The monetary value of marine environmental change

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Summary

The marine ecosystems are fundamental for human welfare. A number of current environmental pressures need attention, and the formulation of management strategies requires information from a variety of analytical dimensions. The linkage between environmental change and resulting implications for human welfare is one such dimension.

This thesis presents studies on welfare implications from hypothetical future policies which improve the state of the marine environment. The method for these studies is economic valuation. The studied scenarios concern eutrophication in the Baltic Sea (including the Kattegat) and oil spill risk from shipping in the Lofoten-Vesterålen area in the Arctic Barents Sea. The thesis shows that the economic benefits from undertaking policies to improve or protect the marine environment in these cases are substantial and exceed the costs of taking measures.

In addition to providing new monetary estimates, the thesis also provides new insights concerning 1) what type of scenario to use when valuing an environmental improvement and 2) whether there may exist trade-offs between precision in estimates and the level of ambition with respect to survey instrument complexity and econometric models when conducting valuation studies. The findings suggest an end of an era for studies in which the environmental change is unspecified or based on a single environmental indicator while the actual consequences of the suggested measures are more multifaceted. In contrast, relevant scenarios to study are well-specified and holistic. The thesis further reveals that it might not always be worth the effort to go for the most advanced scenario presentation or statistically best-fitting model specifications. This is something that needs to be further discussed among practitioners in order to allocate valuation resources wisely and not waste resources on unnecessarily elegant valuation studies.

Keywords

Contingent valuation, choice experiment, benefits transfer, cost-benefit analysis, ecosystem services, eutrophication, oil spills, Baltic Sea, Arctic.

Sammanfattning

Den marina miljön är avgörande för vår välfärd, men ekosystemen hotas av en mängd olika miljöbelastningar. För att utforma rätt förvaltningsstrategier behövs information från flera discipliner och analysmodeller, varav kopplingen mellan miljön och människors välfärd är en.

Denna avhandling belyser välfärdskonsekvenser till följd av hypotetiska framtida policyåtgärder som förbättrar tillståndet för den marina miljön. Metoden för detta är ekonomisk värdering. De studerade scenarierna avser övergödning i Östersjön (inklusive Kattegatt) och oljeutsläpp från sjöfarten i Lofoten-Vesterålen i Barents hav. Avhandlingen visar att nyttorna av strategier för att förbättra eller skydda den marina miljön i dessa fall är betydande och överstiger kostnaderna för att vidta åtgärder.

Förutom att presentera monetära skattningar ger avhandlingen också nya insikter beträffande 1) vilken typ av scenario som ska användas vid värderingen av en miljöförändring och 2) om det kan finnas avvägningar mellan precision i beräkningarna och ambitionsnivå i utförandet av värderingsstudier, rörande frågeformulärets komplexitet och valet av ekonometriska modeller. Resultaten tyder på slutet på en era för studier där miljöförändringar är ospecificerade eller baserade på en enda miljöindikator medan de faktiska konsekvenserna av de föreslagna åtgärderna är mer mångfacetterade. Istället bör relevanta scenarier vara väl specificerade och holistiska. Avhandlingen visar vidare att det inte alltid är värt besväret att använda den mest avancerade scenariopresentationen eller den statistiskt bästa modellspecifikationen. Enklare upplägg kan ge tillfredsställande resultat. Detta är något som utförare av värderingsstudier behöver diskutera ytterligare i framtiden för att klokt kunna allokera de resurser som behövs för att genomföra studier.

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List of appended papers

Paper I — **Non-market valuation of the coastal environment – uniting political aims, ecological and economic knowledge**.
Östberg, K., Hasselström, L., Håkansson, C., 2012. Journal of Environmental Management 110: 166-178.

Paper II — **Benefit transfer for environmental improvements in coastal areas: general vs. best-fitting models.**
Östberg, K., Håkansson, C., Hasselström, L., Bostedt, G., 2013. Canadian Journal of Agricultural Economics 61 (2013) 239–258.

Paper III — **Detailed vs. fuzzy information in non-market valuation studies – the role of familiarity.**
Hasselström, L., Håkansson, C., 2014. Journal of Environmental Planning and Management 57 (1): 123-143.

Paper IV — **Public preferences regarding use and condition of the Baltic Sea – An international comparison informing marine policy**.
Ahtiainen, H., Artell, J., Czajkowski, M., Hasler, B., Hasselström, L., Hyytiäinen, K., Meyerhoff, J., Smart, J., Söderqvist, T., Zimmer, K., Khaleeva, J., Rastrigina, O., Tuhkanen, H., 2013. Marine Policy 42: 20-30.

Paper V — **Benefits of meeting nutrient reduction targets for the Baltic Sea – a contingent valuation study in the nine coastal states**.
Ahtiainen, H., Artell, J., Czajkowski, M., Hasler, B., Hasselström, L., Huhtala, A., Meyerhoff, J., Smart, J., Söderqvist, T., Alemu, M.H., Angeli, D., Dahlbo, K., Fleming-Lehtinen, V., Hyytiäinen, K., Karlõševa, A., Khaleeva, Y., Maar, M., Martinsen, L., Nõmmann, T., Pakalniete, K., Oskolokaite, I., Semeniene, D., 2014. Journal of Environmental Economics and Policy 3(3): 278-305.

Paper VI **Baltic Sea nutrient reductions – what should we aim for?**
Ahtiainen, H., Artell, J., Elmgren, R., Hasselström, L., Håkansson, C., 2014. Journal of Environmental Management 145: 9-23.

Paper VII **Valuation of oil spill risk reduction in the Arctic**.
Noring, M., Hasselström, L., Håkansson, C., Soutukorva, Å., Gren, Å., 2016. Journal of Environmental Economics and Policy. DOI: 10.1080/21606544.2016.1155499.

Paper VIII **Costs and benefits associated with Arctic marine oil spill prevention**.
Noring, M., Hasselström, L., Håkansson, C., Soutukorva, Å., Khaleeva, J. Submitted manuscript.

Author contributions

Paper I: KÖ, LH, CH designed research jointly. LH conducted data collection based on the survey that was jointly developed by KÖ, LH and CH. KÖ and CH conducted data analysis with input from LH. KÖ, LH and CH wrote the paper jointly with leadership by KÖ.

Paper II: KÖ, CH, LH and GB designed research jointly. LH conducted data collection. KÖ conducted data analysis with input from CH, LH and GB. KÖ, CH, LH and GB wrote the paper with leadership by KÖ.

Paper III: LH and CH designed the research jointly. LH conducted survey. CH conducted data analysis with input from LH. LH and CH wrote the paper jointly.

Paper IV: HA, JA, MC, BH, LH, KH, JM, JS, TS, KZ, JK, OR and HT designed research jointly. LH and TS coordinated data collection. HA, TS, JA and MC analyzed data with input from all authors. HA, JA, MC, BH, LH, KH, JM, JS, TS, KZ, JK, OR and HT wrote the paper with main contributions from HA and TS.

Paper V: HA, JA, MC, BH, LH, AH, JM, JS, TS, MHA, DA, KD, VFL, KH, AK, YK, MM, LM, TN, KP, IO, DS designed research jointly. LH conducted data collection in Sweden, including pretesting (focus groups and pilot study) which fed into the final design of the survey. Data analysis was conducted by HA and JA with input from all authors. LH and HA coordinated working paper which was the basis for journal article manuscript. The paper was written jointly by all authors with leadership from HA.

Paper VI: LH and CH developed the original research idea, which was then refined and fully developed by HA, JA, RE, LH and CH. The dataset collected for paper V was used. Data analysis was conducted by JA and HA with input from RE, LH and CH. The paper was written jointly by HA, JA, RE, LH and CH.

Paper VII: The research idea was developed jointly by MN, ÅG, LH, CH and ÅS. The survey development was coordinated by LH and the data collection was conducted by LH. LH was lead author of working paper which formed an early basis for the journal manuscript. MN, ÅG, LH, CH and ÅS wrote the paper jointly under MN's leadership.

Paper VIII: MN, LH, CH, ÅS and JK developed the research idea jointly. Data on benefits was based on the dataset gathered for paper VII. Data on costs was collected by MN with assistance from all authors. The paper was written jointly by MN, LH, CH, ÅS and JK under MN's leadership.

Contents

1. Introduction

1.1. Background

Functioning ecosystems are a prerequisite for the well-being of our societies. In addition to the provisioning of goods such as food, fiber, pharmaceuticals, and industrial products, the ecosystems also provide life-supporting processes of air and water purification, pollination, climate regulation, flood protection, primary production, etc. The understanding of linkages between the ecosystem and human societies is a developing scientific field. As an overarching concept, the term 'ecosystem services' (MA, 2005; TEEB, 2010) is becoming widely used in science, administration, politics and rhetoric – protection of the environment is not only for the environment's own sake: human well-being depends on it. The very idea of ecosystem services is however not new. One of the first records of the idea is from Plato (400 BC) who expressed the concern that deforestation could lead to soil erosion and the drying up of springs (Daily, 1997).

Some of these dependencies are visible on markets, such as when drought leads to negative impacts on agriculture. Other dependencies appear off the market, such as health impacts from pollution or the well-being generated from recreation in a nice coastal area. The valuation of these non-market benefits in monetary terms dates at least back to Harold Hotelling's famous letter to the US Park Service in 1949, in which he proposed the study of travel costs of visitors to assess the benefits associated with visits (Hotelling, 1949). Since then, the amount of 'valuation studies' has grown steadily to become an own scientific field.

The field of economic valuation is usually motivated by the demand for decision support concerning trade-offs in society that involve potential changes in environmental quality (cf. TEEB, 2010; Freeman et al., 2014). Measures to avoid environmental degradation or to improve the state of an already degraded ecosystem are associated with costs and benefits, and by illuminating the benefits, more informed comparisons between costs and benefits can be made.

The global marine environment is of enormous importance to human well-being, but the ecosystems are threatened. For example, a recent report by WWF (2015) shows that the populations of fish species utilized by humans have halved since 1970. This thesis focuses on the Baltic Sea and the Lofoten-Vesterålen area in the Arctic Barents Sea. The coastal and marine environment in the Baltic Sea area is suffering from pressures such as excessive nutrient loads, hazardous substances, overfishing, acidification, global warming, underwater noise from shipping and development activities, physical disturbances to the sea bottom, and habitat degradation (Helcom, 2010). The Arctic marine areas, being of global importance for biodiversity and ecosystem services, are of increased strategic interest for development, including shipping and exploration for gas and oil (AGP, 2010). Climate change, industrial development and resource exploitation lead to a number of changes in the ecosystem in the region, such as decreased habitat quality, changes in vegetation, and decreasing populations of many species (CAFF, 2010).

The thesis, including the eight research papers, focuses on eutrophication in the Baltic Sea (including the Kattegat) and on the risk for oil spills in the Arctic. The management of these marine environments requires a holistic perspective encompassing an understanding of drivers behind various environmental pressures, the state of the ecosystem, the impacts on society from state changes, and potential policy responses (Scharin et al., 2016). This systems approach is usually referred to as the DPSIR model (Drivers, Pressures, State, Impact, Response; OECD, 1993; Turner et al., 1998). In order to formulate sustainable management, the information from a number of analytical dimensions is needed. Often, at least three dimensions are mentioned – environmental, social and economic sustainability (as reflected in e.g. the EU directive for marine spatial planning (Directive 2014/89/EU)).

An immense array of analytical tools is available for assessing management options within these three dimensions. These tools can be categorized based on their characteristics in a number of ways (Finnveden & Moberg, 2005; Ness et al., 2007). Cost-benefit analysis (CBA) is one of these tools, with an explicit focus on the economic dimension of sustainability. Scharin et al. (2016) mention CBA as a possible first step in deciding measures and enabling policy instruments and suggest that this type of assessment should be accompanied with other types of assessments such as analyzes of distributional effects of the measures and scenarios for possible future developments, including the risk for sudden state changes, regime shifts or even system collapses.

An overarching question concerning the economic dimension of sustainability concerns trade-offs. CBA is specifically oriented towards addressing such trade-offs from a welfare economics point of view. In order to conclude on whether a specific measure is economically motivated or not, the negative and positive consequences of the measure need to be compared. For measures resulting in changes of the state of the ecosystem, welfare implications from such changes need to be addressed. This is where the economic valuation of environmental change becomes relevant (while there are also other motivations for valuation, see section 2.3).

Valuation of environmental change in economic terms is a rather mature scientific field. However, in order to develop methods further and to provide estimates from various applications as input to management, an infinite amount of research gaps exist. For the Baltic Sea environment, an overview of valuation studies was conducted in 2009 (Söderqvist & Hasselström, 2009), suggesting a need for estimates covering not only additional local areas but also the Baltic Sea as a whole. For the Arctic marine areas, not much socioeconomic research has been previously conducted. A particularly timely research gap concerns oil spill risks due to the potential increase of economic activity, shipping and oil exploration in the region. Methodologically, a number of recent developments generated new research needs at the time being when the research in this thesis was formulated. The establishment of the EU Water Framework Directive (WFD; Directive 2000/60/EC) and the EU Marine Strategy Framework Directive (MSFD; Directive 2008/65/EC) has stipulated studies on the costs and benefits of measures to reach the environmental objectives. First, this implies a need for tailor-made valuation scenarios. Second, given limited budgets available for the potentially massive amount of economic assessment needed in relation to these directives a methodologically interesting question is raised: Is it possible to make assessments themselves more cost-effective?

Concerning the cutting of costs for conducting valuation methods, at least two options are available: To reduce the often time-consuming design of valuation studies at the possible expense of lower precision in the estimates, or to conduct so-called benefit transfer (BT). BT is based on simulating a primary study in an area (policy site), using results from existing studies at another site (study site). One method for conducting BT is so-called function transfer, where a statistical individual's willingness to pay (WTP) for an environmental improvement is constructed in the primary study as a function of the individual's characteristics such as gender, age, income, and other variables. Then this function is transferred to the policy site assuming the same relation between WTP and these characteristics. Another method available is so-called point value transfer, in which the mean WTP is simply transferred.

The usage of BT can potentially reduce the resource need for studies in line with the WFD and MSFD significantly, however also at the expense of increased uncertainties in the estimates.

Much previous research has been conducted also concerning BT (see overviews by Navrud & Ready, 2007; Johnston & Rosenberger, 2010; Hasselström et al., 2014; Kriström & Bonta-Bergman, 2014). However, a particularly relevant research gap in the light of the framing above concerns the potential trade-offs between precision in the transfer and the efforts needed to conduct the transfer.

1.2. Aims

Given the research gaps identified above, this thesis has three overarching aims:

1) To present and discuss monetary estimates for the value of environmental improvements concerning eutrophication in the Baltic Sea, both locally and on a Baltic Sea-wide scale, and reduced risk for oil spills in Lofoten-Vesterålen.
2) To investigate the feasibility of using a holistic, policy anchored scenario as a basis for valuation in relation to the WFD.
3) To contribute to a new discussion on potential trade-offs between precision and level of ambition in valuation studies through new case studies.

Below, the specific aims of the respective papers are described. Further background to the papers can be found in section 3.

<u>Paper I</u>

- To examine the feasibility of using an approach for estimating willingness to pay for marine environmental improvements based on a holistic, policy-determined scenario.

<u>Paper II</u>

- To test whether a best-fitting model for BT based on function transfer really performs better than a general model which is based on easily available data.

Paper III

- To test whether the detail-level of ecological information associated with the scenario description affects willingness to pay and valuation uncertainty for a water quality improvement.
- To explore the role of familiarity with the environment subject to valuation in relation to the effect of information level.

Paper IV

- To examine the recreational use of and public perception of the Baltic Sea among the inhabitants of the nine countries with a Baltic Sea coastline.

Paper V

- To estimate the willingness to pay for reductions of eutrophication effects in the Baltic Sea according to current political agreements.

Paper VI

- To explore public preferences connected to:
    - type of eutrophication effect – are some effects perceived as more important than others?
    - the role of time and lags between action and environmental improvement.
    - the spatial dimension – are improvements near the coast more important than improvements in the open sea and are some parts of the sea more important than others?

Paper VII

- To estimate the value of reducing the risk of oil spills from shipping in the Lofoten-Vesterålen area.

Paper VIII

- To investigate costs and benefits together, including financial and distributional effects, of reducing the risk for oil spills from shipping in the Lofoten-Vesterålen area (i.e. this paper builds further on the findings in paper VII).

The structure of the cover thesis is as follows: In section 2, a number of general theoretical concepts concerning monetary valuation of environmental change are outlined and discussed. In section 3, a more specific methodology is presented along with the respective papers, including the methods, results and conclusions in each paper and a comparison of results between some of the papers. Section 4 provides a broader discussion on the core of valuation and its role in relation to sustainability research and decision-making, and section 5 provides overarching conclusions.

2. Methods - microeconomics and valuation

This section outlines and discusses some theoretical and methodological concepts of valuation. The discussion is a selection of observations concerning valuation which are particularly relevant in the light of the studies presented in papers I-VIII. Section 2.1 starts from a rather broad view on what may constitute a value, and portrays the research in this thesis within a rather delimited definition of value. Section 2.2 discusses the concept of 'economic value'. Section 2.3 discusses usual motivations behind monetary valuation. Section 2.4 provides an overview of some of the usual valuation methods found in the literature. Section 2.5 discusses why the value of a marginal environmental change is a more suitable setup to study than the value of the 'environment as such'.

2.1. Value pluralism - delimitations

The Oxford dictionary defines 'value' as: "The regard that something is held to deserve; the importance, worth, or usefulness of something". Gómez-Baggethun et al. (2014) conclude, based on similar definitions that "valuation refers to the understanding of the worth or importance of something" (p.6). Clearly, this concept may include valuation in monetary terms, but also other forms of expressing something's worth or importance.

Norton and Noonan (2007) discuss the concepts of 'monism' vs. 'pluralism' in values related to the environment. They argue that while 'monistic' approaches, such as the monetary valuation of environmental change reported in the papers in this thesis, may have an advantage in that they can claim to be comparing comparables (e.g. dollars with dollars), they fail to take account for the nature of environmental problems as being multidimensional. Further, they argue that pluralistic theories on the other hand, which allow a multitude of expressions for value, may seem messy and confusing to interpret, but that environmental problems are in fact messy.

Values may be expressed in many ways. Kenter et al. (2015) present an overview of so-called 'shared values', including transcendental values, cultural and societal values, communal values, group values, deliberated values, other-regarding values, and value to society, as defined in Table 1. They argue 1) that there may exist a difference between individual values and collective values, and 2) that the relevant elicitation processes may differ for different types of values.

Table 1. Types of shared/social values. From Kenter et al. (2015).

Type of shared/social values	*Definition*
Transcendental values	Conceptions about desirable end states or behaviors that transcend specific situations and guide selection or evaluation of behavior and events (Schwartz and Bilsky, 1987).
Cultural and societal values	Culturally shared principles and virtues as well as a shared sense of what is worthwhile and meaningful. Cultural values are grounded in the cultural heritage and practices of a society and pervasively reside within societal institutions (Frey, 1994). Societal values are the cultural values of a society; societies may be more or less homogenous, so there may be multiple sets of cultural values in one society that overlap to a greater or lesser degree with each other.
Communal values	Values held in common by members of community (e.g., geographic, faith/belief-based, community of practice or interest), including shared principles and virtues as well as a shared sense of what is worthwhile and meaningful.
Group values (within valuation)	Values expressed by a group as a whole (e.g., through consensus or majority vote, or more informally), in some kind of valuation setting.
Deliberated values	Value outcomes of a deliberative process; typically, but not necessarily, a deliberative group process that involves discussion and learning.

Other-regarding values	As contextual values: the sense of importance attached to the well-being of others (human or non-human). As transcendental values: regard for the moral standing of others.
Value to society	Benefit, worth or importance to society as a whole.

Norton and Noonan (2007) conclude:

> ”If we seek an integrated and comprehensive system for evaluating environmental and ecological change, we must embrace and develop a pluralistic, but integrated, system of evaluation and policy. Such an integrated system of evaluation would of course involve economic indicators and considerations—but it would be pluralistic in the sense that it counts values other than units of human welfare measured in terms of aggregated WTP. The pluralistic approach subsumes the monistic valuation exercise.”
>
> Norton and Noonan (2007, p. 667)

The following parts of this cover thesis focus on a traditional microeconomic understanding of value, being based on preferences of individuals rather than collective value formulations. This implies delimitation in relation to the many types of values exemplified above.

2.2. What is an economic value?

2.2.1. Trade-offs

A ‘value’ is, as indicated in the previous section, not always the same thing as an ‘economic value’. While economists tend to use the two terms interchangeably (which is also the case in this thesis), the everyday use of the word is of broader character, including a large amount of possible interpretations. In economics, the term is used in relation to a rather strict theoretical framework. The starting point is trade-offs that an individual is willing to make. For example, an individual makes daily trade-offs in consumption choices. Valuation of ecosystem services can be seen as a study of which trade-offs individuals are willing to make in relation to the environment.

Environmental valuation methods aim to derive a value of an environmental change (leading to a change in the supply of ecosystem services) based on clues concerning these trade-offs. For market goods and services, consumption choices tell us something about people's valuations, but for other goods and services, it might be more difficult. Changes in environmental quality is in many cases off-market, which means that the use of prices for providing information to valuation is not possible.

A key prerequisite for valuation is that there is a willingness to make trade-offs. In some situations, individuals might be unwilling to make any trade-offs. For example, the resource might be seen as having an infinite value ("invaluable"). In this situation, the key prerequisite for valuation – a willingness to make trade-offs – is not fulfilled. Under these circumstances ('non-substitutability'), the microeconomic approach to valuation is fruitless. This has implications for the types of studies that are relevant within a microeconomic perspective – see further discussion in section 2.5.

2.2.2. The endowment effect

Apart from substitutability, an additional aspect to consider is the difference between the willingness to pay (WTP) for gaining something and the willingness to accept compensation (WTA) for giving something up. Kahneman et al. (1991) illustrate this phenomenon with a simple example.

> "A wine-loving economist we know purchased some nice Bordeaux wines years ago at low prices. The wines have greatly appreciated in value, so that a bottle that cost only $10 when purchased would now fetch $200 at auction. This economist now drinks some of this wine occasionally, but would neither be willing to sell the wine at the auction price nor buy an additional bottle at that price."
>
> (Kahneman et al., 1991, p. 194)

This example suggests that there is a difference between WTA and WTP. One can assume that this wine collector would in principle be willing to sell a bottle of wine given some proper price (WTA). And further that he or she would be willing to purchase an additional bottle of wine given some proper price (WTP). The example illustrates that there may be a difference, for the same good, in WTP and WTA. In this example the auction price is lower than WTA and higher than WTP, suggesting also that WTA is higher than WTP.

The same pattern may exist for an environmental change. In a level of environmental quality z1, an individual may be willing to accept a project leading to a reduced environmental quality z0 given sufficient compensation (WTA). On the contrary, assuming an environmental quality z0 as a starting point and a project leading to an improved environmental quality z1, this individual may be willing to pay in order to realize the project (WTP). It is however fully possible that WTP for going from z0 to z1 would be lower than WTA to go from z1 to z0.

Thaler (1980) study this type of loss aversion (‘endowment effect’), and further discussion on the size of this potential gap and how to understand the gap can be found in e.g. Haneman (1991), Kahneman et al. (1991), Zhao & Kling (2001) and Plott & Zeiler (2005). The studies in this thesis are based on studying WTP and not WTA. The reasons for this are of a practical nature. The first is that the projects being evaluated are all based on scenarios for environmental improvements compared to the current state. While it would be possible to formulate a question for WTA to refrain from an environmental improvement, such a scenario would be more difficult to communicate to the respondents. The second is that WTP has the advantage of being limited by a budget restriction, which WTA is not. This makes it possible to at least somehow validate the results (in relation to respondents’ incomes).

2.2.3. Utility functions

Formally, the willingness to make trade-offs is a consequence of substitutability of goods. Equation (1) presents a standard utility function that may help explaining this substitutability[1,2].

$$U = f(x_{1;}\, x_2; \ldots ; x_n; B) \qquad \text{(eq. 1)}$$

[1] The variable names sometimes differ between various authors. Sometimes Z is used for environmental quality, sometimes B is not included in the functions for various reasons concerning temporal aspects. Here, a version is presented that suits the thesis the best.

[2] The function is not consistent with ‘lexicographic’ preferences, in which an individual would prefer any amount of one good to any amount of another. For example, this might imply preferring one apple and one pear over zero apples and an infinite amount of pears.

In Equation 1, U represents the utility level of an individual, where utility could be seen as synonymous with ‘happiness’, ‘well-being’, or other similar words. U is a function of current consumption of various goods or services; x1; x2; x3, and so on. Importantly, these goods or services are not necessarily strictly traditional ‘consumption items’ but may also represent other utility creating entities such as healthcare, leisure time, or a good environment. B represents remaining budget, which in theory equals future consumption possibilities. This equation implies that there may exist possible substitutions between good 1, good 2, good 3, and so on that still leave the individual on the same resulting utility level. For example, it might be the case that this individual is as well off with 2 pears and 5 apples as with 3 pears and 3 apples. In that case, the individual would have nothing against trading away 2 apples for an additional pear, and the economic value of the two apples for this particular individual can be derived.

In environmental valuation, environmental quality is treated as another of those goods and services, and the valuation is based on studying the sacrifices the individual is willing to make to achieve an improvement in environmental quality, or the compensation he needs in order to be willing to accept a degradation in environmental quality. This implicit ‘commodification’ of the environment is sometimes seen as problematic (e.g. Baggethun & Pérez, 2011). This is a broad issue with an ethical dimension. While it is possible to argue that this ethical dimension is non-economic in its nature, and hence that such discussions can be used to complement economic analysis, there might be problems also within the economic dimension due to the substitutability aspect. When the environment is associated with an infinite value, a valid argument is that it simply cannot be substituted as any other good. For example, individuals might not be willing to accept a project that would lead to the complete drying out of the Baltic Sea, for any amount of compensation. However, while the ‘environment’ as such, or a very dramatic environmental change, may be associated with an infinite value, a more marginal environmental change is in many cases more reasonable to discuss in the light of trade-offs and utility functions. The papers in this thesis are based on studying the value of a marginal environmental change, rather than the value of the ‘environment’ as such. This issue is discussed further in section 2.5.

2.2.4. Different types of economic values

An environmental change is related to many types of economic values. A commonly used concept for describing these values is TEV, Total Economic Value (Pearce & Turner, 1989). It has some important implications.

TEV is seen as the sum of different types of values generated by an environmental improvement. TEEB (2010, p.195) provide an overview, and the description below is based on their framing. Most importantly, value consists of both *use values* and *non-use values*. Use values are divided into direct and indirect, where direct use implies e.g. the benefits generated from food production and recreation, and indirect use implies e.g. benefits generated from functions of the ecosystem such as pest control, pollination, water regulation, etc. There are also option values, representing future use values. Non-use values are represented by existence values (satisfaction of knowing that a species or ecosystem exists), bequest value (satisfaction of knowing that future generations will have access to nature's benefits), and altruist value (satisfaction of knowing that other people have access to nature's benefits).

An ecosystem service generates economic benefits through the usage of it, but also through its mere existence. People attach a value to the knowledge that an ecosystem is in good shape, even though they might never even visit or anticipate visiting the specific area. This idea of non-use values was introduced by Krutilla (1967). Studies such as Carson et al. (2003) on the value of protecting the coastlines against oil spills, as well as the papers in this thesis, show clear evidence of non-use values (see especially paper V and VI).

Non-use values, including existence values, are not necessarily the same thing as *intrinsic values*. Sandler (2012) provides deeper insights based on literature studies. He contrasts intrinsic values with *instrumental values*, where the former relate to 'the value that an entity has in itself, for what it is, or as an end', while an instrumental value is 'the value that something has as a means to a desired or valued end'. (p.1). Sandler continues with concluding that there are two quite contrasting views on intrinsic values – in one of the views, it is created by human valuing. He calls this the view of *subjective intrinsic value*. The contrasting view could be called the view of *objective intrinsic value*, in which intrinsic values are seen as something that is not humanly conferred. This would imply that the value is independent of any human's attitudes or judgments. Related discussions can be found in e.g. Moore (1922), O'Neill (1992), and Vilkka (1997)[3].

A conclusion from the above-mentioned literature is that the concept of intrinsic values is challenging. However, importantly, given the possibility of objective intrinsic values, it becomes clear that economic valuation methods cannot be used to measure them.

2.3. Why valuation?

The market economy is often seen as blind to the value of ecosystem services. For example, it may at least in the short term be profitable to build new houses in urban green areas and agriculture might profit from using fertilizers containing nutrients which eventually leak into the sea and cause eutrophication effects such as dead sea bottoms and intense cyanobacterial blooms. A common economic interpretation of this market blindness is the concept of *externalities* – the production or consumption of something implies negative or positive spillover effects on other parts of society (e.g. Pigou, 1920). However, while the market economy is usually blamed for causing environmental degradation, it is also dependent on ecosystem services – in the case of the marine environment this is particularly obvious for tourism and fishery.

Our dependence on healthy ecosystems requires that we protect the environment against our human pressures from everyday life. Usually it is argued that we need policy intervention since private incentives are insufficient. Environmental commitments by governments have a long history. For example, NPTEL (2012) note an Indian treatise *Arthashastra* dating to 300 BC that prescribed punishments for polluters, suggesting an early appreciation of environmental values.

Taking measures to prevent environmental degradation or to improve the state of the environment often requires investments or other types of costs. However, there are also benefits, and by pointing out the monetary value of these benefits, the costs can be seen in a broader economic context. A common argument for economic valuation of environmental

[3] Intrinsic values are sometimes treated synonymously with 'final' values, i.e. something that is valuable 'as an end', 'for its own sake', rather than for the sake of something else. See e.g. Rabinowicz and Rasmussen (1999) for further discussion (who acknowledge that 'intrinsic' values are 'final' but argue that there are 'final' values that are not 'intrinsic').

change is that it assists inevitable trade-offs between costs and benefits. Hanley & Barbier (2009) use two examples in their introduction to clarify what this may mean:

- “Deciding on whether to introduce or reform a particular government policy, such as introducing a new energy tax; or
- Deciding on whether to go ahead with a particular investment project, such as a new motorway or hydroelectric scheme.”

Hanley & Barbier (2009, p.1)

These decisions may be informed by using cost-benefit analysis (CBA). Many projects or policies have an impact on the environment and in order to be able to paint a richer picture in a CBA of the pros and cons with a policy or project, information on the value of the impact on the environment (being positive or negative) is useful.

Another motivation for environmental valuation is to help internalizing externalities. This thought relates back to Pigou (1920) who advocated taxes on emissions of pollutants based on the value of the damage they cause. When externalities are not properly accounted for, negative impacts to the environment from production or consumption become invisible to the market. This relates to the well-used concept ‘Polluter Pays Principle’.

A third motivation is the concept of ‘green accounting’. It is commonly accepted that GDP is not a sufficient measure of welfare development over time. There are several reasons for this, and one of them is that it only partly accounts for changes in natural capital. A recent report commissioned by the Swedish Government (SOU 2013:68) recommends the government to improve the visibility of ecosystem services in the yearly national accounts. Monetary valuation might be a useful tool, particularly for studying welfare changes over time due to marginal changes in the supply of ecosystem services.

A fourth motivation is to help governments make trade-offs between different environmental problems that need attention. For example, paper VI studied the preferences for reducing different types of eutrophication effects in the Baltic Sea, such as cyanobacterial blooms and low water transparency. To some extent, different nutrients control different eutrophication effects and knowledge on the preferences of the public may be a factor that should feed into discussions on nutrient abatement targets for the Baltic Sea, for different types of nutrients. Another recent example related to the Baltic Sea, and perhaps exemplifying a fifth motivation to valuation, is paper V, in which it is concluded that the citizens of some of the Baltic Sea countries seem to value water quality higher than the citizens in other Baltic Sea countries. This could be important knowledge for policy-makers preparing for international negotiations. Scharin et al. (2016) emphasize that in order to incentivize international participation in negotiations, costs for measures should be distributed such that all countries feel that they have something to gain from taking measures, which would imply allocating more costs to countries that benefit the most from an improved state of the environment subject to the negotiations.

2.4. Valuation methods

As described in section 2.1, an attempt to capture the economic value of environmental change needs information on the trade-offs individuals are willing to make. Economic valuation of environmental change can usually not be made without knowledge about underlying functions of the ecosystem. Provided that this kind of knowledge exists to some extent, two standard approaches are available for non-market valuation. *Revealed Preferences* (RP) and *Stated Preferences* (SP). See e.g. Champ et al. (2003) or Freeman et al. (2014) for an overview.

Common for both RP and SP methods is that they aim to result in an estimate of WTP or WTA associated with a specified change in environmental quality or quantity. These terms represent measurements of the economic value of the change. RP methods derive the value of ecosystem services from observed (revealed) behavior. For example, costs spent on travelling to a certain recreational area may be used to provide information about the willingness to pay for visiting the area ("Travel Cost Method"). In principle, the visitors are willing to pay at least the amount they have already spent. Vesterinen et al. (2010) is an example of a travel cost study where information about recreational travels to different lakes in Finland was used to estimate the WTP for improved water quality. The key to the method is the variation in water quality over the country. Evidently, people are willing to travel further or more often to locations with good environmental quality. This is a reflection of a value and it can be measured using behavioral clues.

Other examples of RP methods are the Hedonic Pricing Method where the price of a good such as housing is assumed to depend on a number of variables, among these environmental quality; and the Defense Expenditure Method, which uses people's expenditures to protect themselves from negative impacts resulting from environmental degradation. An example of the latter method could be to study the expenditures on water filters to clean drinking water coming from a contaminated groundwater source.

Similarly, the relation between the state of an ecosystem and its ability to produce market goods can be used to value changes in environmental quality ("Production Function Method" [4]). For example, market values of fish may be used to estimate the value of an ecosystem that supports fish production. If the ecosystem is disturbed in some way, the production of fish might be reduced, and some of the value forgone due to this deterioration is reflected in the market for fish. This approach was used by Paulsen (2007), who estimated the potential cost of algal mats – an undesirable consequence of eutrophication since it disturbs the nursery grounds for plaice – using clues on the phenomenon's effect on profits in plaice fishery. The study is based on an ecological model, linking the quality of habitat to changes in the plaice population, and an economic model linking fish recruitment with fisheries' profits over time.

SP methods are to some extent more direct. These methods rely on the creation of a hypothetical market situation using surveys to ask respondents about their willingness to make trade-offs (preferences), often in terms of a willingness to pay for fulfilling a scenario in which the environment improves in some way. For example, a scenario can be set up where it is explained that the quality of the coastal bathing water is dependent upon the amount of nutrients in the water, and that the effluents of these nutrients can be reduced by improving the capacity in sewage treatment plants. This, however, would be costly, and financing is required. The scenario could suggest that e.g. municipal taxes are increased to finance the policy measure, and the respondents are then asked about their willingness to pay to realize the scenario.

There are two main types of SP methods, the Contingent Valuation Method (CV) and Choice Experiment (CE). These methods are more similar than different – the main difference lies in the elicitation format of WTP. CV is a more straightforward approach where respondents could for example be asked how much they would be willing to pay to realize a certain scenario, whereas in CE, respondents are given choice options, each including scenarios for changes in various environmental attributes

[4] The production function method is usually seen as being outside the RP group since it is based on market valuation. It is included above since the methodology resembles RP methods to a large extent.

and a cost to be (hypothetically) paid by the respondents. The respondents are then asked to choose which of the options they prefer. A further description of SP methods is available in section 3.2, and a thorough overview of SP methods can be found in Meyerhoff et al. (2007).

Apart from these methods, there are also other approaches available, however some of them being less rooted in economic theory. One example is the replacement cost method, in which the cost to replace an ecosystem service with other means is estimated. For example, if a wetland is ditched it will lose water treatment capacity. This capacity might be replaced by a man-made water treatment plant. However, using the cost of the treatment plant as a measure of the value of the wetland’s water treatment capacity is problematic and would only be a valid measure of economic value if (i) the man-made water treatment system is equally good at treating water as the wetland, and (ii) the man-made water treatment system is the cost-effective way of replacing the wetland’s water treatment, and (iii) people would in fact be willing to pay the costs for the replacement system if the ecosystem service is no longer available (Freeman et al., 2014). In this sense, the methodology is different from the defense expenditure method since the latter method is studying choices by individuals that have already taken place.

2.5. Marginal values

An important discussion in relation to environmental valuation is the concept of *marginal values*. In a heavily publicized and debated report, Costanza et al. (1997) estimate the total value of the world’s ecosystems by transferring estimated ecosystem service values from different regions and studies and aggregating them on a global level. A critique of this approach is Bockstael et al. (2000) who argue that this methodology is inconsistent with economic theory since the results of the many primary studies should be seen in the light of the particular environmental change of that particular study, holding constant all other things, including the level of other ecosystem services. Further, as Daily et al. (2000) point out, environmental valuation works best ‘when increments are small, so that a change in one service will have minimal feedbacks through the rest of the system’ (p.396). For example, when changes are large, there may be secondary effects on other markets which are difficult to predict.

Besides this specific methodological critique, the whole question on the total global value of ecosystem services may be seen as irrelevant relating to economic theory. Valuation is about studying trade-offs that people would be willing to make. The results from Costanza et al. (1997) do not really suggest that there is a specific amount that we could be compensated with in order to be willing to give up these services, nor do they suggest that if we ever end up with zero ecosystem services, the study's results could be used as a measure of the willingness to pay to restore the ecosystem to today's level. However, Costanza et al. (2014) address this type of critique, arguing e.g. that these measurements have an accounting role similar to that of GDP, and that the measurements can be used to study changes in well-being over time.

3. Applications to the marine environment

3.1. Introduction

The Baltic Sea is one of the largest bodies of brackish water in the world, and it is, as highlighted in the introduction to paper IV, also one of the most threatened marine environments in the world. The Baltic connects with the North Sea only through the narrow straights in the Belt Sea and the Kattegat channel and the water exchange is slow. Nine countries have coastlines to the Baltic: Denmark, Germany, Poland, Russia, Lithuania, Latvia, Estonia, Finland and Sweden, and approximately 85 million people live in the catchment area. The sea suffers from several anthropogenic pressures, such as hazardous substances, overfishing, global warming, acidification, physical disturbances to the sea bottom, underwater noise from shipping, habitats degradation and, one of the most highlighted pressures, eutrophication (Helcom, 2010).

Eutrophication is caused by an excessive amount of nutrients flowing into the sea, which implies changes in the ecosystem. As described and referenced in paper I, some of the effects of eutrophication are decreased water transparency, increased growth of filamentous macro algae, increased blooms of cyanobacteria, deterioration of habitats, and changes in fish species composition.

In order to tackle deteriorations of the marine environments (as well as inland waters) in the EU, there are in particular two important EU-directives: The Water Framework Directive (WFD; European Parliament, 2000) and the Marine Strategy Framework Directive (MSFD; European Parliament, 2008). These directives require member states to classify marine and inland waters with regards to their environmental/ecological status[5], and to take measures in order to reach good environmental/ecological status. Further, there is a political (non-binding) agreement between the nine Baltic Sea countries, the Helcom Baltic Sea Action Plan (BSAP; Helcom, 2007).

The development of the Arctic region is of interest both from an ecological and from a socioeconomic perspective. While the region is highly important for global biodiversity, the Arctic is also an area of strategic interest for the extraction of natural gas and oil, fisheries, tourism, shipping, etc. (AGP 2010). An increasing global scarcity of vital natural resources, advances in extraction and transportation technology and global warming will change the stakes of nations and other actors in the area. Trade-offs between different financial, social and ecological interests are unavoidable, and the crucial challenge is to find trade-offs that are consistent with sustainable development. To better assess these trade-offs there is a need for, among other things, better information associated with the social costs of environmental degradation.

In papers I-VI, various aspects concerning the public's preferences in relation to eutrophication effects in the Baltic Sea are studied. Papers I-III focus on the coastal environment in two case-study areas in Sweden: Himmerfjärden south-west of Stockholm on the east coast and the areas around the islands of Orust and Tjörn north of Gothenburg on the west coast. Papers IV-VI focus on the Baltic Sea as a whole. Except from paper IV, all papers are based on SP data. Paper IV is a study on use of and attitudes towards the Baltic Sea, based on a survey in the nine countries. This study was made partly as preparation for the valuation study reported in paper V, but with interesting results in itself. Papers I-III are more oriented towards presenting and discussing methodological issues, while papers IV-VI are more oriented towards presenting statistics and estimates.

[5] And chemical status, in the case of WFD.

Papers VII and VIII focus on the risk for oil spills in the Lofoten-Vesterålen area in the Barents Sea. Paper VII presents an SP study and paper VIII utilizes these results, along with complementing data, in a CBA for the reduction of probability for accidents leading to oil spills from shipping.

Section 3.2 provides a deeper description of the method related to SP studies. Section 3.3 to 3.5 reports the background, method and results for papers I-III, IV-VI, and VII-VIII, respectively.

3.2. Methods – stated preferences studies and benefit transfer

As described in section 2, SP studies are based on the creation of a hypothetical market situation. The studies reported in the papers (except paper IV) are based on SP, in which respondents to a survey, one way or another, are asked to state their preferences towards an improvement of the marine environment. In papers I, III, V, VI, and VII, contingent valuation (CV) is used, in which the respondents are asked to state their willingness to pay directly (WTP), either by using an open-ended question (papers I, III and VII) or by using a so-called payment card with fixed intervals (papers V and VI). In paper II, a Choice Experiment (CE) is instead used. The WTP is then derived statistically from the survey data. Below, a number of aspects in relation to study design are summarized.

3.2.1. Sample

First of all, a target population needs to be established. Theoretically, the population should include everybody who might have a WTP for the change of environmental quality or quantity that is subject to the study. For example, if one believes that a specific environmental issue is of national interest, the population (and correspondingly the sample) should theoretically be national. A too delimited population may imply that the WTP, when aggregated from the sample to the population, is underestimated. However, for practical reasons the population (hence also the sample) is often more delimited. Importantly, SP studies capture both use values and non-use values, which implies that a delimitation to a population of users, such as visitors to a specific place, might lead to a downward bias of the results given that others may attach non-use values to the studied scenario.

Secondly, as for survey studies in general, the selected sample needs to be representative if the purpose is to use the results for aggregation, otherwise the results will be biased. The desired size of the sample should be decided in relation to the need for precision and the need for statistical analysis.

A note in relation to sampling is that the data in papers I-III and VII-VIII, and the Swedish data for papers V and VI was based on web surveys using a web panel provided by a survey institute. The use of web panels in these surveys implies some different sample characteristics compared to a 'usual' random sample:

- Panelists are invited beforehand (randomly selected invitations) to participate in the panel, and are regularly asked to respond to surveys.
- No information is given on the topic of the survey until the respondents enter the questionnaire.
- Panelists are compensated for their efforts by the survey institute.

Concerning the statistical quality of web panels vs. postal-based surveys, the issue was investigated in Östberg et al. (2010), which is a working paper prior to the publication of paper I:

> "Regarding representativeness and selection issues, all kinds of survey methods are likely to have their specific selection and representativeness problems, and web panels might for example suffer from a) bias towards types of respondents who have internet access as well as b) bias towards respondents who like/have time to respond to a lot of surveys. Problem a) is not likely to be of great magnitude in Sweden, since a large share[6] of the Swedish households have internet access in their homes, but problem b) is a fact which one should bear in mind. However, this problem is likely to exist in some extent also for other types of survey methods. Regarding comparisons in the literature between different survey modes, Olsen (2009) tests for survey mode effects in a CE setup and

[6] Around 85 % of Swedes had access to Internet in 2010 (Findahl, 2011).

shows that WTP estimates in internet surveys don't differ significantly from mail surveys. Denscombe (2006) and Deutskens et al. (2006) also make this conclusion. Lindhjem et al. (2008) compares, in a CV setup, in-person interviews with an Internet survey and concludes that WTP estimates do not seem to differ significantly between the two survey modes. As of potential effects from the compensation to respondents, they also conclude that the compensation incentives do affect the number of respondents and their response times, but do not affect what the respondents answer. Dennis (2001) investigates the effect of responses from panel members vs. non-panel members and concludes that the panel effects are minimal."

(Östberg et al., 2010 p.8-9)[7]

3.2.2. Survey design

There are a myriad of details concerning survey design, and many studies have tried to estimate the effects of different formulations in the questionnaire. See e.g. Meyerhoff et al. (2007) for an overview and the papers in this thesis (especially papers I and III) for deeper literature

[7] References to articles in Östberg et al. (in order of appearance):

- Olsen, S.B. (2009) Choosing between internet and mail survey modes for choice experiment surveys considering non-market goods, Environmental and Resource Economics. 44 (4), pp. 591-610.
- Denscombe M. (2006) Web-based questionnaires and the mode effect, an evaluation based on completion rates and data contents of near-identical questionnaires delivered in different modes, Social Science Computer Review. 24 (2), pp. 246-254.
- Deutskens E., (2006) An assessment of equivalence between online and mail surveys in service research, Journal of Service Research. 8 (4), pp. 346-355.
- Lindhjem H. & Navrud S. (2008) Internet CV surveys – a cheap, fast way to get large samples of biased values?, MPRA Paper 11469, University Library of Munich, Germany.
- Dennis J.M. (2001) Are internet panels creating professional respondents? A study of panel effects, Marketing Research. 13 (2), pp. 34-38.

review. Below are a number of overarching aspects concerning survey design.

Carson and Hanemann (2005) studied best practices of CV studies and concluded that a typical survey instrument (questionnaire) includes:

> “(1) an introductory section identifying the sponsor and general topic, (2) a section asking questions concerning prior knowledge about the good and attitudes toward it, (3) the presentation of the CV scenario including what the project was designed to accomplish, how it would be implemented and paid for, and what will happen under the current status quo situation if the project were not implemented, (4) question(s) asking for information about the respondent’s WTP/WTA for the good, (5) debriefing questions to help ascertain how well respondents understood the scenario, and (6) demographic questions.”
>
> Carson and Hanemann (2005, p.825)

Concerning the scenario description, its role is to describe the good subject to valuation, i.e. the hypothetical change in environmental quality or quantity that is being studied. In order to properly describe this hypothetical change, the reference alternative, or status quo alternative, needs to be formulated, along with a description of the expected results of some suggested measure (e.g. a policy, plan, legislation or project). Paper I presents a deeper discussion concerning this description, highlighting the importance of anchoring the scenario description in proper ecological research as well as in a relevant policy context. Further, time is an important variable. When can we expect the effects to be realized and what is their expected duration?

In association with the scenario description, the payment vehicle needs to be specified, i.e. a description of how the payment would be collected (e.g. through the municipal tax or through a one-time payment to a certain organization, etc.). Further, the conditions for “delivery” of the project is often seen as important to include in order to make the scenario realistic. For example, one way of formulating this is to ask the respondent to assume that the project will take place given that the total sum of WTP exceeds the costs of the project (see e.g. Brännlund and Kriström, 2012).

The *elicitation format*, i.e. the way to derive WTP varies between studies. As noted, a difference between CE and CV is that CE is oriented towards more indirectly deriving WTP through a setup of different choice situations, whereas CV is more direct. Concerning CV, however, there are a number of different formats available, such as open-ended ("what is the maximum amount you would be willing to pay to...") and closed-ended ("would you pay X SEK to...") questions. Additionally, the interval open-ended question (Håkansson, 2008) is available, in which the respondents state their WTP as any chosen interval. Interval open-ended elicitation was used in papers I, III, and VII[8], payment card in papers V and VI, and closed-ended CE elicitation in paper II.

3.2.3. Hypothetical bias

Hypothetical bias refers to a potential gap between a respondent's stated WTP and his or her real WTP (or similar for WTA). An obvious problem with stated preferences studies is that they are based on hypothetical situations, and a respondent might have chosen differently in a real situation with real transactions. A large number of studies have been made on hypothetical bias, and on methods to avoid this bias and make the respondents answer 'as if' they were faced by a real and not hypothetical choice. An early study was made by Bishop and Heberlein (1979) on the WTA to sell hunting permits for goose. In one sample, respondents were given hypothetical purchasing offers in a closed-ended CV setup. In the other sample however (also on hunters), real cash offers and real transactions were used, implying a possibility to compare the results. The mean WTA was approximately 60 % higher in the hypothetical situation than in the situation with a real transaction.

Since then, a number of similar studies have been conducted to test the existence and size of hypothetical bias, as well as methods to reduce this bias. A literature overview in Freeman et al. (2014, p. 401) suggest that the majority of the literature note exaggerated values in SP studies. A popular method to reduce this bias is "cheap talk", in which survey participants are somehow reminded to treat the choice situation as if it were a real choice. In the papers in this thesis, standard types of cheap

[8] In paper VII, the respondents could choose to present their WTP as an interval or as a point value.

talk scripts were used, in which the respondents are reminded about the budget constraint and asked to answer as if it were a real choice situation. The effects of this type of addition in the survey instrument is debated and different tests have generated different results as summarized by Svensson (2009). Further, Svensson concludes that there is yet no theoretical basis for understanding how, when and why cheap talk might work.

3.2.4. Benefit transfer

Benefit transfer (BT) is in principle a way of avoiding costly primary studies when the value of an environmental improvement needs to be estimated. Instead, the method is based on 'simulating' a primary study in the area (policy site), using results from existing studies, such as CE or CV at another site (study site).

The literature generally differentiates between two main methods for BT (cf. Navrud, 2007; Kriström & Bonta Bergman, 2014). Unit value transfers are based on transferring a single value from a similar study site, while function transfers instead transfer a WTP function. For example, a unit value transfer may imply that an estimated mean WTP is used as a basis for aggregation to another population with the same assumed mean WTP. A function transfer on the other hand implies that the WTP at the study site is estimated as a function of variables among the population such as age, gender, education, usage of the specific resource and so on, and that function is then used for estimating WTP at the policy site, based on the specific characteristics of the population at the policy site with respect to these variables. The latter method has been said to allow for transfers to sites that are less demographically similar (Navrud & Ready, 2007).

3.3. The role of policy in valuation study design (papers I, II and III)

This section presents papers I-III. In section 3.3.1, the overarching research questions for the papers are outlined, and then a separate section follows for each of the papers.

3.3.1. Overarching research questions

Section 2 discussed some overall concerns related to valuation in general. This section instead discusses some more operational concerns:

- How to define a suitable environmental improvement to assess.
- Given that definition, what are the trade-offs between assessment quality (precision) and cost and time required for assessment?

Water quality problems across Europe has led the EU to respond by introducing two directives – the Water Framework Directive (WFD) and the Marine Strategy Framework Directive (MSFD). According to these directives, water quality has to be classified according to a quality ladder and Good Ecological/Environmental Status (GES) has to be reached by 2020 or 2021, respectively. Further, the directives require socioeconomic assessments of programs of measure to reach GES. This speaks for a need to conduct economic valuation studies. The studies have to be relevantly designed in order to inform decision-makers about the benefits of reaching GES, and further, due to the many water bodies and coastal water areas in the EU and Sweden, the cost of conducting studies is a significant factor. These issues motivate the three papers.

The value of what? (paper I)

The choice of environmental change to study is not always aimed at providing policy relevance. For example, there are a number of studies in the literature estimating a WTP for environmental change such as “improved biodiversity”, which really does not make much quantitative sense. In order to provide relevance for decision-making concerning trade-offs, the variable to which a value is supposed to be attached needs to be a key factor in the planning of a valuation study (cf. Söderqvist & Hasselström, 2009).

Further, if a CBA is to be performed, the variable needs to be linkable to information about costs. This is a long chain of events in practice, implying many uncertainties. A measure has to exist, and the quantitative link between the measure and its effect on an environmental variable has to be possible to assess. Further, the cost of the measure has to be possible to assess. Given this information, changes in the variable needs to have some sort of direct or indirect effect on benefits (cf. Boyd, 2007; USEPA, 2009 on *ecological endpoints*).

When measures are taken to reduce the nutrient loads to tackle eutrophication, a number of effects can be expected in the ecosystem. For example, actions to reduce effluents of nutrients are likely to affect several ecological indicators together, such as water clarity, algae growth and vegetation. A valuation of one of these indicators separately would thus just indicate a fraction of the economic effects of the nutrient reductions. However, a summation of valuations of separate indicators that are correlated, such as water clarity and algae growth, would lead to double-counting risks. Hence, a more correct value estimate would be provided by an approach which presents all these indicators together as a holistic “package” corresponding to the expected results of a policy program.

There are several existing valuation studies regarding water quality in Northern Europe, utilizing or touching on a holistic scenario construction as presented in paper I (e.g. Hanley et al., 2006, Magnussen, 1992, Frykblom, 1998 and Kosenius, 2010). However, none of these papers have an explicit focus on a thorough description of environmental quality change as described by the national implementation of WFD. Paper I therefore aims to illustrate and test such scenario development, being based fully on environmental quality objectives rather than more traditional pollution reduction targets.

Level of ambition (papers II and III)

The great information need in relation to the requirements of socioeconomic analyses of measures to fulfil the WFD and the MSFD suggests a need for cost-efficient information gathering processes. The studies presented in Paper II and III suggest possible trade-offs between precision and cost of a study.

Among methodologically-oriented economists, the research in relation to valuation has often been aimed at improving statistical methods and developing as informative valuation scenarios as possible. While this research has been important to advance science into providing credible results, the difference between e.g. different econometric procedures in their results on value estimates may be small while the difference in efforts needed may be large.

The basis of Paper II is BT. The method by default implies uncertainty. However, in controlled settings, the validity of BT can be tested. This needs a primary valuation study in both the study area and the policy area. The results from different transfer methods can then be tested against the result from the primary study. Such transfer tests tend to result in a transfer error in the range of 25-40 %, but in some cases, the transfer error is as large as several hundred percent (Navrud, 2007). One of the factors that tends to lead to lower transfer errors is the similarity between the policy site and the study site.

When using function transfer, WTP is explained by a number of variables. As noted by Spash and Vatn (2006), a discussion about why certain covariates are included is often missing in BT studies. Most of them only include a few standard socio-economic variables (e.g. age, education, income, and gender) but provide no motivation for choosing these. Instead of choosing covariates on an ad hoc basis, some kind of decision rule or theoretical motivation should be applied. To examine this matter further paper II compares a "best-fitting" model, that is, a model that gives the best statistical fit but requires the collection of detailed information (e.g. use of or familiarity with the area in addition to demographic variables), with a "general" model that only requires the collection of readily obtainable demographic data at the policy site. How much more precise is the BT when using a best-fitting function compared to when using a function based on only easily available data such as general demographics? Or conversely, how much precision is lost when a model based on easily available data is used instead of the best-fitting model?

Paper III further studies the potential trade-offs between precision and cost-efficiency in valuation by approaching the issue of ecological information given to respondents in a CV survey. Earlier research has shown that the quality and quantity of information in SP studies is important for the understanding of the valuation scenario and affects WTP (e.g. Ajzen et al., 1996; Baron & Maxwell, 1996; Aadland et al., 2007). Arrow et al. (1993, p. 32–33) provide recommendations about information that should be provided in CV-studies, and state that ''adequate information must be provided to respondents about the environmental program that is offered''. However, the meaning of 'adequate' is likely to be context-dependent, and although much research has been devoted to this subject, there is no consensus regarding information that should be provided in terms of describing the 'good', 'environmental change' or 'valuation scenario'. The provisioning of a detailed scenario description to the respondents requires time and resources. How much precision is lost when a simple scenario description, however still anchored in a rather well-defined environmental change, is used?

3.3.2. Paper I – Non-market valuation of the coastal environment – uniting political aims, ecological and economic knowledge

Background

Knowledge on the benefits of measures taken to reach Good Ecological Status (GES) connected to the WFD is needed. While GES is a rather holistic concept, including a number of well-defined parameters, much of the valuation literature on water quality is based on single indicators for environmental improvement (such as improved water transparency) and/or an unspecified environmental change (such as "improved biodiversity" without much further specification of what this means). Using these kinds of estimates as inputs to a cost-benefit analysis is difficult. A valuation based on a single indicator implies a too limited scenario since measures are likely to affect many environmental characteristics simultaneously. A valuation based on an unspecified amount of change means that it is difficult to compare benefits and costs. In this study, a valuation scenario is developed which is holistic in terms of covering many eutrophication effects simultaneously, and well-specified in terms of the studied environmental change, where the status classes of the WFD are used as a backbone for the scenarios.

The study is based on two case study areas: Himmerfjärden south of Stockholm and the area around the islands of Orust and Tjörn north of Gothenburg. Both areas are well-used for recreational purposes such as walking, bathing, boating and fishing. However, both areas are also eutrophied and none of the areas reach the requirements for GES today.

Method

The study is based on a CV survey distributed to a web panel supplied by the survey company Norstat. The panels consist of randomly selected adults who have agreed to participate in the panel, thus regularly getting queries for surveys regarding different topics. Two scenarios were used – improved water quality and less noise and littering, as illustrated in Appendix A of the paper. The scenarios were developed with the help of ecologists with good knowledge of the local conditions in our study areas. The suggested policy actions for improving water quality and reducing noise and littering were developed considering realism and credibility among the respondents, with assistance from ecologists and the county administrative boards in the counties of Stockholm and Västra Götaland.

Water quality at present in different parts of the study areas, as well as the expected water quality if measures are taken, was presented using five quality classes based on water clarity (measured as “Secchi depth”) and the different types of vegetation typical for each class. The respondents were shown underwater photographs chosen to be representative for each of the water quality classes, textual descriptions of each quality class, and maps.

For noise and littering, status quo was defined as “no specific policy action is taken against the problems”. The improvement scenario was based on the introduction of so-called special consideration zones (SCZs) in certain parts of the study areas. Three SCZs would be introduced on the West coast and three on the East coast.

An implementation of these policy actions would imply a) two classes improvement in water quality from the present quality level in the respective parts of the study area, or b) less noise and littering as described by the restrictions in the SCZs. These restrictions involved several aspects, e.g. speed restrictions to boat traffic, restrictions to noisy water activities, and restrictions to littering and sewage discharge, supported by improved facilities for taking care of litter and sewage from boats.

As of elicitation format, the study was based on an “interval open-ended” (IOE) question (Håkansson, 2008). The format is based on the idea that while individuals have one true point of valuation, some people can only state the value they give to a certain environmental change within a range.

Main results

297 persons on the East coast and 251 persons on the West coast responded to the questionnaire. These populations represented the general public living in or very close to the study areas, within approximately 20 km. Mean WTP for improved water quality according to the scenario (SEK per month and household between 2010 and 2029) is for the East coast 102 SEK and for the West coast 71 SEK. For reduced noise and littering according to the scenario the corresponding estimates are 46 and 38 SEK, respectively. The paper also presents descriptive statistics and explanatory factors to WTP.

Further, the paper includes results from a number of follow-up questions in which the respondents were asked to state the degree to which they agreed to a number of statements. The respondents did not seem to find the language difficult and they seemed to understand the information given and considered the scenarios credible. We conclude in the paper that these indications are promising for future studies with this method. However, many of the respondents found the questionnaire complicated and time-consuming, which led us to continue the research on this topic – see paper III.

3.3.3. Paper II - Benefit transfer for environmental improvements in coastal areas: general vs. specific models

Background

The main argument for conducting this study was the requirements connected to the WFD to estimate costs and benefits of catchment management plans. We argue in the introduction to the paper that "since it is unlikely that management agencies in EU countries will have the time or funds to conduct original nonmarket valuation studies in every specific case, BT remains the only viable option. This has emphasized the performance of BT in many respects" (p.240).

Johnston and Rosenberger (2010) conclude in a literature overview that previous BT-research has often focused on methodological advances at the expense of policy-relevant empirical estimates. Further, they conclude that little consensus has been reached concerning different issues that need to be settled before BT can be systematically applied by government agencies. In this paper, we try to further the knowledge on BT in relation to its potential practical use by testing different specifications of functions used for function transfer. More specifically, we compare a “best-fitting” model, i.e. a model that gives the best statistical fit but requires the collection of detailed information at a policy site, with a “general” model that requires only easily obtainable data based on standard demographic databases. The line of argument behind the study is basically that we know that the general model will perform worse, but the question is how much worse in relation to the saved efforts.

Except from the contextual background related to BT, the study is based on the same study areas as paper I.

Method

While BT as such is based on the transfer of results from a primary study in one area (“the study area”) to another (“the policy area”), this particular study needs primary studies in two areas in order to test the match of a transfer. This means that a transfer can be conducted based on different function specifications, and the equivalence can be observed, something that would not be possible without primary study results in both areas. The study was conducted in the same two study areas as paper I, however based on CE instead of CV.

Concerning the scenarios, the CE format allows a finer resolution in the attributes. Hence, for water quality, the WTP for both a one-class and a two-class improvement is studied. In the paper, we describe the design of the variation in attributes which follows standard CE procedures.

Main results

In total, the survey generated 803 responses on the east coast and 502 responses on the west coast, based on two sample groups in each study area. The *locals* represent a sample of the population living very close to the study areas, and the *non-locals* represent a sample of the population living in the southern part of Stockholm county (east coast) but excluding locals, or in the western part of Västra Götaland county (west coast) but excluding locals. The mean WTP for the scenarios (per month and household 2010-2029) is as follows:

- One class improvement of water quality: 266-391 SEK depending on sample group.
- Two classes improvement of water quality: 432-490 SEK depending on sample group.
- Reduced noise and littering: 38-68 SEK depending on sample group.

For the east coast we also included the occurrence of cyanobacterial blooms. The present situation was described as *high risk for one large-scale bloom in the study area every year*. The improvement scenarios were instead every third year (WTP: 288-314 SEK) or every tenth year (WTP: 259-317 SEK). The fact that WTP does not increase with a greater improvement is a sign of insensitivity to scope in the WTP estimates. See section 3.6 for additional general discussions on scope and WTP.

The best-fitting model included a number of demographic variables such as income, gender, age and education level and a number of variables connected to the use of the study area, such as previous experience of poor water quality, types of recreational activities conducted in the study area and the attitude toward the Swedish coastal environment. A full list for each of the samples can be found in table 3 in the paper. The general model, on the other hand, only included easily accessible variables such as gender, income, age, and education level. The test of performance of the respective model specifications were based on transfers and equivalence tests in both directions, i.e. from the east coast to the west coast as well as the other way around. It is concluded that the difference in fit is relatively small. However, the transfer errors are in any case between 20 % and 60 % depending on attribute and direction of transfer.

3.3.4. Paper III - Detailed or fuzzy information in non-market valuation studies – the role of familiarity

Background

Also this paper takes its stance in the requirements of the WFD to conduct studies on costs and benefits of the measures suggested to reach GES. We wanted to assess whether a study based on a less detailed scenario description might generate results that are equivalent to those of a study with a highly detailed scenario description such as the one used in paper I. Acknowledging the result from a literature review (presented in the introduction of paper I), which suggests that information is needed to make the respondent more familiar with the particular good or service, we investigate whether there is a difference in the effect of improving the information level between respondents who are familiar with the good vs. unfamiliar respondents. The hypothesis is that the effect on WTP from increased information is low for familiar respondents, whereas it is higher for unfamiliar respondents.

Method

We applied a split sample survey in the west coast study area (the same study area as for papers I and II). One of the samples received the same questionnaire as in paper I (Detailed information), whereas the other sample received a questionnaire that had been stripped down on details concerning the ecological information given (Fuzzy information), to represent something like an 'economist's best effort' to describe status quo and the environmental change.

For example, Secchi depth (a measure of water transparency) in meters for each potential quality class was defined in the Detailed version, as a descriptor of water clarity, whereas only 'low', 'moderate' or 'good' Secchi depth was used to describe water clarity in the Fuzzy version, with no further specification. The precision in information concerning status quo also differed. In the Detailed version, eight spots in the study area were marked with their present water quality class, whereas in the Fuzzy version, status quo was only described as varying between very low, low or moderate (the three lowest quality classes). Further, there were underwater photos representing each quality class in the Detailed version, whereas in the Fuzzy version there were only two underwater photos exemplifying overgrowth of filamentous algae as a phenomenon in general.

Except from the description of the water quality scenario, the questionnaires were identical. The scenarios are presented in full in the appendix to paper III. The WTP elicitation format was the same as in paper I, i.e. interval open-ended (IOE).

Main results

The survey generated 251 and 250 responses for the *Detailed* and *Fuzzy* samples[9], respectively. We outline three main findings, based on t-tests:

- When only information is accounted for (and not familiarity), the amount of information has no effect on WTP.
- Independent of amount of information, familiar and unfamiliar respondents have different WTP.
- When accounting for both familiarity and information we conclude that:
    - For familiar respondents, information does not affect WTP results;
    - For unfamiliar respondents, information has an impact.

The questionnaire also included a follow-up study similar to the one presented in paper I. More detailed information slightly improved the credibility of the scenario, as well as the respondents' understanding of the measures and their results.

3.3.5. Discussion and notes on paper I, II and III

As academic articles, papers I, II and III were framed around advancing the methodology for SP studies in relation to the need for future estimates. However, importantly, these are all case studies based on only two study areas and the results should hence not be taken as general in terms of their statistical conclusions. For example, the results from paper II do not imply that it never matters whether a general or a best-fitting model is used for BT, and the results from paper III do not imply for future studies that as long as respondents are assumed to be familiar with the good, information on status quo and the environmental change does not need to be detailed. Rather, these studies represent empirical proof that it is not always better to use best-fitting models or detailed scenario descriptions. Future research in this area could replicate these studies in order to build up a greater number of tests.

[9] The samples represent *locals*, i.e. live within approximately 20 km from the study area.

Concerning paper I, we conclude that a holistic, well-detailed, ecologically anchored, and policy relevant scenario description works as basis for a SP study, in terms of respondents being able to digest the information, and in terms of providing WTP estimates. However, ferociously, we were not that worried beforehand that it wouldn’t work and the readers of the article are likely not that surprised that it did. The difference between this type of scenario description and many other in the literature is not enormous. Based on this – what is the real contribution here? I would argue that the main contribution is to provide a demonstration and illuminating a discussion on the problems associated with poorly defined and too delimited environmental change as basis for the scenario description. For example, which is also illuminated by the findings in paper III on familiarity, when formulating an environmental change that does not encompass all, or at least the important, effects of a certain measure (such as stating that a certain measure will improve water clarity when in reality it will also lead to effects on vegetation, cyanobacterial blooms, etc.), the respondents will be misinformed.

A note concerning paper II is its implicit starting point that a function based on explanatory factors behind WTP is to be used as a basis for BT. However, another method is to use a so-called point value transfer, which is an even less demanding transfer option, and so far the literature does not provide a clear answer to whether this method provides better or worse precision in transfers. Hasselström et al. (2014) provide a brief overview of the literature on this topic. Further, ibid. argue that there are several other, perhaps more severe, problems involved in benefit transfer methodology than choosing between point value transfer and function transfer, or between different model specifications for a function transfer. For example, the unit of transfer is not always obvious. By nature, the study area and the policy area to which a result should be transferred are not identical in terms of the amount of water bodies or the areal of water. Hasselström et al. (2014) exemplify by the following possible units:

1) *Aggregated WTP per catchment area*, i.e. the WTP in total for a certain geographic area is transferred to another similar geographic area, irrespective of e.g. population size and number and area of water bodies.

2) *WTP per household per catchment area*, i.e. similar to 1) but the transfer is based on aggregating to the population at the policy site.
3) *WTP per household per water body*, i.e. similar to 2) but also adjusting for differences in the number of nearby water bodies.
4) *WTP per household per water area unit*, i.e. similar to 3) but instead adjusting for differences in the area of the nearby water bodies.

In an example calculation it is shown that the difference between 2), 3) and 4) could be remarkable (Hasselström et al., 2014). Each one of the units are associated with their own set of assumptions for what determines WTP and given that there is only one 'study area', it is not possible to test which assumptions are the most accurate (there is no variation in these variables between respondents in one and the same study area).

The unit used for the transfers in our case is rather similar to 2). In the light of these great uncertainties, the relevance of reducing data collection in relation to WTP models, as suggested in this paper, becomes even more relevant. If it does not affect the transfer results to a large extent, one might argue that the use of a 'best fitting' model rather than a model based on easily available population data may imply a workload that is disproportionate in comparison to its contribution to the reduction of uncertainty in the transfer.

Finally, a note concerning paper III. Our conclusion points to a reasonable finding: high familiarity with the area and its environmental state implies that the marginal effect of providing more detailed information is small or zero. However, it may be hard to know beforehand whether the respondents will be 'familiar' or not with the area and it does not make much sense to first investigate familiarity in order to decide whether to develop a questionnaire with detailed information or to settle with providing less detailed information. Rather, the results points to the conclusion that, unless one is investigating an issue for which familiarity is certainly high, a detailed scenario description is preferable.

3.4. Baltic Sea studies (papers IV, V and VI)

3.4.1. Overarching research questions

Papers IV, V and VI were conducted in the research network BalticSTERN, with participants from all nine countries surrounding the Baltic Sea. As the name suggests, the research was inspired by the Stern review on climate change (Stern, 2006), which presented a cost-benefit analysis of taking measures to reduce the emissions of climate gases. Compared with papers I, II and III, the academic contributions of papers IV, V and VI are more oriented towards presenting estimates and data and less towards methodological issues.

There is one particularly overarching question which was the spark for all three studies: What are the benefits of improving the state of the Baltic Sea, considering eutrophication effects, in accordance with the present international agreements? Previous research on this topic is summarized in Söderqvist and Hasselström (2009), who studied 40 previous publications related to the benefits of an improved environmental state in the Baltic Sea and conclude that the existing studies at that time were mainly based on local regions, which made it hard to draw general quantitative conclusions in relation to the Baltic as whole. This contributed to motivating a new study, as reported in paper V.

While the estimation of the monetary value of improvements was an aim in itself, also the use of, and attitudes towards the Baltic Sea entails economic information, however not monetarily estimated. In order to better understand the role of the Baltic Sea for the well-being, based on preferences of the inhabitants of the Baltic Sea countries, there was a need for new data. This type of data is reported in paper IV.

The term “eutrophication effects” is complex. For example, different nutrients may cause different types of eutrophication effects (Elmgren and Larsson, 2001) and the spatial dimension is also crucial when taking measures since there are important and sometimes complicated interactions between the measures taken close to the coast and the status of the open sea (Brattberg, 1986; Savage and Elmgren, 2004). Public preferences concerning different types of eutrophication effects and the spatial dimensions could provide input to decision-making.

3.4.2. Paper IV: Public preferences regarding use and condition of the Baltic Sea – an international comparison informing marine policy

Background

Söderqvist & Hasselström (2009) concluded that there is a need for further quantification of social preferences for ecosystem services which marine ecosystems provide. Recreation was identified as a crucial area to study. The usefulness of adopting an ecosystem services approach has been emphasized in e.g. MA (2005) and TEEB (2010), and coordinated research across the nation borders is important in the context of MSFD, as the management of the Baltic Sea requires international cooperation (Bertram & Rehdanz, 2013).

Prior to this study, there existed no overarching study on neither the use of – nor the attitudes towards – the Baltic Sea among the inhabitants of the nine countries with coastlines to the Baltic Sea. Previous studies identifying recreational use patterns or ecosystem values in the Baltic Sea have been local, or at best, national (Söderqvist & Hasselström, 2009). For example – how large a percentage of the populations visit the Baltic regularly for recreation? What are the activities undertaken? To what extent do people see the state of the Baltic Sea as a problem?

Method

A survey instrument was developed and translated to national languages, consisting of five sections: (1) an introduction including a definition of the Baltic Sea; (2) questions about respondents' connection to and general use of the sea and their place of residence; (3) details of their most recent visit to the sea; (4) attitudinal questions, and (5) questions concerning socio-demographic characteristics. The interviews were conducted via telephone in all nine countries except for Estonia, Latvia and Lithuania where face-to-face interviews were instead conducted. In all countries, a random sample of approximately 1000 individuals was surveyed. In Russia, the sampled population was delimited to the Kaliningrad and St: Petersburg regions. The questions were based mainly on Likert scales 1-5, from *totally disagree* to *totally agree*.

Main results

In total, over 9 000 responses were collected. Comparisons with national statistics indicated that the sample characteristics were reasonably close to national averages, with a slight overrepresentation of female and older respondents in some countries. A summary of main findings:

- The vast majority of respondents had at some point participated in Baltic marine recreation, with the lowest share in Russia (49 %) and the highest in Sweden (97 %).
- Beach recreation, followed by swimming, were the most popular activities measured by the average number of visits.
- Baltic water quality was in general not seen as restricting recreation (but with some differences between countries – especially Russians and Poles were somewhat more convinced that water quality was restricting recreation, whereas Danes and Swedes were more convinced that water quality was not restricting recreation).
- In general, respondents viewed the status of the Baltic Sea environment to be neither bad, nor good (but respondents living in the eastern regions regarded its condition to be worse than those living in the western region).
- On average, respondents were worried about the state of the Baltic Sea.
- Respondents were, on average, not willing to contribute additionally financially for actions to improve the state of the Baltic Sea. For Swedish respondents, 49.7 % disagree or rather disagree than agree to the statement "I am prepared to contribute more financially for funding actions", while 21.9 % neither disagree nor agree, and 28.4 % agree or rather agree than disagree. See section 3.4.5 for further discussion around this finding.

3.4.3. Paper V: Benefits of meeting nutrient reduction targets for the Baltic Sea – a contingent valuation study in the nine coastal states

Background

The Baltic Sea is shared by nine heterogeneous countries. The countries differ with respect to e.g. income levels, political stability, attitudes (see paper IV) and geography, where Sweden and Finland have particularly long coastlines, whereas others have shorter coastlines (e.g. Lithuania), some have connections to other sea areas (e.g. Denmark and Russia) and some have parts of the country very far away from the Baltic Sea coastline (especially Germany and Russia). There are no previous studies on WTP involving these nine countries together.

The aim of this study was to estimate the benefits of reaching the targets set by the Helcom Baltic Sea Action Plan (Helcom, 2007), in which Denmark, Germany, Poland, Russia, Lithuania, Latvia, Estonia, Finland and Sweden have agreed to take measures to tackle the degradation of the Baltic, not the least concerning the loads of nitrogen and phosphorus to the Baltic.

Method

The scenarios used in the survey were developed using extensive modelling combining information on the current state of the sea and present nutrient loads and assumptions about the future development of economic sectors. Thus, the models provide a quantitative link between benefit estimates and the environmental state, and subsequently to costs of measures. Hence, the whole chain from the cost of measures to their environmental results, to the benefits of these results is tied together (cf. paper I).

The study was based on a CV survey conducted simultaneously in all nine countries in 2011. The data collection was based on Internet surveys in most countries and face-to-face interviews in Latvia, Lithuania and Russia. In Russia, the population was delimited to mainly the western regions. The elicitation format was payment card (see section 3.2).

The scenario was based on mainly two components: a table describing five status classes in words, where each status class was assigned a color (see appendix I to the paper), and maps using these colors describing the current and potential future state (given measures) of the different sub-basins in the Baltic in 2050. The respondents were then asked to state their WTP for a scenario in which the BSAP targets would be fulfilled. The mean WTP was then aggregated to the respective populations. Due to the rather large proportion of respondents stating zero WTP, a spike model (Kriström, 1997) was used to estimate mean WTP. The spike model has characteristics which takes these zeros into account when modelling the mean value.

Main results

In total, 10 564 responses were collected. Mean WTP in each country, adjusted for purchasing power, was calculated to between 5.5 and 75.7 Euros per year per person, with the highest values in Sweden, Finland and Denmark (in falling scale) and the lowest values in Latvia, Russia, Lithuania and Poland (in rising scale).

In most countries, the proportion of respondents being willing to pay was around 50-60 %. However, the proportion was lower in Russia (31 %) and higher in Sweden (73 %). The drivers behind WTP in terms of explanatory variables were investigated for each of the countries. In general, income positively and significantly influences WTP, whereas age has different effects in different countries. Women tend to have stated higher WTP than men in Denmark, Finland, Germany and Poland. Highly educated respondents stated a higher WTP than others in Finland, Germany, Poland, Russia and Sweden. Further, WTP decreases with distance to the Baltic Sea in Denmark, Finland, Latvia, Poland and Sweden. In Russia, the same pattern exists, in that respondents from the coastal regions stated a higher WTP than those living further away from the sea. Further, plans of future recreational visits to the Baltic and the lack of substitute recreational areas tend to increase WTP, along with personal experience of or prior knowledge about eutrophication.

The aggregation was based on multiplying the estimated mean WTP with the respective populations. For all countries except Russia, this implied an aggregation to the national adult populations. In Russia, the aggregation was made to the western regions in which the sample was based. The total aggregated WTP was estimated to 3 600 million Euros per year.

3.4.4. Paper VI: Baltic Sea Nutrient reductions – what should we aim for?

Background

Eutrophication has a range of effects on the Baltic Sea ecosystem. Some effects, such as decreased water clarity, increased cyanobacterial blooms and increased growth of filamentous algae have a direct effect on the recreational quality, whereas other effects affect us humans more indirectly, such as changes in fish species composition, deterioration of underwater habitats such as sea grass meadows, and oxygen deficiency in bottom sediments. Whereas paper V focused on eutrophication effects rather holistically, information on preferences also concerning more detailed aspects of eutrophication might be relevant to support future management decisions. For example, there is a debate on the roles of nitrogen vs. phosphorus for the eutrophication of the Baltic, where it has actually been suggested that an increase in nitrogen loads might help reducing cyanobacterial blooms by stimulating growth of other phytoplankton competing with cyanobacteria for the supply of phosphorus (Elmgren and Larsson, 2001). Schindler (2012) suggests that cyanobacteria should be the primary concern when deciding on Baltic nutrient reduction management strategies, essentially ignoring other eutrophication effects.

Further, knowledge on preferences towards the dimensions of time and space might contribute to decision support. The improvements resulting from measures take decades to realize, and there might exist trade-offs both in time and space concerning specific methods. For example, reduced inputs of phosphorus to coastal areas might imply quick, localized reduction of eutrophication but at the price of a greater export of nitrogen to outer areas (Brattberg, 1986).

Method

The study was based on the dataset used in paper V. Perceptions of eutrophication effects were analyzed based on responses to a series of questions measuring how problematic respondents considered different effects of eutrophication to be, measured on a Likert scale 1-5, from “not at all a problem” to “a very big problem”. Respondents who stated a positive willingness to pay were asked to state which of the effects of eutrophication they had in mind when stating their WTP. The respondents were also asked to state the extent to which they considered the whole sea vs. a specific sea area, and the open sea vs. coastal areas when stating their WTP. This data was used to gain insights on preferences in relation to the spatial aspects. Concerning the time dimension, the respondents were asked to state the main reason for being willing to pay, where one of the options was "future generations will be able to enjoy the water quality improvements". Data from these three sources was the foundation for the statistical analysis in the paper.

Main results

The main results are summarized below:

- All effects of eutrophication were not deemed as equally problematic, but the differences between the effects were in most cases small. Poor water clarity was in general considered as less problematic than the other eutrophication effects in all countries, whereas oxygen deficiency was seen as more problematic than the other effects among German, Latvian, Lithuanian, Russian and Swedish respondents. In Finland, cyanobacterial blooms and oxygen deficiency were seen as most problematic.
- The consideration of different eutrophication effects when stating WTP varied between countries, with no clear, general, pattern. However, cyanobacterial blooms seemed most important in Finland and Poland, water turbidity in Lithuania and Russia, oxygen deficiency in Denmark, and fish species composition in Sweden. In Latvia and Estonia, respondents ranked several effects of eutrophication about equally important. One potential explanation to the differences in responses between the two types of questions for eutrophication effects might be that the second data source is based only on respondents who stated a positive WTP, i.e. the samples are different.
- More than half of the respondents who were willing to pay stated that they considered the whole Baltic Sea rather than a specific area when answering. However, there were some differences between countries. Sweden had the highest proportion of respondents who considered the whole sea when stating WTP, followed by Germany,

Denmark, Finland and Lithuania. One possible explanation for this is Sweden's long coastline. Respondents in Latvia, Russia, Poland and Lithuania were more often willing to pay for an improvement in a specific area of the Baltic Sea than respondents in other countries.

- On average, respondents thought slightly more about the coastal areas than the open sea areas when stating their WTP, but the general pattern is for responses to be close to the middle of the Likert-scale.

3.4.5. Discussion and notes on paper IV, V and VI

The three papers together provide a common picture. The inhabitants of the nine Baltic Sea states use the sea to a large extent, care about the sea and many are willing to pay for improvements concerning eutrophication effects. The willingness to pay for improvements is directed at the environment as whole, rather than towards very specific effects of eutrophication, or towards specific areas of the sea. Further, the responses to the questionnaire used as basis for papers V and VI indicate that people are patient concerning the long time lag between measures and results.

The differences between responses in different countries entail important information. For example, the relatively high use of the Baltic Sea in Sweden, along with a particularly high WTP suggests that Swedes tend to have high stakes in the Baltic Sea environment, whereas other countries, relatively speaking, have lower stakes. This type of information could be useful in preparation for international negotiations.

There is a discrepancy between paper IV and V concerning the share of respondents who expressed a positive WTP for measures. In paper V, shares range from 31 % (Russia) to 73 % (Sweden), with the majority of countries ranging from 50-60 %. Paper IV only presents the average score for responses on a Likert scale (1-5 where 1 is totally disagree and 5 is totally agree) to the statement "I am prepared to contribute more financially for funding actions" [for improving the state of the Baltic Sea environment]. Lithuania (1.64) and Latvia (1.75) had the lowest scores, followed by Russia (2.16), Germany Germany (2.36), Sweden (2.43), Estonia (2.46), Finland (2.74), Denmark (3.78) and Poland (3.23). Hence, the scores are below 3 in all countries except Poland and Denmark which can be interpreted as a non-WTP on average in most countries. In paper

IV however, no information was given to respondents neither on the current environmental state, nor on the need for financing of measures or on the possible measures that can be taken and the potential results of such measures. In this sense, the results from paper V are more reliable concerning this specific question.

Interestingly, the results of paper IV suggest that the citizens of the Baltic Sea countries consider the state of the Baltic Sea to be neither bad, nor good. This could be put in relation to the presentation of the actual present status, presented in paper V (along with much other literature, see Helcom, 2010), which suggests that the Baltic is in fact in a bad shape. In Hasselström (2008), the same type of pattern is visible. In that study, deep interviews were conducted with representatives from the tourism industry in all nine countries, and the main conclusion was that the selected respondents stated that their industry is more or less unaffected by any current marine environmental problems, while future degradation is seen as a risk to the industry. One possible explanation is that we are used to the poor status of the sea. Another is that the status is poor in ecological terms, compared to a pristine state, but that it yet doesn't affect our daily lives to a large extent. In any case, people seem to worry about the future development and many are willing to pay for measures leading to an improvement.

Concerning the estimates in paper V, there are a number of sources of uncertainty. Except from those outlined in Section 3.2., such as hypothetical bias (i.e. "are respondents really willing to pay the stated amounts"), also the aggregation is equipped with uncertainty due to representativeness issues. Further, the treatment of non-responses is complicated by the different survey methods. The details to consider and the possible tests and statistical models available are almost infinite.

The analysis made in paper VI was based on the dataset provided by the study conducted in relation to paper V. The most theoretically sound way of investigating preferences towards different aspects of eutrophication, time and spatial scales would be to conduct a choice experiment. However, there was no opportunity to make a new study of such scale. Hence, the paper is based on the existing data.

A final remark concerning the benefits estimates: It was later shown that the benefits exceed the costs (SwAM, 2013; Ahlvik et al., 2014). Importantly, this does not necessarily mean that the financial flows from the measures will be possible. Rather, the results suggest that we will gain in welfare (or "well-being" or "utility"). The WTP measure is a proxy for the gross benefits perceived by individuals, measured in monetary terms in the absence of any other standardized unit for welfare, and it does not represent a financial flow. A less complicated way of communicating this type of result is however: We would be willing to pay more than it costs.

3.5. Oil spills from shipping in the Arctic (paper VII and VIII)

3.5.1. Overarching research questions

Papers VII and VIII were conducted as parts of the project "Arctic Games", within the "Arctic Futures in a Global Context" research program. The development of the Arctic region is of global interest, and the area provides several important ecosystem services: supporting services such as maintenance of habitats; regulating services such as climate regulation; provisioning services such as food for consumption; and cultural services, such as recreation and cultural heritage (Magnussen et al. 2010). Several of the habitats and plant and animal species in the area are threatened (CAFF 2010), and the preservation of well-functioning ecological systems in the area is an important policy task. Further, the Arctic is the region where the environmental impacts of climate change are most strongly expressed (Gradinger et al. 2004).

The Arctic is one of the most important oil-producing regions in the world (OECD/IEA 2008), yet it is particularly vulnerable to oil spill damages due to sensitive ecosystems, increasing human pressure, and natural circumstances such as cold water and harsh weather. These circumstances suggest a difficult clean-up process and a slower recovery from oil damages. Further, oil spill response and preparedness in the region is expensive relative to more populated regions. Gautier et al. (2009) show that 13 percent of the world's undiscovered oil may be found in the Arctic and new extraction opportunities are increasing due to rapidly melting Arctic ice. Increased drilling activity implies an increased spill risk at the point of extraction and from increased oil tanker traffic. Further, the forecasted increase in overall economic activity in the region leads to increased non-tanker shipping, which also poses an increased oil spill risk.

Papers VII and VIII study the benefits and costs of measures to a) reduce the probability a future oil spill in the Lofoten-Barents Sea region, and b) measures to reduce the consequences of a spill, once it has occurred, through improved response preparedness. Together, these two papers provide information of relevance for policy-making concerning how to handle oil spill risks in the region: What ecosystem services are threatened? What are the costs to tourism and fishery, and what is the willingness to pay for preventive measures vs. measures aimed at reducing the consequences of a spill, once it has occurred? What are the costs of such measures? Information on the potential costs and benefits of policy measures could feed into discussions on economic trade-offs.

3.5.2. Paper VII: Valuation of oil spill risk reduction in the Arctic

Background

The recent (Swedish) chairmanship of the Arctic Council highlighted the importance of oil spill prevention in the Arctic region (Swedish MFA, 2011). Future increase in oil and gas drilling in the region as well as in the general economic activity, along with new northern transport routes can be expected to lead to increased shipping, both for tanker traffic and for other traffic (OECD/EIA, 2008).

Globally, a number of studies have attempted to estimate the cost of a large scale oil spill accident, either ex ante or ex post (the former asks respondents about their willingness to pay (WTP) to prevent a spill before it happens, while the latter estimates the actual damage after an oil spill has occurred). Ahtiainen (2007) uses CV to estimate the WTP for improving the oil spill response capacity in the Gulf of Finland. Liu et al. (2009) use CE to estimate the WTP for improving oil spill preparedness in the Wadden Sea (along the German, Dutch and Danish coasts). Carson et al. (2003) report on a CV study performed after the Exxon Valdez oil spill (Alaska 1989) which aimed to estimate non-use values lost from a typical oil spill. Loureiro et al. (2006) estimate the cost from the Prestige oil spill along the Spanish coast in 2002. As a basis for valuation, they use market prices of lost catches in commercial fisheries and other seafood industries such as fish farming and the fish processing sector. Further, they estimate the cost of environmental losses by using reposition costs of birds and mammals. They also include clean-up costs in their estimations. Van Biervliet et al. (2006) use CV to estimate the WTP for preventing hypothetical oil spill scenarios along the Belgian Coast. The Arctic is not a well-studied area in the context of oil spills.

The study in paper VII investigates three different research questions: First, it is investigated to what extent the respondents' own ("subjective") judgement of the probability of a potential oil spill steers WTP for reducing probability. This question relates to the role of correct information to the public concerning the oil spill probability. Second, the difference between WTP for reduced a) probability and b) probability and consequences from an oil spill accident causing negative impacts on ecosystem services is assessed. This question provides guidance concerning whether policy should be oriented towards probability reduction or preparedness in case an accident occurs, or towards both probability and preparedness. Third, the respondents' preferences for different ecosystem services in the Arctic are analysed. This question provides insights concerning potential trade-offs in what ecosystem services are relatively more and relatively less important to protect in the public's eyes.

Method

A CV study was carried out in the autumn 2012 in the three northernmost counties in Norway (Nordland, Finnmark and Troms) focusing on Arctic ecosystem services at risk from a potential future oil spill from shipping in the Lofoten-Vesterålen area in northern Norway. The survey was executed as a web panel study, with respondents representing a random sample of the adult population (18 years old and above) in the three counties. Two scenarios were used; one in which the probability for an oil spill was hypothetically reduced through a set of measures (Probability scenario), and one in which both the probability and the impact, should an accident occur, were reduced (Total risk scenario). Two versions of the questionnaire were sent out, with variation in the ordering of the scenarios. The purpose of this was to test whether or not the order of the scenarios had any impact on WTP.

As a basis for the scenarios, a case was used with a ship accident outside Lofoten-Vesterålen causing a potential leakage of 60 000 tons of crude oil, drifting towards Lofoten and Vesterålen, as outlined by von Quillfeldt (2010). The current estimated probability of such an accident is once every 350 years, however increasing to once every 150 years if no new measures are taken. Given measures such as new shipping routes, the probability would remain unchanged despite increased shipping.

The scenarios are summarized in tables 2 and 3, respectively. In the Probability scenario, the probability of an oil spill is targeted, and additional measures imply that the probability is kept at once every 350 years instead of increasing to once every 150 years. The consequences to wildlife, fisheries and coastline, should an accident occur, remain unchanged. In the Total risk scenario, both the probability and the consequences are affected. In this scenario the consequences to wildlife are reduced in terms of extent and duration. Further, the fishery restrictions can be shortened to 1-2 years instead of 2-3 years, and the coastline affected by the spill will be shorter and stay affected for two months instead of four.

Table 2. Impacts on probability of an oil spill and consequences on ecosystem services due to the implementation of measures as presented in the Probability scenario.

Probability scenario				
	Without additional measures		**With additional measures**	
Probability of an oil spill	Once every 150 years		Once every 350 years	
Wildlife	*Affected share of population*	*Time period affected*	*Affected share of population*	*Time period affected*
Birds	50%	Some months to some years	50%	Some months to some years
Mammals	30%	Some months to some years	30%	Some months to some years
Fish	10%	Some months to some years	10%	Some months to some years
Fisheries	Restrictions 2-3 years		Restrictions 2-3 years	
Coastline	**1000 km, 4 months**		**1000 km, 4 months**	

Table 3. Impacts on probability of an oil spill and consequences on ecosystem services due to the implementation of measures as presented in the Total risk scenario.

Total risk scenario				
	Without additional measures		**With additional measures**	
Probability of an oil spill	Once every 150 years		Once every 350 years	
Wildlife	*Affected share of population*	*Time period affected*	*Affected share of population*	*Time period affected*
Birds	50%	Some months to some years	25%	Some months
Mammals	30%	Some months to some years	15%	Some months
Fish	10%	Some months to some years	5%	Some months
Fisheries	Restrictions 2-3 years		Restrictions 1-2 years	
Coastline	**1000 km, 4 months**		**500 km, 2 months**	

The respondents were asked to state their WTP for the respective scenarios through a classical and interval open ended (CIOE) question, such as (for the probability scenario):

How much is your household willing to pay for avoiding a change from today's situation with a probability of an oil spill once every 350 years to once every 150 years?

My household is willing to pay between …… NOK and …… NOK

or

…… NOK

each year in order to avoid such a change.

Main results

In total, the survey generated 400 responses, 200 in each version of the survey. The mean WTP per household per year was estimated to 851 NOK for the Probability scenario and 888 NOK for the Total risk scenario. This (small) difference in WTP between the scenarios is not statistically significant. Further, the share of respondents stating that they are not willing to make any monetary contribution to fulfil the scenarios is similar for the two scenarios. Our interpretation of these data is that respondents are primarily concerned with avoiding an oil spill to take place at all.

Another finding concerns the respondents' perception of the current probability for an oil spill such as the one outlined in the scenarios. The majority of respondents perceive this probability to be higher than the expert judgement made in the report by von Quillfeldt (2010). Further, the perceived ('subjective') probability does not affect WTP. However, since the 'objective' or 'expert judgment' probability is presented to the respondents later on in the questionnaire, before the WTP question, this can be assumed to explain this otherwise somewhat surprising result.[10]

A third finding is that the shares of 'protest bids' is high. A bid was in this study assumed to be a 'protest bid' when stated WTP is zero due to reasons such as 'it is wrong that I should pay', 'costs should be covered by existing taxes', ' the polluter should pay', 'I don't think the measures would work', or 'I have received too little information'. Stating a zero WTP due to these reasons may imply that the respondent actually has preferences pro realizing the scenarios, and a positive WTP, but that reasons such as those above mean that these preferences are not visible in the data. There are thus reasons to believe that the true WTP of these respondents may be non-zero despite that the respondents stated a zero

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[10] Respondents were first faced with the "perceived probability question", then later with information on the "objective/expert judgement probability", and then they answered the WTP question (the respondents were not able to go back in the questionnaire and change their answer to the "perceived probability question"). Given this setup, there are no apparent reasons for hypothesizing that "perceived probability" is correlated with WTP, assuming that respondents have read and understood the information given on "objective" / "expert judgment" probability.

WTP. A sensitivity analysis is made in the paper in which protest bids are a) included as zero WTP, or b) excluded from the dataset.

Concerning the respondents' preferences towards different ecosystem services, the data indicate that 'clean and non-polluted sea areas' was the most important of the ecosystem services listed, followed by 'biodiversity', 'food' and 'aesthetical values', while 'recreation', 'cultural values' and 'open sea areas and transport routes' were seen as somewhat less important, however with rather small differences between the services.

3.5.3. Paper VIII: Costs and benefits associated with Arctic marine oil spill prevention

Background

There are many areas in the Arctic where balance is needed between the use of natural resources and the protection of ecosystem services. There are potential conflicts of interest between different stakeholder groups, sometimes from different states. The assessment of economic costs and benefits of different scenarios can provide insights into these conflicts. However, only a limited number of estimates of costs or benefits connected to resource related conflicts in the Arctic are to be found. To our knowledge, there is no example of a full CBA i.e. one that includes both market values and non-market values (e.g. values of ecosystem services). For oil spills, previous studies have focused on whether or not oil extraction should be carried out (e.g. Ibenholt et al, 2010) rather than on potential oil spills from shipping due to increased transports in the region. Oil spills can be a consequence of more oil and gas extraction, but also of increased consumption in general. Only a limited number of papers are devoted to the economic valuation of the changing provision of ecosystem services due to an oil spill accident in the Arctic (e.g. Lindhjem et al., 2013; Paper VII in this thesis, and Carson et al., 2003). Magnussen et al. (2010) provide a thorough overview of marine ecosystem services in northern Norway and point out e.g. maintenance of habitat and diversity, climate regulation, provision of food, recreation, and cultural heritage as particularly important services.

This paper uses CBA to analyse a resource conflict in the Arctic. A case study based on a hypothetical future oil spill event from shipping in the

Lofoten area in northern Norway is used, in a setup based on the scenarios presented in paper VII. This case study, involving conflicts related to oil and how the sea and the ecosystem services that it provides should be managed, can be viewed as an example of Arctic resource conflicts where many different actors are involved.

Method

The CBA in this paper compares a) a business as usual (BAU) scenario with b) a policy option implying measures that decrease the probability of a marine oil spill accident from shipping. The business as usual (BAU) option implies that no policy action is taken and that the probability of an oil spill increases from once every 350 years (0,003 in a given year) to once every 150 years (0,007 in a given year), similarly to the scenarios used in Paper VII. The policy option implies measures which keep the probability constant at once every 350 years. Table 4 outlines the types of costs and benefits involved. On the cost side, there are the costs of measures such as tug boat preparedness. The resulting benefits are avoided losses of income to industries such as fisheries and tourism, as well as avoided losses of ecosystem service value. For loss of income, the *expected value of damage* is used, which is based on the product of a probability of an unwanted event and the damage cost (D), should that event occur. Given the probabilities above, the expected avoided loss of income can be calculated as (0,007-0,003) * D.

Table 4. The costs and benefits of measures to reduce probability of oil spill.

Costs	Benefits[1]	
Costs for measures such as tug boat preparedness	Expected avoided loss of income	Avoided loss of ecosystem service value

[1] Another benefit, not included in this study, is avoided clean-up costs.

Literature studies were used for the estimation of costs of measures as well as the expected avoided loss of income. For avoided loss of ecosystem service value, the estimates presented in Paper VII are used. The costs and benefits are calculated as net present value (NPV) based on a 15 year time period and a discount rate from 0-6 %.

Main results

The costs of measures were estimated to NOK 857 – 907 million (calculated as present value (PV)). This estimate is based on Kystverket (2012) and includes investment costs for three emergency tug boats being placed in the area. On the one hand, this is as such an underestimation since operating costs are not included in the analysis and since the effect of these three tugboats are estimated to reduce the probability with 30 % rather than 60 %, which is the scenario used in the CBA. On the other hand, however, the initial probability is higher in Kystverket (2012), which complicates the analysis. Further, there might be other measures available which are more cost efficient.

The benefits were estimated to NOK 1,5 – 3,3 billion (PV). This estimate is entirely based on the avoided loss of ecosystem service values, as presented in Paper VII. An aggregation of the WTP per household was made to the adult population in the three counties Nordland, Finnmark and Troms, assuming an average household size of 2,2 persons (SSN, 2013).

The reason for basing the estimate entirely on the avoided loss of ecosystem service values, and not adding the avoided loss of income in fisheries and tourism is that there may be double counting risks since shares of these values may be reflected in the WTP estimates. However, these financial effects are useful for analyzing distributional effects. For fisheries and tourism, the benefits are estimated to NOK 138 – 201 million (PV) and NOK 14 – 39 million (PV), respectively.

Two major conclusions can be drawn from the analysis: Willingness to pay for measures outweigh the costs, and the benefits in terms of avoided loss of (non-market) ecosystem values seem to be greater than the avoided loss of financial values in tourism and fisheries sectors. This highlights the importance of not delimiting CBAs to only financial effects.

3.5.4. Discussion and notes on paper VII and VIII

The distributional aspects of the CBA illustrate two important issues: First, while benefits of measures seem to exceed the costs, benefits and costs are distributed over different actor groups. In the example with using additional emergency tug boats to prevent oil spills, the costs may be borne by public organizations financed by taxes. This implies a connection between beneficiaries (being the tax-paying public) and financing, however in a rather complex structure. For other measures, such as rerouting traffic or requiring certain equipment onboard, the costs would be borne by the transporters. In that case there is a gap in incentives, where the shipping industry may be negative to additional measures and the public would be positive to them. In the longer run perhaps the costs would anyway be reflected in market prices, but this connection is long term, indirect and very complex. Hence, despite benefits exceeding costs, it might be difficult to execute even well-motivated policies.

Second, the WTP measure is a measure of well-being translated into monetary terms, and does not necessarily represent any financial flow. It was clear from paper VIII that the value of avoided expected financial damage to fisheries and tourism due to measures is in fact lower than the costs of measures. From a purely financial perspective, measures would, according to the analytical setup, cost more money than they generate. However, the capital costs of the oil forgone, damages to the ship, and clean-up costs are not included, which leads to an underestimation of the financial benefits of avoiding accidents.

Concerning the WTP survey presented in paper VII, a few additional remarks can be made. First, a rather large share of the respondents (30 %) gave protest bids. The characteristics of these respondents were analyzed and the results showed that respondents who are interested in environmental issues reveal protest answers to a greater extent compared to those who are not. This strengthens the suspicion that their statement of zero WTP is in fact a “protest” rather than that they do not value increased protection from oil spills.

The aggregation of WTP estimates to a total estimate, as presented in paper VIII, is likely to lead to an underestimation since it can be assumed that other people than those in the three northernmost counties in Norway have a WTP for the measures. For example, it is not unlikely that also people in Oslo would value these measures, given the status of the Lofoten area as a natural heritage.

Finally, the large difference between the estimated financial effects and the estimated loss of non-market values are important to recognize from a liability perspective. Currently, as discussed in Fejes et al. (2011), the operator may be responsible for recovering financial costs resulting from an oil spill from shipping, but not for non-market costs due to damage to ecosystem services. If the operator would be responsible for restoration and compensation of losses in non-market values, this might provide increased incentives for avoiding accidents and/or minimizing the consequences from accidents, by internalizing an externality that is today not being sufficiently recognized. However, the results from stated preferences studies are debated in legal contexts, see e.g. Fejes et al (2011).

3.6. Comparison of estimates between studies

Table 5 presents a summary of the mean WTP estimates for the primary studies conducted in this thesis (as presented in paper I, II, III, V and VII). For paper V, the table presents the WTP by Swedish respondents. The units for WTP have here been expressed as SEK per month per household (as in paper I-III), implying that recalculations have been made for paper V (from EUR per person per year) and paper VII (from NOK per household per year). This type of recalculation can be problematic since there may not be full correspondence between estimates based on per household WTP and estimates based on per person WTP multiplied by the average number of persons in a household and. Further, there may not be full correspondence between per month WTP and per year WTP divided by twelve. However, this recalculation is needed for the purpose of comparisons.

Table 5. WTP estimates from Paper I, II, III, V and VII (SEK per month per household).

		Eutrophication: Water quality, 2 classes improvement	**Noise and littering:** Restrictions leading to less noise and littering
Papers I-III	Paper I, east coast (CV)	51-153	24-67
	Paper I, west coast (CV)	32-71	19-58
	Paper II, east coast (CE)	440-490	37-38
	Paper II, west coast (CE)	431-488	66-68
	Paper III, west coast, detailed info (CV)	34-89	--
Paper V (CV)		**Eutrophication:** Reaching BSAP ambitions	
		125*	
Paper VII (CV)		**Oil spills:** Probability scenario	**Oil spills:** Total risk scenario
		46-102**	52-106***

*original estimate 75.7 Eur per person per year
** original estimate 481.4-1065.0 NOK per household per year
*** original estimate 540.1-1105.4 NOK per household per year

Figures used for recalculation:
Number of persons per household in Sweden = 2.2 (SCB, 2016a; data year 2012)
Exchange rate SEK/EUR = 9.0335 (Riksbanken, 2016; accumulated year-average 2011)
Exchange rate SEK/NOK = 1.15 (Valutakurser.net, 2016; 1 Oct 2012)

Note: Figures are not adjusted for inflation. The inflation measured in development of consumer price index (KPI) between the earliest and the latest survey (fall 2009-fall 2012) was approximately 5 % (SCB, 2016b).

In addition to what has previously been discussed with regards to the respective studies, a number of findings are particularly interesting to discuss in the light of this comparison:

1) Mean WTP estimates are between 19 and 490 SEK per household per month.
2) A comparison of CE vs. CV results for the same scenarios in paper I-III shows large differences.
3) Mean WTP for reaching BSAP ambitions (paper V) (i.e. improvements in the whole Baltic Sea including the Kattegat) is

within the upper range of the interval for the mean WTP derived by CV for reaching good ecological status locally in the east coast and the west coast study areas, respectively, but lower than the mean WTP derived by CE (paper I-III).

Below, each of these findings are discussed:

1) *Mean WTP estimates are between 19 and 490 SEK per household per month.*

From a budget restriction perspective, the estimates are reasonable, constituting rather small shares of the after-tax household income[11]. This implies some certainty concerning the question whether there is actually an ability to pay the stated amounts (on average). Also the very fact that there is a variance in estimates (also between CV estimates) somewhat signals reasonable results given that different people have responded to different scenario constructions. While not being tests of validity, findings where mean WTP is very high compared to after tax income or where there is no variation between WTP despite that entirely different scenarios are studied would be suspicious.

2) *A comparison of CE vs. CV results for the same scenarios in paper I-III shows large differences.*

While some of the differences in WTP between the east coast and west coast could be driven by differences in preferences and the difference between paper I and paper III may be due to different survey timings (one and a half years in between), the difference between the CE and CV studies represented by Paper I and II are very large. The estimates are based on data from the same questionnaires, only using different elicitation formats. The CE estimates are several hundred percent higher than the CV estimates.

This result is hard to explain, and unfortunately for the trust in the precision of SP methods in general, it is not entirely uncommon (although there are not many studies available where such a test is possible - usually, there is a choice made between CE and CV à priori). Boxall et al. (1996) studied environmental quality changes arising from forest

[11] Average SEK 17 000 – SEK 39 000 per household and month after tax depending on age group (SCB, 2016c)

management practices on recreational moose, and their CV estimate was twenty times as large as their CE estimate for the same environmental change. A similar result, although with smaller differences between CE and CV estimates, and in the opposite direction, can be found in Barrett et al. (1996), where CE estimates were four to five times higher than the CV estimates.

There are a number of important differences between CE and CV. First, the CV study in these surveys used an open-ended interval (IOE; Håkansson, 2008) format for eliciting WTP while CE is based on eliciting WTP through (hypothetical) choices made by the respondent in relation to a number of fixed options, each involving a certain, pre-determined cost. While CE is far from equivalent to closed-ended CV, the difference between open-ended and closed-ended formats may impact WTP. The literature is diverse concerning whether WTP in open-ended formats tends to be higher, lower or not significantly different from WTP in closed-ended formats (e.g. Kealy & Turner, 1993; Frew et al., 2003).

The bid vector, i.e. the vector of fixed cost amounts included in the choice sets, is an important factor in CE design. Carlsson & Martinsson (2007) show in a split-sample survey study that the level of the bid vector had a significant impact on the marginal willingness to pay. The marginal willingness to pay (to reduce power outages) was higher in the survey version with higher costs.

In general, it has been argued that CE is more suitable under certain circumstances and CV in other circumstances (see e.g. review by Hynes et al., 2011). A common argument is that the choice between CV and CE depends on the purpose of the valuation, where CV is more suitable for valuing a policy package, while CE is more suitable for valuing the individual components of this package due to its ability to elicit trade-offs made by the respondents with regards to these components. However, with regards to paper I-III, defining the “policy package” is not straightforward. One might argue that respondents have valued two distinct policy packages; “less eutrophication” and “less noise and littering”; or one, including the attribute “less eutrophication” and the attribute “less noise and littering”.

Hence, it is not possible to conclude on which of the elicitation formats generates the best estimates. In relation to paper II on BT, this implies that the choice of primary data for transferring is important. A rather large transfer error of, say, 100 % would still be small compared to the observed differences between CE and CV estimates in papers I-III. In any case, BT implies a larger degree of uncertainty in estimates compared to the implementation of a new primary study at the policy site, but the question still remains concerning whether CV or CE at the policy site should be conducted.

3) *Mean WTP for reaching BSAP ambitions (paper V) (i.e. improvements in the whole Baltic Sea including the Kattegat) is within the upper range of the interval for the mean WTP derived by CV for reaching good ecological status locally in the east coast and the west coast study areas, respectively, but lower than the mean WTP derived by CE (paper I-III).*

The mean WTP among swedes for realizing the scenario studied in paper V, i.e. measures to fulfil the BSAP ambitions for reduced nutrient loads, with corresponding state changes in the ecosystem as a consequence, was estimated to (presented as a point estimate) 125 SEK per month per household (recalculated from 76 Euros per year per person). The WTP for less eutrophication locally in the east coast and west coast study areas in paper I and III was estimated to 34-153 SEK per month per household. This observation raises the question: shouldn't WTP be a lot higher for such a large-scale change compared to a local improvement?

Concerning scope, economic theory would suggest a higher WTP for larger (or additional) environmental improvements, although with a diminishing rather than linear WTP-to-scope dependence due to diminishing marginal utility (e.g. WTP for 100 days of air without smog is more, but less than the double of WTP for 50 days of air without smog). However, a quite large number of studies have previously concluded on scope insensitivity using split-sample studies. For example, Tolley & Randall (1986) found that WTP for 10 days or 180 days of improved atmospheric visibility was about the same. Similar examples for various applications can be found in e.g. Kahneman & Knetsch (1992), Desvouges et al. (1993) and Sckade & Payne (1994). Additionally, "adding up" tests have been performed, where e.g. three different samples are asked either WTP for improvement A (group 1) or WTP for improvement B (group 2), or WTP for improvement A and B together (group 3). In theory, mean WTP for group 3 should be higher than mean WTP for group A and B, respectively, and the sum of mean WTP for group A and B should not exceed WTP for group 3. Violations of this test have been demonstrated by e.g. Diamond et al. (1993) and Frederick & Fishhoff (1997).

Except from the geographical scale, the scenarios used in papers I-III and paper V are rather similar, both representing a shift from the current eutrophied state to “good ecological/environmental status”. Hence, could a simple version of an “adding up test” be to add the east and west coast mean (CV) WTP estimates from paper I and III and compare this sum with the mean WTP from paper V, based on the assumption that the scenario in paper V would fulfil also the local scenarios in paper I-III? The answer is no. First of all, such a test would rely on the assumptions that e.g. 1) the west coast sample would have a WTP for water quality improvements on the east coast as well as in other Swedish coastal areas, and 2) this WTP is equally high as that for water quality improvements in the ‘own’ area. This is unlikely the case.

Further, the scenario in paper V was specified as an improvement in the entire Baltic Sea, divided in 8 sub basins, also covering the open sea. The effects on the coastlines were not spelled out to the respondents since this is likely to vary between locations. Respondents in Sweden were on average, as shown in paper VI, considering open sea areas and coastal areas rather equally when stating their WTP. This means that there is a principal difference in the environmental change that was valued in papers I-III compared to paper V (and VI). In paper I-III, local coastal improvements were considered and in paper V, basin-wide improvements and also open sea areas were considered. Hence, the adding up of a number of improvements similar to the scenarios in papers I-III all along the Swedish coast line would not be equivalent to the scenario in paper V.

4. A broader discussion on valuation

The type of data presented in the eight papers can be useful as arguments in environmental policy debates, pointing to a public demand for measures, but also to the trade-offs between costs and benefits. In this section, four additional aspects are discussed, using the results from the papers and the review of methods from section 2 as a background. First, the concept of valuation as being an eye-opener is discussed. Second, the text touches on some moral aspects in relation to economic theory.

Economics and moral philosophy is a large field and the text presented here represents only a few observations. Third, the concept of uncertainty is discussed, where a conclusion is that results such as those provided in the papers in this thesis are actually far more uncertain than what is usually described in environmental economics papers. Fourth, the important difference between 'value' and 'price' is discussed.

4.1. Valuation as an eye opener

A likely motivation behind the formation of environmental policies during the last decades is a build-up of knowledge concerning a) environmental impacts from various pressures and b) the consequences of these impacts for us humans. Such knowledge could for example have driven the Montreal protocol on emissions of ozone depleting substances in the end of the 1980s, the phase out of lead in gasoline between the 1970s and 1990s and of asbestos in construction material in the 1980s. One of the focuses of today's environmental policy is greenhouse gases, and a contributing factor behind the agreements during the COP21 conference in Paris in the autumn 2015 is a greater common understanding concerning the environmental, social and economic risks associated with inaction.

Hence, an instrument to drive the debate on environmental policy is to develop further information showing the consequences of various environmental pressures. One possible source of information is e.g. ecological and medical research. Another source of information is to use environmental economics, including the valuation of public preferences. Environmental economics is in its theoretical essence a tool to study trade-offs. Low valuations may be an argument for less rigid environmental policies and high valuations may be an argument for more rigid environmental policies. However, not only do the estimates matter, but also the inherent need for developing causal chains from pressures to environmental impacts to its resulting implications for society. Valuation is not possible without these chains and a systemic approach.

There are many possible bases for decision-making, including e.g. deontology which focuses on rules or norms (Hausman and McPherson, 1996) and consequentialism. Environmental economics is rooted in the latter, where information on consequences from an action, divided into pros and cons, is seen as a basis for decisions. Information concerning consequences from various policy options may not (and perhaps should not) be solely decisive. However, there are reasons to believe that studies based on environmental economics shine a light on dependencies between the ecosystem and humans that would otherwise not be as visible.

Paper VII estimated the WTP for measures to prevent oil spills in the Lofoten-Vesterålen area. This result can be understood as an eye-opener to the fact that the public in the area values such measures, and the underlying ecosystem services approach also helps illuminating why the public expresses these preferences. Certainly, there is prior knowledge of potential consequences from an oil spill in ecological terms which may be sufficient to motivate measures, but the ability of environmental economics methods to bridge the ecological effects to human well-being provides an additional dimension to the information available to decision-makers. Further, since the aggregated WTP exceeds the cost of measures (as shown in paper VIII), this type of information could be interpreted as an expression of public support for more stringent measures.

A similar discussion is relevant in relation to the estimates provided in paper V. Prior to this study, there was already consensus in the scientific literature (e.g. as summarized in Helcom, 2010) that the Baltic Sea is heavily eutrophied and that this affects the ecosystem to a large extent. The study however expands the understanding concerning the link between the ecosystem impacts and human well-being. Further, the aggregated WTP for improving the state of the Baltic Sea with respect to eutrophication (estimated to 3 600 million Euros per year according to paper V) is greater than the estimated cost of measures (Swam, 2013; Ahlvik et al., 2014). This type of information can serve as an eye-opener, indicating that more stringent policies are not only necessary to improve the state of the ecosystem, but would also make society better off from a perspective of economics.

4.2. Economics and moral judgment

The field of economics has a deductive core assuming several premises. Two of them are particularly important to discuss in the context of this thesis:

- Rational agents: utility maximizing consumers and profit maximizing producers.
- An economic value is measured through marginal changes in consumer and producer surpluses.

The concept of rationality is sometimes criticized based on distaste for the perceived ascription of egoism to individuals. However, this is not an argument against the premise (further, utility maximization does not necessarily imply egoism, as is discussed in Hausman & McPherson, 1996). To at least some extent, there is empirical evidence for the premise of rationality as defined in economics. When prices go up on a good, demanded quantity often falls. When the water quality at the nearby beach becomes bad, some people go to another beach instead.

The assumption of rationality is useful, e.g. for creating models. This assumption and the resulting economic deduction offer possible explanations to some of the causes behind environmental degradation and are also useful for presenting some solutions. For example, public goods and externality theory are likely to have contributed to regulations and other environmental policies, as well as fruitful discussions on how to better protect the environment.

Nevertheless, rationality remains - being at the very core of microeconomic theory - a much debated concept and the scope of the term tends to vary in the literature (cf. Hammond, 1997). Much of the debate related to rationality in the field of non-market valuation concerns the formation of preferences. In the light of paper III in this thesis, an interesting aspect of preference formation is discussed in Spash (2002), who concludes that information given to respondents in a CV study is not only informing respondents but also forming their preferences. Similar discussions can be found in e.g. Parks & Gowdy (2013).

If rationality is understood as acting to maximize welfare given a set of preferences, a possible interpretation of this type of preference formation is that rationality is context dependent. What is rational behavior at one moment may be irrational at another, given new information. In terms of WTP for e.g. water quality improvements such as in paper III, it is not surprising that WTP is affected by more detailed ecological information for respondents who were previously unfamiliar with the good – the information may have contributed to forming respondents' preferences (although this result may also have been caused by only informing respondents with an already established set of preferences). These discussions raise another question: What else than information in a survey forms people's preferences and how does that affect decision-making?

This question relates to value pluralism, as outlined in section 2, and to the second premise: "An economic value is measured through marginal changes in consumer and producer surpluses." One of the most common and relevant arguments against this summing of individual values to a 'social welfare' measure is the underlying utilitarianism – (e.g. disregard of distribution) and consequentialism (e.g. disregard of means to achieve benefits). These issues have been well-discussed in many formats, see e.g. Sagoff (2008) and Hansson (2007). Clearly, the economic definition of value is narrow and technocratic in relation to the everyday usage of the term. 'Value' is a word that means different things to different people.

The rationality premise is an assumption of behavioral traits among individuals and firms and one might argue that the intention of this assumption is to explain, understand and predict behavior rather than to say something about how society should be. The value definition however has very clear normative implications, leading to conclusions for example about relevant environmental ambition levels. It functions well in the models to match the rationality assumption as an operative instrument for explaining and predicting behaviors in society, but there is no sufficient argument to conclude that this value definition should be decisive in a broader decision-making process.

Apart from this discussion on the very core of microeconomics, there are also several cases where valuation and CBA need to rely on additional implicit normative judgments (cf. Wegner & Pascual, 2011). This concerns for example the geographical limits within which values should be aggregated. In theory, anybody that attaches a value to something should count in the analysis but this is in practice impossible. In papers I, II, III, VII and VIII, the geographical limits are likely too narrow to include everybody who might have a WTP for improvements in the area. For example, it is fully possible that somebody from Denmark has a WTP for water quality improvements in local areas in Sweden and that somebody from Sweden has a WTP for reduced oil spill risks in northern Norway. Further, the discussion around the discounting of future benefits is an area with a high involvement of normative judgments. The discussions around the Stern report (Stern, 2006) is a relevant example.

There are likely many other such examples. The conclusion from the discussion above is that the deductive method is informative given its context, but the deductive part of economic science is only a part of the science. Another part of the science is based on a (more) normative premise. The results of an analysis need to be understood in relation to the premises. This implies that the results from economic studies can only be a part of solving any dilemma concerning environmental management. As argued in Scharin et al. (2016), economic valuation and CBA should be complemented with other studies. In their article, this is illustrated with a study of distributional impacts and the construction of future scenarios including the risk for regime shifts. There are however also many other assessment tools available and different topics are likely to require different combinations of assessment methods. This being pointed out, weighing different information pieces together is a challenge. How should a decision-maker act when different assessment tools generate different answers, e.g. when a CBA shows that measures are not economically profitable while other assessments point to a risk of unrepairable negative impacts on the ecosystem if measures are not taken? While there are tools available for facilitating the process of weighing various information pieces together (such as MCA; e.g. Söderqvist et al., 2015), decisions are made by decision-makers and not by assessment tools.

4.3. Uncertainty

The concept of valuation uncertainty is usually discussed in relation to SP studies and the respondents' uncertainty of WTP. In paper III, there is a discussion on the topic, suggesting several possible reasons for uncertainty. For example:

- Uncertainty of future ability to pay.

- Uncertainty about own preferences.
- Uncertainty of whether the project will be carried out in accordance with the given scenario.
- Uncertainty of the scenario as such, due to too limited information.

The last of the bullet points suggests that information can be used to reduce some of the uncertainty. However, the concept of 'ecological illiteracy' (see for example Söderqvist et al., 2004) suggests that respondents only state their preferences concerning what they believe to be the consequences in a certain scenario. This belief might correspond with how the scenario is formulated, but it also depends on previous knowledge and own interpretations of the information given; two factors which are difficult to control for. This is a problem that might lead to partial and/or uncertain valuation results. Consider for example a study on bathing water quality and nutrients - if the effluents of nutrients decrease, this does not only affect the bathing water quality. Also other ecosystem consequences might (or might not) occur, such as on fish stocks, algae blooms, and species composition. This might in turn lead to effects on the ecosystem's resilience, i.e. buffer capacity against future environmental disturbances. And so on. These other 'hidden' effects may provide benefits for the public in the end, which should be reflected in a willingness to pay to achieve these effects. However, the respondents might only state their willingness to pay for the 'direct' effect of improved bathing water quality. Hence, the discussion brought up in paper I on valuation scenarios also has a bearing on uncertainty.

The ecosystems are complex. It will never be possible to model an entirely correct prediction of the environmental consequences from many of the measures studied in SP studies. There are knowns, unknowns and unknown unknowns. By using as much knowledge as possible from marine ecology and providing this information to survey respondents, it is in theory possible to cover the knowns. That is – one can provide a thorough picture to the respondents about the expected consequences of environment-improving actions, not only a highly fractionalized picture. Unknowns are more problematic. One might assume some correlation between certain ecological variables, but this correlation may not have been evaluated. For example, when reducing nutrients, we might believe that this causes changes in the food web, which might affect several species positively or negatively. However, we might not know which species would be affected and to what extent. In this case – how would the respondents be able to know? Finally, there are certainly effects that we don't know that we don't know about. These are effects which are by definition unexpected.

Hence, the terms 'ecological illiteracy' and 'valuation uncertainty' should not only be used to describe a potential problem caused by uninformed respondents to a survey. We are all uninformed and systems ecology research combined with methodological development in scenario construction is key to more precise estimates in the future.

4.4. A value is not a price tag

Sometimes the terms 'value' and 'price' are used interchangeably. However, the terms are not synonyms. An economic value is a measure of well-being associated with the consumption of a good or service ('consumption' in this sense could also be related to the knowledge that the ecosystem is in good shape, see section 2 on non-use values). Values are based on the trade-offs people are willing to make. A price on the other hand could perhaps be best described as the cost (for the buyer) or revenue (for the seller) associated with a market transaction. In competitive markets, according to economic theory, price will be set to equal marginal value and marginal cost for the last unit sold. In that sense, under certain circumstances value and price may end up being equal. But that doesn't mean the concepts are the same.

In order for something to have a price, there need to exist possibilities for transactions, i.e. there need to exist some sort of market for this good or service. The mere fact that most ecosystem services are referred to as non-market services indicate that they are not usually priced. Valuation studies in themselves do not imply transactions.

There are, however, a large number of examples in which an ecosystem service, or something that serves to generate an ecosystem service, is given a price. Cole et al. (2012) provide an overview on the usage of markets for ecosystem services and conclude: "As of 2010 there were 39 active global markets targeting for biodiversity, with transactions totaling more than 2.9 billion USD, 57 active water quality markets, and voluntary carbon markets trading 131 million tons of CO2" (exec. Summary p. xxii). An interesting question relating to these ecosystem service markets that so far has not been much studied is the relation between value and price. A hypothesis that is not so far-fetched is that these are far from equal. CO2 markets may serve as an example. The social damage cost (i.e. value of avoiding emissions) associated with one ton of CO2 emission is difficult to assess (see Noring, 2014; Isacs et al., 2016). The price of emission allowances on the other hand, is a function of supply and demand. Supply is in theory controlled by the emission cap set by policy, and demand is a function of production techniques, global demand for goods, transports, etc. It seems unlikely that price and value in this case would end up being equal. Many other examples of markets are likely to have the same type of properties. It may however be desirable to strive to price ecosystem services according to their marginal value. This is what is implied by the theories of internalizing externalities originally based on Pigou (1920).

The inequality between value and price is important in relation to the discussion on commodification of ecosystem services (e.g. Baggethun & Pérez, 2011). A valuation of an ecosystem service is not the same as saying that anyone can buy the right to exploit nature for a price based on this value. Instead, valuation is information about trade-offs individuals are willing to make. As such, it is simply the measurement of preferences. However, monetary valuation may in the longer run be a driver of commodification, as is argued in Baggethun & Pérez (2011).

5. Conclusions

The studies presented in papers I-VIII represent examples of monetary valuation of environmental change in the marine environment. All papers except paper IV are based on data from stated preferences studies. In this section, conclusions are presented from two perspectives – the first being the broader discussions on valuation as such in the cover thesis, the second being paper I-VIII along with the more paper-oriented discussions in the cover thesis.

5.1. Broader conclusions based on the cover thesis

Results from SP studies that estimate the monetary value of environmental change as decision support can contribute to more informed decisions. However, there are a number of complexities involved with 1) methodological questions such as study design, aggregation of results, provision of information to respondents, etc. and 2) the usage of economics as a basis for decisions. This has bearings on the interpretation of results from a valuation study. There are certain methodological standards to fulfill in order for the study to be reliable, but different choices within the frames of these standards, such as choosing between CE and CV, choosing which population to survey, which payment vehicle is the most relevant, and so on, are factors that may affect the results to a great extent. This is a problem for the methods, as it is often not possible to validate the results in terms of controlling whether respondents would actually be willing to pay the amount stated in response to the hypothetical scenarios.

Still, decisions have to be made concerning relevant trade-offs. Economic information constitutes one dimension of the decision-making context, and despite the various problematic aspects outlined in this thesis, SP studies contribute to filling a gap that is difficult to fill with other types of studies, especially concerning non-use values. However, sustainable management of the environment requires a multitude of viewpoints among which environmental economics is only one.

5.2. Conclusions based on paper I-VIII and related discussion

WTP estimates, CBA results and evidence of public preferences

Three different setups were used to study the benefits of environmental improvements in marine areas in the Baltic Sea and in Lofoten, using stated preferences. The first setup (papers I-III) was based on local improvements in water quality with respect to eutrophication, in line with the requirements of the WFD, and introductions of SCZs leading to less noise and littering. For the two study areas, Himmerfjärden and the area around Orust and Tjörn, mean WTP for improved local water quality was estimated to 34-153 SEK per month and household using CV and 431-490 SEK per month and household using CE. For less noise and littering the corresponding estimates are lower: 19-67 SEK using CV and 37-68 SEK using CE. There is no stand-alone explanation available in the literature or in the studies to explain a) the differences between CV and CE estimates concerning water quality, or b) the relative similarities in the same comparison for noise and littering. However, previous literature has shown that it is not uncommon with variations, sometimes large, in WTP estimates based on the elicitation format. This finding illustrates some of the uncertainties involved in stated preferences studies, and also highlights the importance of choosing elicitation format based on suitability for the valuation purpose rather than ad hoc preferences for one method over the other. However, deciding which type of elicitation format is 'best' under certain circumstances is not always straightforward.

The second setup (papers IV-VI) was based on improvements in the state of the entire Baltic Sea, including Kattegat, using a stated preferences survey simultaneously in all nine bordering countries. The scenario would imply fulfilment of the BSAP targets for nutrient abatement, leading to a state which is reasonably close to the fulfilment of the MSFD in 2050. The aggregated WTP was estimated to approximately 3 600 million Euros annually. Countries are heterogeneous with respect to the WTP estimates with Swedes having the highest mean WTP (76 Euros per person per year) and Latvians the lowest (5.5 Euros per person and year). Papers IV-VI also show that inhabitants of the nine Baltic Sea states use the sea to a large extent and care about the sea. The willingness to pay for improvements is directed at the environment as whole, rather than towards very specific effects of eutrophication, or towards specific areas of the sea. Further, the responses to the questionnaire used as basis for paper V and VI show that people are patient concerning the long time lag between measures and results.

The third setup (papers VII-VIII) was based on risk reductions for oil spills from shipping outside Lofoten in northern Norway. Two scenarios were studied, one in which measures would be taken to reduce the probability of a large scale oil spill from shipping in the area, and one in which measures would be taken to a) reduce probability as well as b) reduce the consequences of such a spill through increased preparedness. Mean WTP for the two scenarios were estimated to 851 and 888 NOK per household per year, respectively and the difference in WTP between the two scenarios is not statistically significant (the order of presentation of the scenarios to the respondents was varied between two samples). These findings suggest a stronger preference for measures to reduce probability than for measures to reduce the consequences, once an accident has occurred. The cost-benefit analysis in paper VIII shows that aggregate benefits for measures to reduce probability (1.5-3.3 billion NOK) are exceeding aggregate costs (857-907 million NOK). Further, it shows that the financial values at risk are small compared to the non-market values. Expected avoided loss of income for fisheries and tourism due to future oil spills over the next 15 years is estimated to 138-201 million NOK and 14-39 million NOK, respectively. This finding illustrates the importance of taking non-market values into account in public decision-making.

Matching scenario design with policy needs

Concerning the design of studies, paper I shows that respondents are able to understand and attach a monetary value to valuation scenarios that are 1) anchored in a policy relevant description of environmental change, 2) holistic in the sense that they are encompassing a number of effects from one and the same policy rather than focusing on one of the resulting effects, 3) well-specified in terms of status quo and the state of the environment resulting from measures, and 4) giving the respondents a rather high level of details concerning ecology. This finding is important given the need for conducting assessments of costs and benefits in relation to the WFD and MSFD, and also more generally in relation to the need for evaluating various management options. The finding suggests an end of an era for stated preferences studies in which the environmental change is unspecified or based on a single environmental indicator while the actual consequences of the suggested measures are more multifaceted.

Trade-off between precision and level of ambition in SP and BT

Paper II shows that a BT is not always significantly improved by using the most well-fitting model specification driving WTP, compared to using a model specification based on easily accessible data such as demographics. Naturally, the most well-fitting model gives a more precise transfer, but the case study presented in paper II shows that the difference in fit between the two model specifications is actually quite small. In paper II, the most well-fitting model includes variables such as usage of and familiarity with the water area (i.e. these variables contribute to driving WTP). Collecting such information at the policy site would likely require a new survey study on its own, and in that case a better choice may be to conduct a new primary valuation study on the policy site instead. On the contrary, the usage of a general model may only require the collection of demographics at the policy site, which are accessible through public databases.

While it is not possible to generalize from these results since these are only findings from one single study, the findings provide evidence that more advanced is not always much better concerning BT. Due to the role of BT as a way of avoiding new costly and time consuming primary studies, this finding is of particular policy relevance.

The findings in paper III suggest that there is a need to provide detailed ecological information unless there are good reasons to assume that respondents are highly familiar with the environment subject to valuation (which could be the case e.g. for certain often-debated local environmental issues). For respondents who were familiar with water quality problems in the study area, the sample receiving detailed ecological information did not express a significantly different WTP compared to the sample that received fuzzy information. For unfamiliar respondents however, the WTP was significantly different between the two samples. Hence, the idea of an easy way out by providing less detail in the questionnaires may, in the case of unfamiliar respondents, lead to inadequate estimates.

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