem processes and FES consumption: quantified scenarios where for instance nutrient inputs from agricultural areas change, along with other environmental and societal variables can show the effects of land management change on the consumption of FES. This can also include more specific policy recommendations, showing which choices might affect stakeholders in different ways, as well as which instruments can be effective in reaching different stakeholder groups.

## 5 Concluding remarks

We have found that societal value from our six study sites is highly variable and derived from a variety of sources. Recreation, carbon sequestration and the supporting environment for agriculture tend to yield the largest benefit, though this is strongly dependent on study site characteristics. We have found a variety of environmental and socio-geographic characteristics covarying with FES value. Population density appears to be one of the key drivers for the most valuable FES, further strengthening the notion that it is in the direct interaction between people and nature that most value is created. Access to water is another key ecosystem characteristic driving FES value. We observe that different stakeholder groups value specific types of landscape differently, implying that land use change can lead to conflict. We show that global society, benefiting from climate change mitigation, can suffer if landowners or large extractors choose to increase their direct revenues by changing how they manage their land. Visitors aiming to enjoy the landscape can either suffer or benefit from a similar change, depending on the pre-existing state of the landscape.

We believe this application of the FES framework shows that a rigorous, consistent quantification of ecosystem services in varied landscapes across different countries is possible, and gives insight into what drives the variation in generated value. We believe it can be of value to decision makers by showing how different societal stakeholder groups benefit and may conflict, driving home the point that decision making should be tailored to local circumstances. Further research should focus on refining the toolset, a further integration of ecosystem processes underlying FES generation, and on scenario analysis for future land management, to aid in ensuring a sustainable and mutually beneficial relationship between society and nature.

## Supporting information

**S1 File. Truncated quantification spreadsheet.**
(XLSX)

**S2 File. Overview of spatial datasets.**
(DOCX)

**S3 File. Method description for spatial quantification.**
(DOCX)

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# Paper III

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

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**Abstract**

Policy makers in Nordic countries envisage a developing bioeconomy as an important element in the transition towards a fossil-energy-free future. However, the shape of the implementation of such a bioeconomy is yet unclear. Therefore, a set of five common scenarios has been developed previously from existing benchmark scenarios of societal change in 2050, the Nordic Bioeconomy Pathways (NBPs), labelled, respectively: sustainability first, conventional first, self-sufficiency first, city first and economy first. In the current paper, we adapted an existing integrating framework of ecosystem service delivery for these NBPs and estimated economic value of all final services for six study catchments across the Nordic countries: Odense (DK), Simojoki (FI), Haldenvassdraget (NO), Orrevassdraget (NO), Sävjaån (SE) and Vindelälven (SE). We converted scenario storylines to a set of numerical attributes in a consultation process with stakeholder representatives and local experts. These numerical attributes quantify animal husbandry, crop production, forestry, human population, urbanization, recreation habits, economic growth, energy use and sources, water resources management focus and land cover distribution. For each scenario, we estimated proportional changes in these attributes and coupled them with empirical or logical link equations to annual ecosystem services generation in biophysical and monetary units. We then performed spatial analysis to estimate changes in spatial variation of ecosystem services generation within catchments. Modelling outcomes suggest the following: the value of active nature appreciation, such as outdoor recreation, increases more in 'sustainability' and 'economy first' than in the other scenarios; changes in total economic value vary most among catchments under 'cities first'; the mutual ranking of ecosystem services value within catchments largely remains unchanged under all NBPs. From our analyses we conclude that outdoor recreation, both by locals and tourists, is contributing a high and likely undervalued benefit to society, and it appears highly sensitive to how a future bioeconomy will develop. Scenario-related differences in the effects of changes in agriculture and forestry exist but appear to be less prominent. Overall, both the total estimated value delivered in these catchments and the distribution over different services are highest under 'sustainability' and 'economy first'.

Keywords: Final ecosystem services; total economic value; spatial analysis; stakeholder analysis; catchments.

# 1. Introduction

Societies affect and depend on their surrounding natural environments for their energy and material needs, as well as their immaterial needs (Balmford et al. 2002, Barton et al. 2012, Kettunen et al. 2012). With the explosive growth in human population and resource use in the past centuries came increasing pressures on ecosystems, leading to environmental degradation that in turn affects human wellbeing (Lant et al. 2008, Raudsepp-Hearne et al. 2010, de Groot et al. 2012). However, in recent decades governments have started to become aware of the risks posed by our dependency on ecosystems combined with the degrading effects of our interactions with them (Kettunen et al. 2012, Bateman et al. 2013, Hauck et al. 2013). The destabilizing effect of climate change specifically has become one of the most pressing points on the global policy agenda (Lee et al. 2015, Bouman et al. 2020), and a key part of addressing the issue is to transition society from one with fossil fuel at the core of our material and energy supply, to one based on renewable, biological resources (Belling 2017, Hetemäki and Muys 2017).

The transition to such a bioeconomy would drastically alter our relationship to ecosystems. In 2019, the global economy consumed around 140,000 TWh of fossil fuel based energy[1]. In 2015 it produced more than 380 million tonnes of plastics[2], mostly created from fossil resources, and projections up to 2050 indicate that global energy consumption will grow by 50%[3]. Replacing these fossil flows with flows of renewable, biological resources requires a massive increase in resource extraction from ecosystems. Timber and fiber materials can replace plastics, energy crops and woody products can replace fossil fuels, electricity from hydropower and other renewable sources can replace coal, gas and oil (Belling 2017). With a transformation of our resource use, pressures on ecosystems will change. In the meantime, society also changes: power structures and cultural norms and values affect how we manage land use (O'Neill et al. 2017, Rakovic et al. 2020). Our resource use may continue to grow, or start to decrease, with our rising awareness of our impact on the natural environment. The uncertainty around what such a transition will look like over the next 30 years, as well as how that change will affect ecosystems is the driving force behind this study.

Rooted in the growing concern for environmental degradation, The Nordic Council of Ministers has set the goal of transitioning to a bioeconomy (Gíslason and Bragadóttir 2017). This decision can have far-reaching effects on land use in the Nordic countries, but it does not come alone. The pressure of fertilization practices in agriculture on the surrounding terrestrial and aquatic ecosystems has been a major issue in Nordic environmental policy since the 1970s (Øygarden et al. 2014, Tanzer et al. 2021), when the scale of environmental degradation in the Baltic and North Sea caused by nutrient runoff became apparent. Efforts to curb application of nitrogen and phosphorus, as well as measures to reduce runoff from agricultural areas have had significant impact on the fluxes of nutrients moving through ecosystems

[1] https://ourworldindata.org/fossil-fuels
[2] https://ourworldindata.org/grapher/global-plastics-production
[3] https://www.eia.gov/todayinenergy/detail.php?id=41433

(Tanzer et al. 2021). However, transforming the entire energy system on an international level requires even more significant change. Replacing 140,000 TWh of fossil fuel use with renewable resources demands society to draw from a variety of sources, and will likely lead to both an increase in land use for biomass generation for harvest, as well as an increase in productivity. However, constraints are posed by the pressures of biomass generation on the surrounding ecosystems, including the requirements of the Water Framework Directive on water quality and ecology (WFD 2000), as well as by physical limitations on land use. In Norway for instance, potential areas for afforestation or conversion to cropland are limited by climate and geology: Granhus et al. (2012) for instance estimate that about 175,000 hectares of coastal heathland can be potentially afforested, which would be an increase of only around 2.5% of current forest, suggesting that cropland and forest already share most of the potentially productive land between them.

While there is a consensus on the desired trend, uncertainty regarding future political, technological and societal developments make it impossible to clearly define a model of what a bioeconomy will look like. The shape of the bioeconomy partly depends on the effects of changing land use on ecosystems, and thereby on the benefits we get from ecosystems. Society depends on ecosystems for a wide range of benefits, and can therefore be strongly affected when ecosystems change due to a transition to a bioeconomy (Belling 2017, Heinonen et al. 2018, Vermaat et al. 2020). These benefits are often quantified as ecosystem services. The concept of ecosystem services has a history in research and policy making as a means to quantify the value of ecosystems to society (Costanza et al. 1997, de Groot et al. 2002, Bateman et al. 2011), and are often defined as 'the benefits society receives from ecosystems'. By altering our land management to accommodate a bioeconomy, we also alter ecosystems, such as forests, agroecosystems and aquatic ecosystems, thereby potentially altering the value of ecosystem services that are generated there. Taken to an extreme, one can imagine a rural landscape transformed into an intensively managed carpet of crops and fast-growing coppice woodland, reducing the aesthetical value of the landscape, increasing erosion and flood risk because old growth forest that previously retained water and soil have disappeared, and eutrophication in rivers and lakes, reducing opportunities for water recreation and production of drinking water. Because the generation of ecosystem services depends on a complex web of ecosystem processes linked to human interactions, quantifying the effects of land use change on total value of all ecosystem services is not straightforward. Nevertheless, the effects of land use change on ecosystem services have been estimated before. Examples are river and wetland restoration projects (Boerema et al. 2014, Vermaat et al. 2016, Brouwer and Sheremet 2017), but comparative studies on an international level of the effects of bioeconomy development on ecosystem services have to our knowledge not yet been performed. Such an analysis could contribute to designing land management plans that approximate a societal optimum, because the effects of land use change on ecosystem services supply can vary according to where land use change occurs (Immerzeel et al. 2021).

In this study, we quantify the possible effects of transitioning to a bioeconomy on the value of ecosystem services generated in six Nordic catchments. We do this using a scenario analysis, considering five possible scenarios of bioeconomy development in the Nordics as described in Rakovic et al (2020). These scenario's, called Nordic Bioeconomy Pathways (NBPs), are based on the Shared Socioeconomic Pathways (O'Neill et al. 2014), specifying their effects on society and land management in the context of bioeconomy in the Nordic countries. We quantify the effects of these scenarios on the current state of ecosystem services generation as described in Immerzeel et al. (2021).

There are spatial components to the delivery of ecosystem services (Brander and Koetse 2011). For example, where agriculture is possible depends on physical characteristics such as slope and soil type, recreational value depends on type of land use, openness of the landscape and access to water and roads, and flood damage is affected by slope and land use. This means that a bioeconomy transition can have varying effects on total economic value of ecosystem services depending on location. This can impact how value is distributed among stakeholder groups, since landowners are limited to where they can harvest crops or timber, and recreational visitors are limited by access, so to analyse the effects on different groups, we performed a spatial analysis of the generation of ecosystem services in our study areas under the different NBPs.

In doing so, we aim to answer the following research questions:

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

To structure our inquiry, we formulate the following hypotheses:

1. The NBPs change the distribution of generated value over the different ecosystem services.
2. The effects of the NBPs differ in rural areas compared to peri-urban areas.
3. Different stakeholder groups are affected differently by the NBPs.

# 2. Method

## 2.1 Study areas

We used the same catchments as study areas as those in Immerzeel et al. (2021) (Table 1, Figure 1). In that paper, the current total economic value (TEV) of all generated ecosystem services was quantified. Basing ourselves on the same catchments and using the same methods made sure that comparison is possible among the current situation and future bioeconomy scenarios, as well as among different Nordic areas. For further detail on these areas as well as on the estimation methods of current TEV of ecosystem services, see Immerzeel et al. (2021).

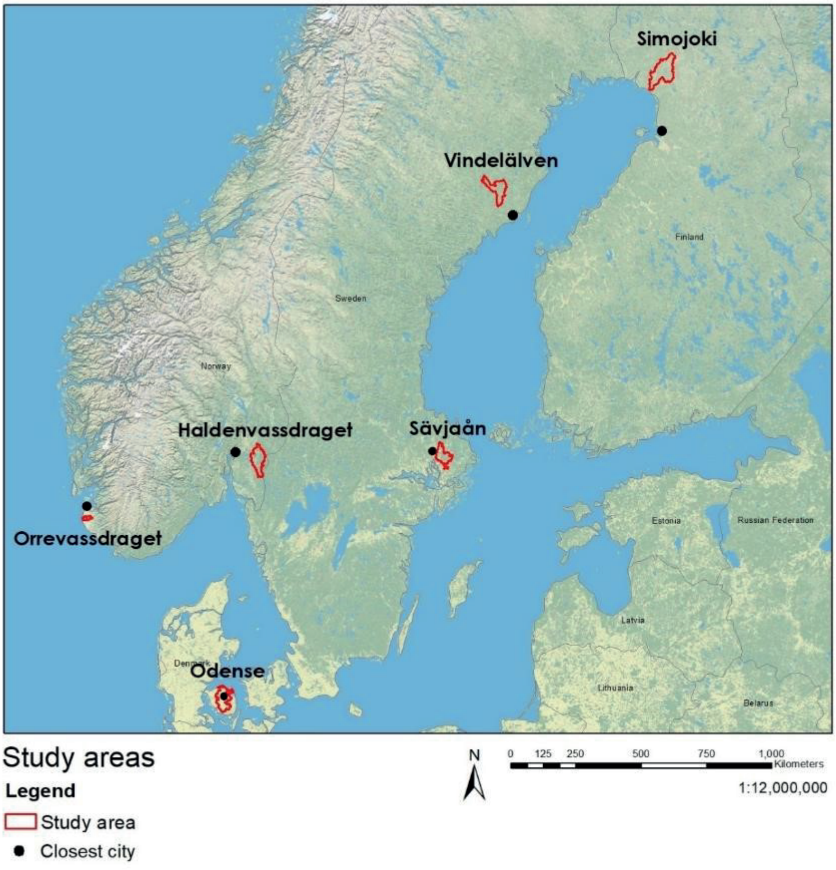


**Figure 1. A map showing the positions of the study areas across the Nordic countries.** The basemap is provided by ESRI[a]. Study area boundaries are shown in red. Black dots show the city closest to the catchment as described in Table 1. This map illustrates the spatial range of study areas across 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 a relatively large proportion of agricultural land, while Haldenvassdraget, Vindelälven and Simojoki are further from densely populated areas and contain relatively little agricultural land.

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

**Table 1. Study area descriptions showing 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.

| | **Halden-vassdraget*** | **Orre-vassdraget** | **Odense** | **Simojoki#** | **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) |

* Northern part

# Western part

## 2.2 Nordic Bioeconomy Pathways

As a basis for what the bioeconomy could look like, we use the NBPs as described in (Rakovic et al. 2020). These are five qualitative storylines, describing different states of bioeconomy in 2050 in Denmark, Finland, Norway and Sweden (Table 2). The general trends are aligned with the Shared Socioeconomic Pathways (SSPs), making our projections compatible and comparable with studies based on these SSPs (O'Neill et al. 2017, Popp et al. 2017, Riahi et al. 2017).

**Table 2. Summary of the NBP storylines.** This gives a short qualitative summary of each NBP storyline. For more detailed descriptions, see Rakovic et al. (2020).

| NBP name | Summary of storyline |
|---|---|
| NBP1: Sustainability first | Societies around the world increasingly recognize the environmental, social and economic costs of disconnected, resource intensive production and consumption patterns. The development thus shifts to a more sustainable path, which respects perceived environmental boundaries and places human well-being ahead of economic growth. The changes in energy systems are directed towards renewables and high resource efficiency, coupled with consideration of the environmental footprint from the cradle to the grave. Along with the low resource intensive lifestyles, this leads to a low overall energy use. |
| NBP2: Conventional first | This world follows typical recent historical patterns with uneven development and income growth. There is a concern for local pollutants but moderate success in policy implementation and slow progress in achieving the sustainable development goals. In the Nordic energy sector, some investments in renewable energy systems are made but society continues to rely on fossil fuels. Bioenergy is a relatively low share of total energy use although there are some investments in novel technology. |
| 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. International trade is strongly constrained and policies are oriented towards security, while there is low priority for environmental issues. The importance of developing the Nordic bioeconomy therefore becomes a matter of regional security, placing self-sufficiency aims high up on the agenda. |
| NBP4: City first | In a world with 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. Environmental policies are centered on local concerns with little attention to vulnerable areas or global issues. In the Nordic countries, segregation between societies in overlooked residential areas and more valued prosperous regions continues to lower societal cohesion. |

| | |
|---|---|
| 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. Regarding environmental policy, there is a focus on local issues with obvious benefits to human wellbeing, whereas global issues receive little attention. In this society, lifestyles are material intensive and diets are meat rich. The energy and resource use intensity is high and there is a heavy reliance on fossil resources. |

In Rakovic et al. (2020), these NBPs are then further defined using 'NBP elements', which describe subsets of society that will be altered by the bioeconomy. Examples are 'Economic growth', 'Social equity' and 'Crop production'. Since these are qualitative elements, the first step we took was to transform these into quantitative variables, so that we could link them to attributes of society and landscape that determine the flow of ecosystem services. We called these quantified variables sub-elements (Table 3). We took into account three key requirements when creating these:

1. Affected by the transition to a bioeconomy as defined by the NBP elements;
2. Quantifiable using scientific literature and publicly available datasets;
3. Connected to catchment attributes that impact ecosystem services flow.

For each sub-element we first quantified its value in the current situation for each of our six study areas. We then set a new value for each of these for every NBP. We based these values on expected trends until 2050 (such as local population projections), previous studies on feasible development and maximum sustainable production over time (Popp et al. 2017, Riahi et al. 2017, Rakovic et al. 2020, Trömborg et al. 2020), crop yields from the EU agricultural outlook (EC 2020), as well as workshops and interviews with experts on each catchment. See Supplement 1 for details on the sub-element levels and their sources.

**Table 3. NBP elements and sub-elements.** This lists all NBP elements considered in this study, based on Rakovic et al (2020), as well as the quantifiable sub-elements and their units. All flows are quantified as annual.

| NBP element | Sub-element | Unit of study |
|---|---|---|
| Animal husbandry | Total livestock | Animals per study area |
| | Grazing livestock | Fraction of total livestock |
| | Grazing livestock density | Grazing animals per hectare grazing land |
| | Indoor livestock | Indoor animals per hectare feedland |
| | Phosphorus fertilization | Kg per ha grassland |
| | Grassland phosphorus export | Kg per ha grassland/forest |
| Crop production | Food and feed crop production | Ton per study area |
| | Bioeconomic land use | Fraction of total cropland used for bio-energy crops |
| | Crop productivity | Ton per ha |
| | Phosphorus fertilization | Kg per ha cropland |
| | Cropland phosphorus export | Kg per ha cropland |
| Forestry | Total wood production | $m^3$ per study area |
| | Wood productivity | $m^3$ per ha |
| | Production forest | Fraction of total forest |
| | Forest phosphorus export | Kg per ha forest |
| Population growth | Total population | Persons per study area |
| Urbanization level | Population in urban areas | Fraction of human population in urban areas |
| Social equity | Outdoor recreation inhabitants | Trips per capita |
| | Outdoor recreation nationals | Trips per capita |
| | Outdoor recreation internationals | Trips per capita |
| | Income inequality | Gini-coefficient |
| | Urban phosphorus export | Kg per ha built-up area |
| | Preference for nature protection | Preference relative to current |
| Economic growth | Regional GDP | Euro per capita |
| Energy | National total energy production | GJ per capita |
| | National forest energy production | GJ per capita |
| | National crop energy production | GJ per capita |
| | National peat energy production | GJ per capita |

## 2.3 Ecosystem services quantification

To estimate the quantified effects of the transition to a bioeconomy, we quantified for each NBP the full suite of values generated from ecosystem services in each study area as € $ha^{-1}$ $year^{-1}$, and laid them next to the estimated value under the current situation. To do so, we used our quantified NBP sub-elements, and linked them to catchment attributes, such as land cover used for specific crops or forestry, electricity generated by hydropower or recreational trips per inhabitant.

A more detailed analysis of the current value of ecosystem services generated in our study areas can be found in Immerzeel et al (2021). We have used this same dataset in the current study, calling it NBP0, with the following improvements:

1. Passive appreciation of nature is now included. This is based on willingness-to-pay estimates for area set aside for nature conservation, based on surveys among inhabitants and visitors to our study areas as described in Immerzeel et al (in review).
2. Production forest areas are now more precisely spatially defined, based on local spatial data and regional production statistics.
3. Sweden's agricultural land now also includes land used for grazing, as defined by the Swedish Board of Agriculture, instead of only arable land.
4. Peat extraction sites are now more accurately spatially defined, based on Bhattacharjee et al. (2021).
5. Land use in flooded areas is now based on more detailed land cover maps where available, instead of lower resolution CORINE land use data.

We quantify the value of final ecosystem services (FES), as defined in Boyd and Banzhaf (2007). After quantifying the value of all FES in all catchments under all NBPs, we performed chi-square tests to estimate whether there are statistically significant differences between the distribution of value over the separate ecosystem services for each NBP compared to NBP0 (the current situation).

We also grouped our study areas into two categories: rural and peri-urban, in which the rural catchments are dominated by forest cover and have relatively low population density, while the peri-urban catchments have a larger proportion of agricultural land cover and higher population density. We then compared the effects of the NBPs on the distribution over ecosystem services for these two categories. Given an on-going trend in peri-urbanisation (Bontje 2001) and the development of various policies for rural and urban regions (Nilsson et al. 2013), this contrast appears meaningful in an analysis relating to future land use change.

## 2.4 Change in spatial distribution

In analysing the effects on spatial distribution of ecosystem services generation, we used current land use and current TEV, as described in Immerzeel et al. (2021), as a starting point. We then took a two-step approach, where first we altered land use to fit the levels to the NBP sub-elements for each catchment, and then altered ecosystem service generation and value according to this new distribution of land use.

For each type of land use, we first defined the necessary growth or reduction within the catchment, according to the NBP, and then selected hectare cells to transform to or from that land use type, based on the attributes shown in Table 4, based on the assumptions that land use clusters together, that forestry and agriculture will most likely occur on suitable soils as well as close to access to infrastructure, that agriculture will expand close to built-up areas rather than remote areas, and that forestry will take more likely take place in fast-growing forests than in slow growing ones. For example, if agriculture needs to increase, it does so in cells close to those that are already agriculture, where the soil is suitable (not bare bedrock) and in proximity to roads, built-up area and habitation, changing the restrictiveness of these variables until the required additional cells are reached.

**Table 4. The effect of landscape attributes on changes in land use and type of production.** This shows the effect of landscape attributes for each hectare cell, and its effect on the likelihood of that cell transforming to a particular type of land use or production. +: positive effect. o: neutral effect. -: negative effect.

| | Proximity to the same land use | Soil quality | Slope | Proximity to urban and roads | Population density | Biomass productivity (site index) |
|---|---|---|---|---|---|---|
| Land use: agriculture | + | + | o | + | + | o |
| Land use: forest | + | - | + | - | - | + |
| Land use: built-up area | + | o | - | - | + | o |
| Land use: other nature | + | o | o | o | o | o |
| Forest: production forest | + | + | - | + | - | + |
| Agriculture: Grains | + | + | o | o | o | o |
| Agriculture: Grass | + | o | + | - | o | o |
| Agriculture: Other crops | + | + | - | + | o | o |
| Peat extraction | + | + | o | o | o | o |

## 2.5 Change in distribution among stakeholder groups

We split the beneficiaries of the generated ecosystem services into four stakeholder groups, as in Immerzeel et al. (2021). These are: landowners (benefitting from agriculture, forestry and flood prevention), large extractors (benefitting from water extraction, electricity generation and peat extraction), visitors (benefitting from active nature appreciation) and global society (benefitting from mitigated climate change and passive nature appreciation). By summing the altered TEV estimates for each NBP and dividing them over these stakeholder groups, we could compare how the NBPs affected the distribution of value over these groups.

# 3. Results

## 4.1 Change in ecosystem services value

The value of ecosystem services generated in our six study areas varies among the NBPs, and the effect of the NBP varies for each study area (Figure 2). The strongest effect is typically found on active nature appreciation, which is the value of direct interaction with nature for enjoyment, such as hunting, fishing, hiking, swimming and appreciating the aesthetics of the landscape. Since this ecosystem service is directly related to population, this also means that the strongest effects on total economic value tend to occur in the densely populated areas: in Odense, the value of active nature appreciation more than doubles from 7,080 to 14,786 € $ha^{-1}$ $year^{-1}$ from NBP0 to NBP5. In Simojoki, peat production is one of the main ecosystem services, and this service is strongly affected by the NBPs, ranging in value from 0 in NBP1 (where all peat production is ceased due to climate considerations) to 122 euros per hectare per year in NBP5. When testing for effects on the distribution of value over the separate FES, most scenarios show significant differences compared to NBP0. NBP3 however does not show significant change in FES distribution.

The effects of the NBPs also depend on the type of study area, especially for NBP4, which focuses on increasing rifts between rural and urban areas. In this scenario, the more agricultural, densely populated peri-urban areas (Orrevassdraget, Odense and Sävjaån) do better than under NBP0, while in the other, more rural, areas less value is generated than currently.

In all catchments except Simojoki, NBP1 and NBP5 generate most value, their ranking depending on how much of the value is generated by produced goods (which is favoured in NBP5) and how much by nature appreciation (which is favoured in NBP1). Only in Simojoki is NBP1 the lowest ranking scenario, because a large part of this catchment's value comes from peat extraction, which is ceased under NBP1, the greenest scenario.

When looking in more detail at Haldenvassdraget (Table 5), we see that the distribution of crop types changes slightly, for instance with the introduction of energy crops into the catchment, but that the general picture remains similar. Roundwood removal is drastically reduced in NBP4 due to neglect of rural areas, and in NBP1 due to a focus on circular economy and reducing resource use, combined with a shift from production forest to more natural forest to protect biodiversity and to increase recreational opportunities. In the other scenarios production increases. Hunting value mostly follows population change, though under NBP1 interest in hunting has decreased due to increasing interest in nature protection, leading to a decrease in the value of shot game. Electricity generation depends on population as well as energy intensity, with highest value under NBP5, where the focus lies on producing economic output, and the lowest under NBP4, where rural areas lose inhabitants. Foraging (collecting berries and mushrooms) and water consumption mostly follow population development, though the value of water as an ecosystem service depends on the cost of generating clean drinking water. This means that in NBP5, where land use is intensive and water quality is reduced by nutrient runoff from agriculture and forestry, the value of access to clean drinking water is dampened by the cost of treating it. Nature appreciation shows the widest spread of all ecosystem services, with highest values under NBP1, where people are willing to pay more for recreation and nature conservation, and lowest under NBP4, where the opposite occurs. Carbon sequestration varies according to the area covered by forest, as well as biomass productivity. The value of flood prevention finally varies relatively little: scenarios where there is more downstream area used for buildings and infrastructure see the highest increase in benefit.

When comparing rural, forested catchments to peri-urban, agricultural catchments, the effects on relative weight of separate ecosystem services varies across NBPs and across type of study area (Figure 3). Typically, nature appreciation, both active and passive, becomes more dominant at the cost of the other ecosystem services, though this effect is strongest in the rural study areas (Haldenvassdraget, Simojoki and Vindelälven). The weight of agriculture and forestry decreases under all NBPs in both types of study areas, except agriculture under NBP3, since this scenario focuses on self-sufficiency. Greatest contrast between peri-urban and rural catchments can be found under NBP4, the scenario where divisions between rich and poor widen. Here the relative weight of active nature recreation increases in the peri-urban catchments, where population and wealth concentrate, while in rural areas population will decline and the focus will shift to large extraction industry for peat and water, as well as carbon sequestration in unmanaged forests.

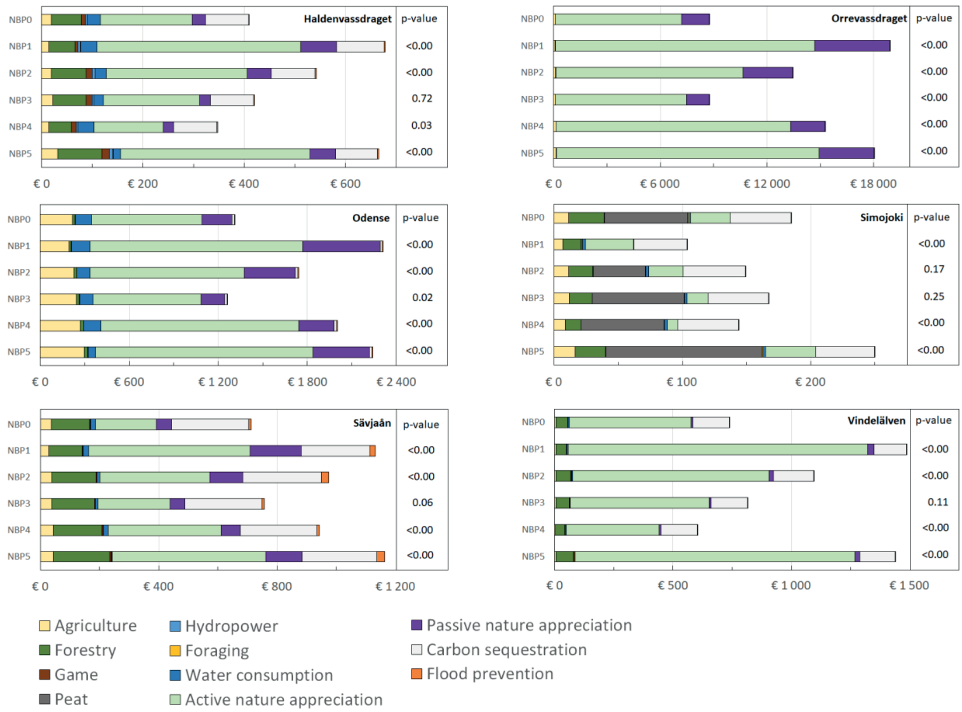


**Figure 2. Economic value of groups of ecosystem services generated in our study areas for each NBP, in € ha$^{-1}$ year$^{-1}$.** This shows per study area the economic value of all estimated ecosystem services. Next to each bar we give a p-value for the chi-square test statistic, indicating whether there is a statistically significant difference in distribution over the different services compared to NBP0. A Bonferroni correction for unplanned multiple comparisons would lead to an individual pairwise p=0.01 corresponding to an overall error rate of p=0.05.

**Table 5. Economic value of separate ecosystem services generated in Haldenvassdraget for each NBP, in € $ha^{-1}$ $year^{-1}$.** For the values in the five other catchments, see Supplement 1. Note that these are means over total catchment area, not over area where the FES is generated.

| | NBP0 | NBP1 | NBP2 | NBP3 | NBP4 | NBP5 |
|---|---|---|---|---|---|---|
| Grains | 11 | 9 | 13 | 13 | 8 | 20 |
| Grazing and fodder | 7 | 5 | 7 | 9 | 5 | 12 |
| Energy crops | 0.00 | 0.07 | 0.05 | 0.06 | 0.04 | 0.08 |
| Other crops | 0.11 | 0.09 | 0.12 | 0.13 | 0.08 | 0.20 |
| **Agriculture** | **19** | **14** | **20** | **22** | **14** | **32** |
| Roundwood removal | 60 | 52 | 69 | 66 | 45 | 88 |
| **Forestry** | **60** | **52** | **69** | **66** | **45** | **88** |
| Hunted big game | 8 | 6 | 12 | 11 | 10 | 14 |
| Hunted small game | 0.01 | 0.01 | 0.02 | 0.02 | 0.01 | 0.02 |
| **Game** | **8** | **6** | **12** | **11** | **10** | **14** |
| Milled peat | 0 | 0 | 0 | 0 | 0 | 0 |
| **Peat** | **0** | **0** | **0** | **0** | **0** | **0** |
| Electricity generated | 4 | 5 | 5 | 4 | 3 | 6 |
| **Hydropower** | **4** | **5** | **5** | **4** | **3** | **6** |
| Berries gathered | 0.15 | 0.26 | 0.21 | 0.16 | 0.11 | 0.28 |
| Mushrooms gathered | 0.22 | 0.38 | 0.31 | 0.24 | 0.17 | 0.43 |
| **Foraging** | **0.36** | **1** | **1** | **0.40** | **0.28** | **1** |
| Water consumed from catchment | 26 | 32 | 22 | 18 | 32 | 15 |
| **Water consumption** | **26** | **32** | **22** | **18** | **32** | **15** |
| Value of hunting | 1 | 1 | 1 | 1 | 1 | 2 |
| Value of fishing | 0 | 0 | 0 | 0 | 0 | 0 |
| Value of recreational trips - inhabitants | 162 | 361 | 249 | 170 | 128 | 335 |
| Value of recreational trips - national visitors | 17 | 37 | 26 | 18 | 7 | 35 |
| Value of recreational trips - international visitors | 1 | 3 | 2 | 1 | 1 | 2 |
| **Active nature appreciation** | **181** | **402** | **278** | **190** | **137** | **375** |
| Area of nature reserve | 26 | 71 | 47 | 22 | 21 | 50 |
| **Passive nature appreciation** | **26** | **71** | **47** | **22** | **21** | **50** |
| Carbon stored in biomass | 82 | 90 | 83 | 82 | 82 | 80 |
| Carbon stored in lakes | 3 | 3 | 3 | 3 | 3 | 3 |
| **Carbon sequestration** | **85** | **94** | **87** | **85** | **85** | **83** |
| Water prevented from flooding land | 1 | 2 | 3 | 2 | 1 | 3 |
| **Flood prevention** | **1** | **2** | **3** | **2** | **1** | **3** |
| **TOTAL** | **410** | **679** | **543** | **421** | **348** | **666** |

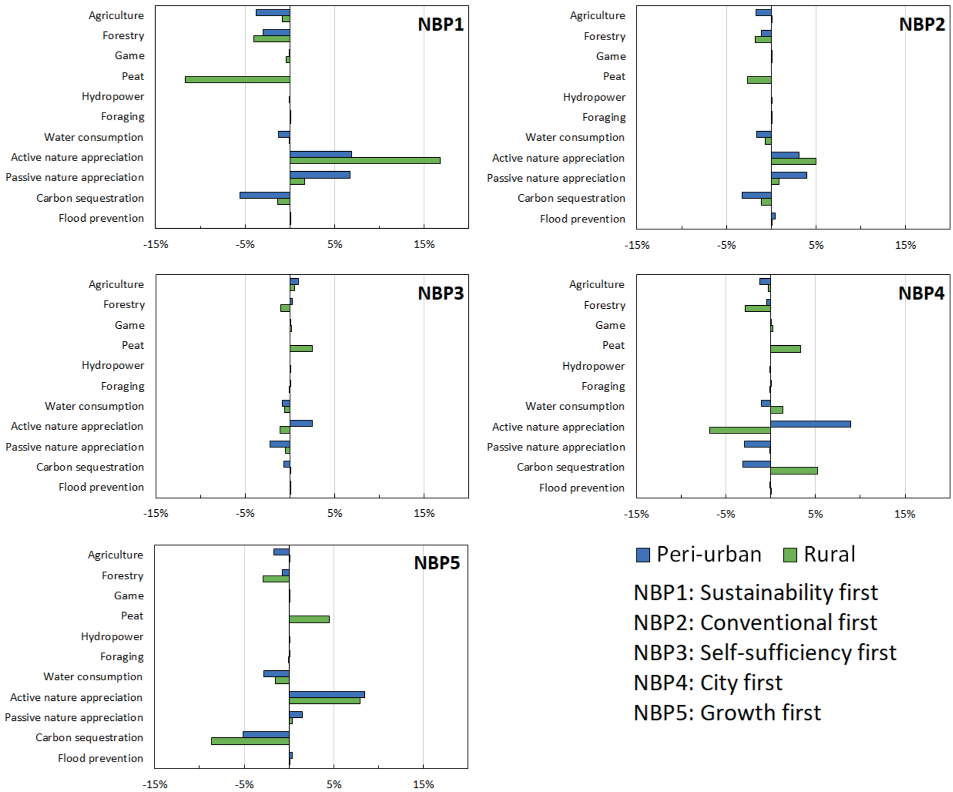


**Figure 3. Change in weight of ecosystem services.** This shows per NBP how the relative weight of the separate ecosystem services has changed compared to NBP0. For example, a decrease of agriculture by 4% under NBP1 means that its share of the total value of all ecosystem services has decreased from 8% to 4%. We make a distinction between peri-urban catchments (Orrevassdraget, Odense and Sävjaån), and rural catchments (Haldenvassdraget, Simojoki and Vindelälven) by taking the average effects for all three catchments within each of the two categories.

## 4.2 Change in spatial distribution

Land use patterns change in all NBPs in all study areas. Taking Haldenvassdraget as an example, comparing NBP0, the current situation, to NBP1 and NBP5 shows clearly visible changes in land use patterns (Figure 4). In NBP1, the green scenario with more efficient resource use and more consideration for environmental issues, agricultural area decreases. This is due to reduced meat production and increased agricultural productivity per hectare. Built-up area increases to accommodate for population growth, and forest increases at the cost of all other land use. In contrast, under NBP5, a scenario focused on maximizing economic production, forest area decreases slightly, while agricultural area increases, since agriculture generates higher income per hectare.

When considering the spatial distribution of TEV, the NBPs show different patterns of change compared to NBP0. In Haldenvassdraget (Figure 5), more remote areas of the catchments decrease in generated value under NBP1, since timber production is ceased there and only continues in highly productive, well connected sites. Nature reserves increase in size, and since these are directly linked to the value of passive nature appreciation in our framework, these extensions see the greatest increase in value. Areas with an appealing mix of land use types for recreation that are well connected also see moderate increases in value. Under NBP5, value decreases in built-up areas and on surface water. This is caused by the decrease of access to clean drinking water, which in turn is caused by increased nutrient runoff caused by the maximization of production of crops and timber. Areas where value increase are mostly found on cropland and in sites with good opportunity for recreation, since under this scenario agricultural production as well as incomes have increased significantly.

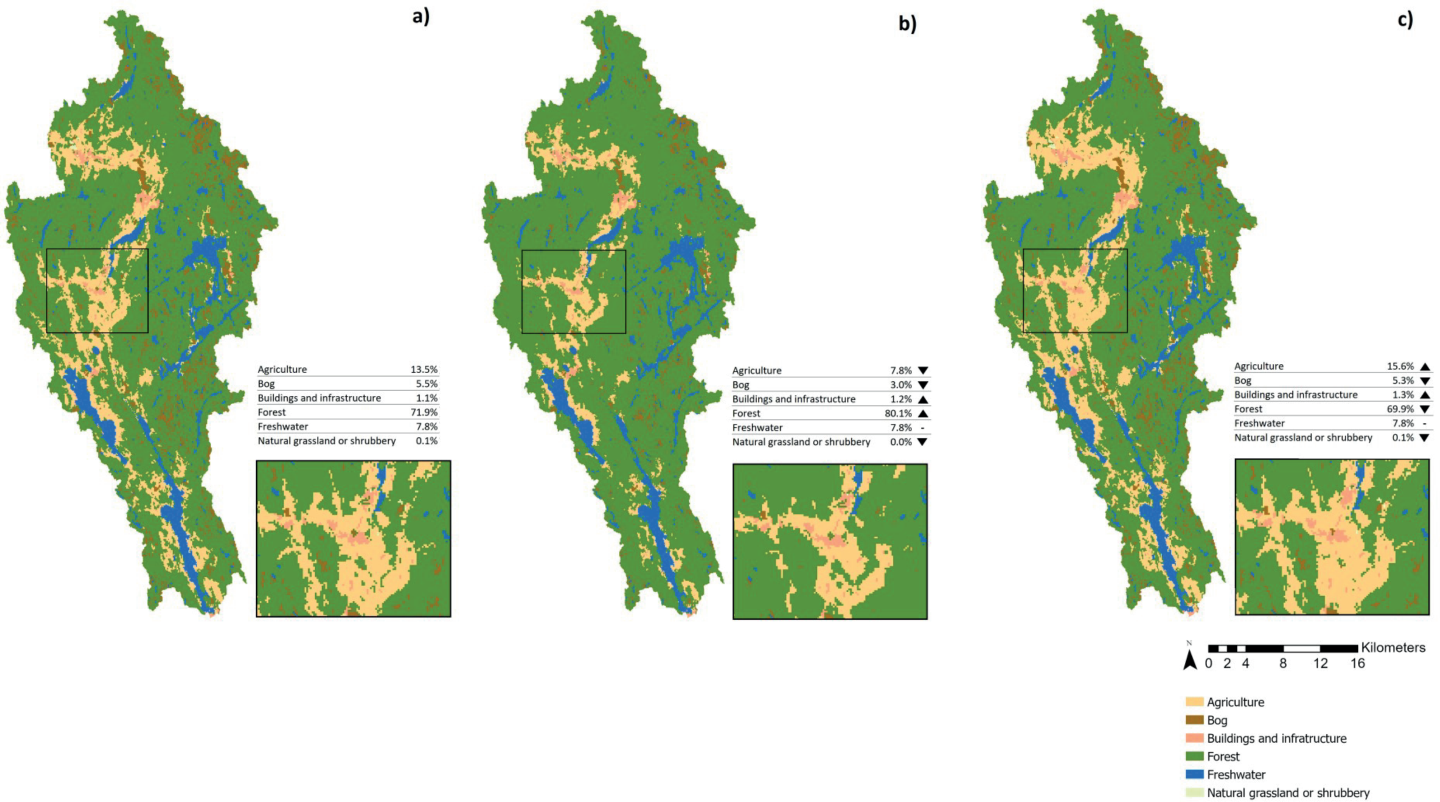


**Figure 4. Land use under various NBPs in Haldenvassdraget.** A): NBP0 (current situation). B): NBP1 (sustainability first). C): NBP5 (growth first). This shows land use under three scenarios in Haldenvassdraget. The table shows percentage of total area per land use type, with arrows indicating change compared to NBP0. The frame shows a zoomed in part of the catchment with mixed land use.

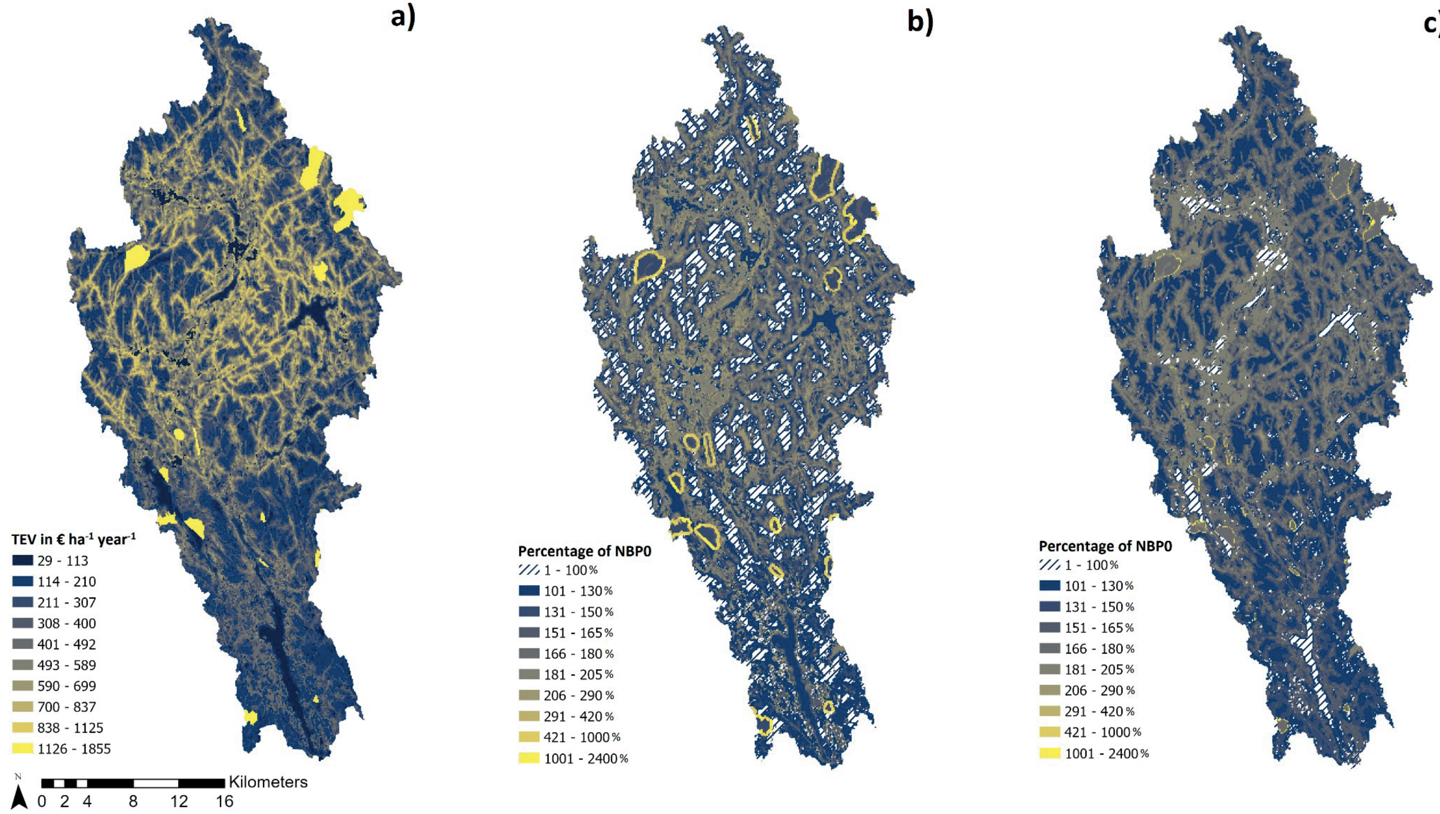


**Figure 5. Change in Total Economic Value (TEV) compared to NBP0 in Haldenvassdraget.** a): absolute TEV per hectare under NBP0. b): Percentage of NBP0 value under NBP1. c): Percentage of NBP0 value under NBP5.

## 4.3 Change in distribution among stakeholder groups

Each scenario has different effects on different stakeholder groups (Figure 6), though their ranking does not change in any of the scenarios. Landowners benefit in all scenarios except NBP1, where agriculture needs to make place for nature. They gain most in NBP5, where increased economic production is the core focus. Large extractors see little change in value, while in contrast visitors see the largest increases in value of all groups. Under NBP1 and NBP5 recreational value increases most, due to a mix of population growth, income growth and, in NBP1, increased interest in nature recreation. Global society, depending on passive nature appreciation and carbon sequestration, are those not directly interacting with the ecosystem. They benefit most under NBP1, where nature reserves increase in size most and carbon sequestration increases most due to reforestation. Conflicts can potentially arise where one group gets increased value while another gets less. An example is NBP1, where landowners and large extractors see their benefits decrease while visitors and global society see a large increase in benefits. There is large variability among study areas though (Supplement 2), as well as within them, so spatial distribution is an important factor when considering potential conflict as well.

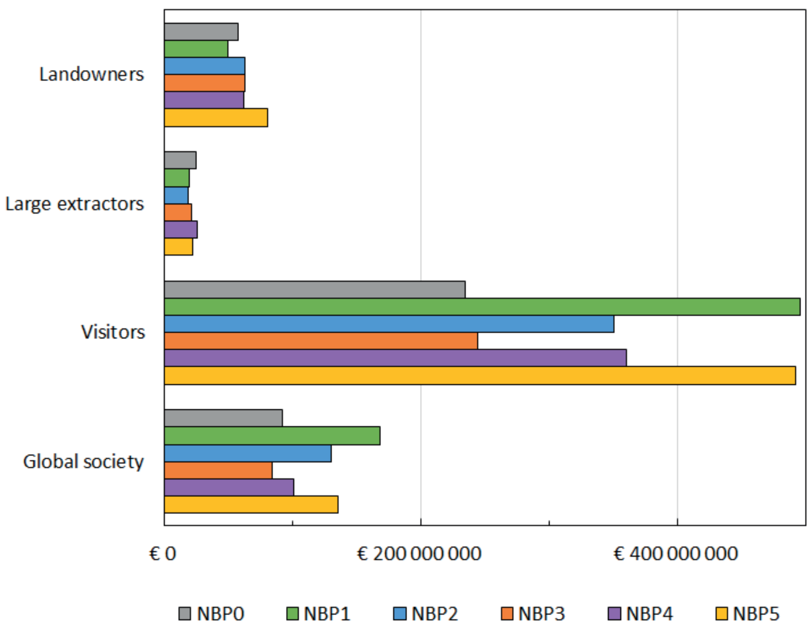


**Figure 6. Ecosystem service value generated per stakeholder group in € year$^{-1}$.** This shows per stakeholder group the summed value from all catchments and how these vary among NBPs. Landowners are those that benefit from agriculture, forestry and flood prevention. Large extractors are those that benefit from peat extraction, electricity generation and water production. Visitors are those that benefit from hunting, foraging and active nature appreciation. Global society are those that benefit from passive nature appreciation and carbon sequestration.

# 4. Discussion

## 4.1 Interpretation of the results

Our estimates indicate that the way in which society transitions to a bioeconomy can have substantial impacts on the amount of value generated by ecosystem services in Nordic catchments. When considering the total value generated in our six study areas, the first thing of note is that NBPs 1 and 5 generate most value (Figure 2). These scenarios represent widely different futures: NBP1 represents a world where human pressures on ecosystems are minimized, where the reduction of greenhouse gas emissions is prioritized and where consumption is more resource-efficient and circular. NBP5 is a world where economic growth takes first priority, with high gross domestic products, continued reliance of fossil fuel though supplemented with green technologies, and production taking account of local environmental issues that might directly impact citizens in the short term. That such different modes of organizing society can both yield improved benefits compared to the current situation suggests there is flexibility in how the bioeconomy is implemented. However, a key requirement seems to be consideration for local environmental impacts. Though NBP5 might have large negative effects on global society and climate, it does try to protect the local environment for the benefit of its inhabitants. In scenarios where local environmental considerations are less strict, NBP3 and NBP4, we see much lower welfare benefits. Overall, NBP3, defined by protectionism and self-reliance, generates the lowest ecosystem service value. This emphasizes that much of the current welfare of the Nordic societies depends on their connections to the rest of the world, and moreover, that weakening these ties can have significant impacts on local ecosystems. The net effect of NBP4 is slightly more positive, though here the distribution of value among catchments becomes more important. Peri-urban catchments clearly benefit more in this scenario than rural catchments, especially since they attract more recreational value. This has policy implications: tensions between developed areas and rural areas regarding economic development and ecosystem management are already part of the political debate in the Nordic countries (Arter 2011, Krange et al. 2017), and can be further exacerbated depending on which route to bioeconomy is chosen.

We also found significant differences in the distribution of value over the separate ecosystem services (Figure 2). NBP3 was most comparable to the current situation, but most others (with the exception of NBP2 in Simojoki) showed significant differences compared to NBP0. This suggests that under different scenarios, benefits are not only of different magnitudes, but are also distributed differently over stakeholder groups. When summing up all value flowing to our four main stakeholder groups (Figure 6), this is indeed what it looks like. Landowners in general benefit from the expected growth in gross domestic product, but they would benefit more under NBP5, and would even benefit less than currently under NBP1. Under this green scenario, large extractors would also take a step back, while visitors and global society receive close to double the value they currently do. This can potentially lead to conflict between resource extractors (landowners and large industry) and those benefiting from non-material ecosystem services such as nature appreciation and mitigated climate change. Whether conflicts arise can also depend on where value is generated within an area. Our illustration using Haldenvassdraget (Figure 5) shows that under NBP1 especially, large parts of forested area will generate less value than currently, precisely because the landowners will not be profiting as much from timber extraction.

## 4.2 Testing our hypotheses

Our findings support the evidence that the NBPs significantly alter the distribution of generated value over the different ecosystem services. This does not hold for all NBPs in all study areas, with NBP3 not showing a significant effect on the distribution in Haldenvassdraget, Simojoki, Sävjaån and Vindelälven, but overall, 23 out of 30 scenario combinations (five NBPs times six study areas) show significantly different distributions over the ecosystem services compared to the current situation.

Our results partially support the hypothesis that the NBPs have different effects in peri-urban areas than in rural areas. In each of the NBPs we find differences between distribution of value over ecosystem services. For example, in NBP1, the share of value going to active nature appreciation increases by more than 15 percent points in rural areas, while it only increases by around 5 percent points in peri-urban areas. However, while the size of the effects vary, the direction is often the same in both types of catchment: in NBP1, active nature appreciation does increase in both types of catchment, and the relative weight of agriculture, forestry and carbon sequestration decrease.

We found support for the hypothesis that different stakeholder groups are affected differently by the NBPs. For example, under NBP1, landowners and large extractors are worse off than now, while visitors and global society are better off. In NBP5, we find that all stakeholder groups receive more value than currently, but recreating visitors benefit disproportionally, with a doubling of value.

## 4.3 Methodological limitations and data gaps

The accuracy of our method is based on several assumptions. Firstly, we base our estimates for what the NBPs will look like on a combination of sources, ranging from population projections to expert judgment. There is of course great uncertainty to what the year 2050 will look like so we cannot claim to predict the future with these scenarios (Nakicenovic et al. 2000). We only aimed to show possible futures, which meant that we focused on what was physically possible. We tried for instance not to increase forestry productivity to a level that no forest could biologically ever reach. In our estimates we did not consider time explicitly. We quantified a static moment and did not take into account annual timesteps, so we cannot show change in distribution over time. We also did not consider the direct effects of climate change, such as changes in biomass productivity due to increased temperature or change in flood frequency. This reduces the realism of our scenarios, but we argue this is not a major issue since we are focusing on the effects of a transition towards a bioeconomy, independently of the effects of climate change, and by ignoring climate change we can more clearly show the effects of what is in essence a societal transition. Finally, catchment scale modelling of hydrology, carbon and nutrient fluxes was outside of the scope of this research. This means the static state we create cannot include interactions between agricultural practices, groundwater flow, surface water quality, and we were limited in our quantifications of the links between catchment processes, such as nutrient cycling, and FES. Therefore we had to rely on rough estimates of effects based on expert judgment.

## 4.4 Further research

We see various avenues of further research that can build upon the results of this work. Firstly, our framework was designed to be flexible enough to apply to a wide range of catchment types. Land use change in a bioeconomy context is a key policy focus in many countries (Hetemäki and Muys 2017, Eyvindson et al. 2018, Issa et al. 2019). The main requirement for our framework is data availability. So as long as statistics on production and land use are available, various scenarios can be applied to estimate their effects on ecosystem services generation. In new applications of the framework, the effects of climate change can also be integrated into the framework, as modulating variables on for instance biomass productivity and flood frequency. If time dynamics are of interest, these can also be incorporated into the framework by estimating ecosystem services generation in time steps and placing these in order, feeding into each other to create a more dynamic framework. This can also allow for the inclusion of discounting of value over time (Drupp et al. 2018). Finally, dynamic catchment models can be linked to this framework to more closely approximate the interactions between catchment processes and ecosystem services generation. SWAT and INCA-P are examples of catchment models that have been applied to these and similar catchments (Farkas et al. 2013, Abbaspour et al. 2015, Molina-Navarro et al. 2018), and which could be linked to our framework.

## Concluding remarks

We found that a transition to a bioeconomy can have widely varying effects on ecosystem services generation in Nordic catchments, depending on the shape of the bioeconomy. Different scenarios can have profoundly different effects on the magnitude of value being generated, the distribution of value coming from different types of ecosystem services, the differences in value when comparing rural, forested catchments to peri-urban agricultural catchments, and on the distribution of value over different stakeholder groups in society. In sum, most value is being generated by NBP1, the scenario of green transition and focus on sustainability and social equity. This is not good news for all stakeholder groups though, as landowners and large extractors will see their benefits deteriorate under this scenario. This illustrates how fragile the balance of wellbeing is, not only on the side of ecosystems, but also on the societal side. Active nature appreciation appears to be the ecosystem service most affected by the NBPs, reinforcing the notion that how we manage land for the production of goods can have strong ripple effects into other processes, such as welfare generated from recreation. If policy makers want to ensure a sustainable bioeconomy on both an environmental and a societal level, they need to take all these aspects into consideration.

Our framework shows how a possible future bioeconomy can affect various groups in society through changes in the generation of ecosystem services. Sensitivity analyses and the consistency of the outcomes give us confidence in its results, and its successful application in six varying catchments, spread across four countries, show that it can be applied to a wide variety of landscapes and scenario settings, making broader application possible. Further research should focus on such applications, as well as on refining the framework to include more dynamic processes.

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# Supplementary materials

1. NBP quantification framework
2. Change in value distributed over ecosystem services and stakeholder groups

Until publication the supplementary materials are available upon request from the first author.

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