Valuing ecosystem services
– linking ecology and policy

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Valuing ecosystem services – linking ecology and policy

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“Poverty and environmental problems are both children of the same mother, and that mother is ignorance”

Ali Hassan Mwinyi, former president of Tanzania, 1998
Abstract

Ecosystem services constitute a precondition for human welfare and survival. This concept has also become increasingly popular among both scientists and policymakers. Several initiatives have been taken to identify and value ecosystem services. Several services are threatened, and it has been concluded that in order to better manage ecosystem services they need to be further investigated and valued. By measuring them using a common metric—monetary value—they can be more easily compared and included in decision-making tools. This thesis contributes to this goal by presenting values for several ecosystem services and also including them in decision-making tools.

Starting with a discussion of the concept of ecosystem services, this thesis aims to present values for certain ecosystem services and to illustrate the use of these values in systems-analysis tools such as cost-benefit analyses (CBA) and a weighting set. Links between ecology, economics and policy are discussed within a broader framework of ecosystem services. Five papers are included, in which two contingent valuation studies (CV) have been used to find values for different ecosystem services. One valuation study is focused on the effects from tributyltin (TBT) in Swedish marine waters. In addition, a quantitative assessment framework has been developed in order to simplify analysis of environmental status, progress in environmental surveillance and the relevance of different measures. It is suggested that the framework should also be used when assessing the impacts of other substances affecting the environment. The second valuation study investigates the risk of an oil spill in northern Norway. The results have been included in two CBAs and a weighting set. The first CBA compares costs for remediation of polluted sediments, caused by TBT, with the benefits of reducing TBT levels. The second CBA compares costs and benefits for reducing the probability of an oil spill. The weighting set includes monetary values on a number of impact categories where marine toxicity is based on the valuation study on TBT.

One study also examines the inclusion of environmental costs in life cycle costing (LCC) in different sectors in Sweden.

Results show that respondents consider ecosystem values to be important. The values of Swedish marine waters and coastal areas outside Lofoten-Vesterålen in Norway have been identified and quantified in terms of biodiversity, habitat, recreation and scenery. In the Norwegian case, an ongoing debate on the issue of oil and gas exploration has had an impact on the number of protest bids found in the study.

Based on the cost and benefits of limiting impacts on ecosystem services derived from the valuation studies, CBAs show that the suggested measures are most likely beneficial for society, and the results contribute to policy recommendations. A weighting set has been updated with new values through value transfer. The weighting set is compatible with LCA. The final study shows that companies and public organisations use environmental costs (internal and external) in a limited manner.

In this thesis the ecosystem service concept is used both as an introduction and a guiding thread for the reader, as a way to frame the studies undertaken. The concept of ecosystem services can be useful, as it emphasises the importance of the services to humans. By finding and presenting values of ecosystem services, such services are more easily incorporated into decision-making.

Keywords
Contingent valuation; Cost-benefit analysis; Ecosystem services, Ecotoxicity; External costs; Life cycle costing; Monetary weighting; Non-market valuation; Oil spill
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1 Introduction

1.1 Background

Humans are fully dependent upon functional ecosystems that provide a number of services, such as food, shelter, energy, climate regulation and aesthetic appeal. People place such demands on these services that several are being degraded, e.g. fish stocks and agricultural land (MA 2005). According to The Economics of Ecosystems and Biodiversity (TEEB 2010) many ecosystems are so degraded that they are approaching their tipping points: the thresholds where their capacity to provide services is threatened. Given that the exact location of such tipping points is highly uncertain, precautionary actions must be taken to keep healthy ecosystems intact and avoid species extinction and habitat destruction. In order to prevent further degradation, it is important to identify these services and the role that they have for human wellbeing and economic activities. Ecosystem services and their values have become more integrated into policy during recent years, and politicians initiate investigations that seek to assess and include the value of ecosystem services in planning. The purpose of organisations such as TEEB is to “show how economic concepts and tools can help equip society with the means to incorporate the values of nature into decision making at all levels” (TEEB 2010, p. 3). Recognising these values may help ensure sufficient conservation (TEEB 2010). In many cases, the importance of ecosystem services has been included in policy-making. For example, in the European Union member states must map and assess ecosystems and ecosystem services as part of policy-making (EU Parliament 2012). One of the goals of the Swedish Environmental Objectives is to identify ecosystem services (see, e.g., Naturvårdsverket 2011) and the United Kingdom have made an assessment identifying ecosystem services as a basis for developing policy responses to ongoing environmental degradation (Watson et al. 2011). In Sweden, one investigation (SOU 2013) underscores the importance of including ecosystem services in private-sector planning and development.

There are a number of decision-making tools that can incorporate ecosystem service values, such as cost-benefit analysis (CBA), lifecycle assessment (LCA) or lifecycle costing (LCC) (Finnveden & Moberg 2005; Ahlroth et al. 2011). In CBA, for example, all project consequences should be included and assigned a positive or negative monetary value, then compared. Some consequences, such as those impacting ecosystem services or natural resources, do not have a market price; however, they still can be assigned a value. Values can be either positive (benefit) or negative (cost). In order for values to be compared, they must be weighted in some way. Communication and legitimacy of a study can also be simplified with a common metric. Ahlroth et al. (2011) present a compilation of different weighting methods. One approach is to use money as a common metric. A generic weighting set can be used when time and research budgets are limited. This save resources, since less time is spent finding values (Ahlroth et al. 2011). Generic weighting can be used together with LCA and LCC. LCA models the potential environmental impacts of a product on the environment from cradle to grave (see, e.g., Baumann & Tillman (2004) for a more in-depth presentation). LCC considers costs associated with a product during various stages in the lifecycle (see, e.g., Hunkeler et al. (2008) for further discussion).

By valuing ecosystem services their importance can be understood to a greater extent. It will also simplify ecosystem management (Pascual et al. 2010), since costs and benefits related to ecosystem services are included, highlighting their effects on human wellbeing and hence improving decision-making. All this sheds new light on the process of policy development (Anon 2007). Multidisciplinary knowledge is crucial for sound public policy formulation and implementation (Watson 2012). Increased knowledge of values can lead to improvements in
ecosystem services (Watson et al. 2011). Few ecosystem services are traded on the market and hence have no market price. Those that do have a market price have direct use values, in contrast to non-use values that are not market-based. Examples of services with direct use values include things such as crops, fish and other consumable goods. Services outside the market can have spiritual or cultural value, such as landscapes or species. With many ecosystem services, values are being estimated: e.g. forest conservation, greenhouse gases, fish stocks, coral reefs, organic food production and beekeeping. Other, more diffuse services, such as pure water or climate regulation, have just started being valued (TEEB 2010).

One of the most common methods for valuing ecosystem services is Contingent Valuation (CV) (Freeman III 1993). CV is a stated preference method, meaning that respondents are asked to state their willingness to pay (WTP) for certain changes in welfare. There are several examples where CV has been used to assign values to ecosystem services. For example, searching the Web of Knowledge search database for “contingent valuation of ecosystem services” yields 227 hits, while a search of ScienceDirect yields 1,990 hits for the same keywords. Google Scholar aggregates a number of other search engines, including the above-mentioned ones, and yields as many as 29,300 hits. Examples of study areas are biodiversity (e.g. Ojea & Loureiro 2011; García-Llorente et al. 2011), water quality (e.g. Jones et al. 2008; Östberg et al. 2012), and oil spills (e.g. Ahtiainen 2007; Carson et al. 2003).

Despite the many studies on valuation of ecosystem services, there are still areas where we lack sufficient knowledge. The value of non-market impacts from harmful substances on ecosystem services, and thus welfare is one such area (Martin-Ortega et al. 2011). Valuation studies for the Arctic region are also inadequate. Some studies have valued the effects of oil spills (e.g. (Loureiro et al. 2009; Bonnieux 2008; Carson et al.2003; Ahtiainen 2007) but only Carson et al. (2003) are concentrated on this specific region, and none of the studies found focus on the risk of oil spills to ecosystem services.

1.2 Aim of the thesis

This thesis is based on five papers. The aim is to present values for certain ecosystem services, illustrating the use of these values in systems analysis tools such as CBA and weighting sets, and then to discuss the links between ecology, economics and policy in a broader framework of ecosystem services.

Below are the main objectives of the papers included in the thesis:

Paper I:

- conduct a valuation study and present estimates for values of ecotoxicological impacts that can be used in CBA and other decision-making tools; and
- evaluate the goal to reach good environmental status (GES) by 2020 in accordance with the Marine Strategies Framework Directive (MSFD) from a welfare perspective, by applying CBA methodology. In order to conduct the CBA, a quantitative environmental assessment framework including several levels of environmental status is developed; including ecotoxicological data, practical feasibility, and valuation methodology.

Paper II:

- investigate to what extent the subjective judgement of the probability of a potential oil spill steers WTP for reducing probability;
address the difference between WTP for reduced a) probability and b) probability and consequences from an oil spill accident causing negative impacts on ecosystem services; and

- analyse the respondents’ preferences for different ecosystem services in the Arctic.

**Paper III:**

- analyse a case study based on a hypothetical future oil spill event in the Lofoten area in northern Norway using CBA by comparing a) a business as usual (BAU) scenario with b) a policy option implying decreased probability of a marine oil spill accident due to some measures.

**Paper IV:**

- present an updated weighting set.

**Paper V:**

- describe life cycle costing (LCC) practices in some Swedish organisations, investigate probable changes and determine whether and how environmental costs (internal and/or external) are considered in current LCC practice.

### 1.3 Outline of the thesis

The thesis consists of this introductory section, followed by a theoretical background, methods, results, discussion and conclusion. In short, the thesis consists of two non-market valuation studies (paper I and II), two CBAs (paper I and III) based on the respective valuation studies, an updated weighting set (paper IV) partially based on one of the valuation studies, and results of how environmental costs are incorporated into the processes of some Swedish organisations (paper V). A framework for assessing environmental status is also developed (paper I).

Chapter 2 presents theoretical background by developing the concept of ecosystem services that forms the basis for the framework used to discuss the findings in the papers. Chapter 3 describes and discusses the methods used in the papers. This chapter also further develops the concepts of environmental valuation and CBA. Chapter 4 presents a summary of the results. Chapter 5 discusses the paper findings and situates them within the context of the theoretical framework, while Chapter 6 explores conclusions.
2 Ecosystem services

2.1 Introduction to ecosystem services

Ecosystem services (ES) is a concept used to describe the benefits that humans derive from ecosystems. The concept has increased in popularity since the 1990s, even though the term can be found in earlier literature. Costanza et al. (1997) and Daily (1997), among others, are considered the initiators of current discussion. Ecosystems can be defined in different ways (see Haines-Young & Potschin, 2009, for an overview). According to the Millennium Ecosystem Assessment (MA) (MA 2005), initiated by the United Nations (UN), an ecosystem is “a dynamic complex of plant, animal, and microorganism communities and the non-living environment interacting as a functional unit. Humans are an integral part of the ecosystems. Ecosystems vary enormously in size; a temporary pond in a tree hollow and an ocean basin can both be ecosystems.” (p27).

Ecosystem services are further defined as “the benefits people obtain from ecosystems. These include provisioning services such as food and water; regulating services such as regulation of floods, drought, land degradation and disease; supporting services such as soil formation and nutrient cycling; and cultural services such as recreational, spiritual, religious and other nonmaterial benefits” (p27).

In the MA classification, provisioning, regulating and cultural services are based on supporting services: e.g. nutrient cycling, soil formation and primary production (Figure 1). From provisioning services humans gain materials that are needed for a good life: construction materials, nutritious food and other necessary goods, but also the possibility of protecting oneself from disasters. Regulating services are also the basis for a good life, providing a safe environment and adequate livelihood. These achievements, in turn, provide health to humans through feelings of wellbeing, e.g. by control of diseases and climate, physical strength, and regulation of the supply of food, clean air and water. Cultural services contribute to several aspects of wellbeing, but also to social relations: the ability to help one other and establish mutual respect through social and cultural conventions. The important thing here is that there is a lot that humans do and benefit from that can be derived from the services provided by ecosystems.

Figure 1. Ecosystem services based on MA (2005).
Additionally, the relationship between ecosystem services and humans is mutual. Human actions also have an impact on ecosystems and their services. Humans threaten ecosystems and deplete resources; a growing population that maintains today’s consumption levels will increase consumption of resources and impacts on ecosystems and their services. Well-functioning ecosystems reduce the risk of natural disaster and human conflict, while poorly managed ones can increase these same risk (MA 2005).

2.2 Valuation

2.2.1 Valuing ecosystem services

Non-market goods such as ecosystem services are usually valued using two different approaches: stated preference or revealed preference methods. Revealed preference methods are based on the actual behaviour of people, estimating the value of a good based on the amount that people spend to use that good. A typical example of a revealed preference method is the travel cost method (TCM), where expenses associated with e.g. a trip to a natural preserve is used to estimate the value of characteristics of that area. One disadvantage of the revealed preference methods, or indirect methods, is that they only capture use values. This means that non-use values such as existence values are not included in the estimate. This implies an underestimation of the value. According to theory, the stated preference methods, or direct methods, can capture these non-use values. In stated preference methods respondents are asked to state their willingness to pay (WTP) for a specific scenario where a change in welfare is described. Common methods are CV and choice experiment (CE). Stated preference methods are associated with a number of problems, e.g. different aspects of uncertainty (further described in chapter 3) or situations where respondents are asked to value something they currently don’t use or are unfamiliar with (see, e.g., Hanemann and Kriström 1995; Li and Mattsson 1995). There are ways of reducing or eliminating these uncertainties (see, e.g., Ellingson et al. 2009), some which will be presented in chapter 3.

CV can be a suitable approach for valuing ecosystem services in general (Barkmann et al. 2008), and biodiversity in particular (Nunes & van den Bergh 2001) when thorough pre-studies of the relevant ecosystem services have been made and questionnaires are carefully designed. There are a number of issues to keep in mind, though. In valuation studies, it is crucial that the benefits obtained from a good are defined in terms of something to which the respondent can actually relate (Barkmann et al. 2008). It is also important to provide respondents with a clear change in the good that is easy to explain (Nunes & van den Bergh 2001). Environmental goods and ecosystem services should be valued in close collaboration with other disciplines. To arrive at relevant policy implications, it is important to have both cooperation between ecologists and economists (Bockstael et al. 2000) and collaboration with other stakeholders, such as different organisations and policymakers (Kinell et al. 2012). Cole et al. (2014) also describes the importance of active stakeholder involvement when making decisions on natural resource management.

In a questionnaire, the change illustrated should be marginal¹, even though this can be problematic due to the fact that ecosystems are complex and possible thresholds are unknown. Beyond the complexity of ecosystems, there is also a risk of double counting, since ecosystem services are intertwined by complementary and competitive attributes, e.g. some ecosystem

¹ A marginal project indicates that it does not affect equilibrium prices on the market even though there are exceptions to this. It is up to the investigator to make this judgment but most project are regarded as small, or marginal (Johansson and Löfgren, in Kriström & Bonta Bergman 2013).
services preclude others, and this should be distinguished before making any aggregation (Turner et al. 2003).

2.2.2 Critiques on the economic perspective

The basis for the neoclassical economic framework used in valuation studies is market essentialism, where it is assumed that goods can be substituted and that technological development enhances this substitution. Further, a utilitarian and totally anthropocentric perspective is used, and rational behaviour among consumers is assumed (Chee 2004). The utilitarian and rational assumptions have been questioned in many studies. It is suggested that people change their preferences in different situations and sometimes prefer alternatives that do not obviously maximise their utility; they state attitudes rather than preferences (see, e.g., Chee 2004; Kahneman et al. 1999 for a further discussion). Hence, the valuation of ecosystem services is not straightforward. Even among economists there is no consensus on how to deal with this matter. However, economic valuation of ecosystem services can be helpful for decision-makers (TEEB 2010; Gómez-Baggethun et al. 2010; Chee 2004). A report by the UK Department for Environment, Food and Rural Affairs states that decision-making will be improved by the valuation of ecosystem services, as the costs and benefits of the natural environment are taken into account and the implications for human wellbeing are highlighted. It concludes by stating that this approach provides new insights into policy development: for example, how to target conservation policies or how to create markets for ecosystem services (Anon 2007).

The main risk with valuation, as Spangenberg and Settele (2010) suggest, except for getting the figures wrong, is that economic instruments become an end instead of a mean. An example of the latter is when the creation of certificates for constructing new habitats instead of preserving threatened pristine ones becomes the goal in contrast to the initial purpose of protecting the environment. Gómes-Baggethun and Ruiz-Péres (2011) suggest that economic valuation should be used together with other methods to capture non-monetary values so as to avoid the risk of believing that valuation results reflect a comprehensive picture that can resolve all problems. They claim that monetary valuation itself acts as a driver for commodification. Chee (2004) concludes that by using different forms of stakeholder participation, mathematical models that provide a better understanding of the ecological processes, and tools that include uncertainty will improve ecosystem services management.

2.3 Ecosystem services and policy

Turner & Daily (2008) describe the context in which ecosystem services exist, benefit humans and are affected by humans. Their framework starts by identifying the issue, i.e. what the ecosystem provides and the contexts in which it is situated. This could be the social, economic or politico-cultural setting. Ecosystem services are modelled, mapped and valued. Finally, the impact of policy measures and management choices on ecosystem services are then analysed. The UK NEA (Watson et al. 2011) is based on a framework, similar to that of Turner and Daily (2008), which connects ecosystem services to goods that humans derive from the ecosystems and the wellbeing associated with those goods. This wellbeing gives feedback to drivers of change, such as demographic or technological changes, as well as management practices. All these drivers of change affect the ecosystem services that provide goods, human wellbeing and so forth. A better valuation might provide better conditions to improve decision-making and investments, create opportunities for wealth creation and jobs, and raise the possibilities for future human wellbeing (Watson et al. 2011). A similar model is also presented by de Groot et al. (2010) for use in spatial planning.
Below is a simplified illustration (Figure 2) of the framework based on the previous references. This illustration follows the structure of the thesis to guide the reader through the framework. In Step 1 the ecosystem services are identified. Step 2 identifies risks on and impacts to them. Based on the risks and impacts, in Step 3 the associated changes in the ecosystem services are valued. The values are then used in Step 4 to arrive at policy implications using tools such as CBA to compare benefits with costs. Implemented policies then function as drivers, Step 5, for existing ecosystem services in Step 1, putting them under certain risk in Step 2, and so on.

One example of incorporating ecosystems and their services and values into policy is the UK National Ecosystem Assessment (UK NEA) (Watson 2012), which was undertaken with the support of more than five hundred experts. This assessment, largely based on the MA (2005), presents a number of principles for how to include ecosystem services and their values within official decision-making. The status of ecosystem services within the country has been identified. The work has assessed the status and trends for different habitats and past and future uses for them (Watson 2012).

A recent Swedish investigation (SOU 2013) finds that it is crucial to include ecosystem services in planning and decision-making as biodiversity continues to be eroded. They further conclude that monetising the values provided by ecosystem services has the potential to improve decision-making insofar as value is highlighted and it is simpler to include ecosystem services in inclusion in decision-making processes. In practice, however, supporting services are difficult to value, as they are complex and threshold effects are uncertain. Instead, the authors suggest that provisioning, regulating and cultural ecosystem services be valued (SOU 2013).

Sections 2.4 and 2.5 present the ecosystem services that form the basis of paper 2 and 3, respectively. In terms of the framework, the identification of ecosystem services and the risks to and impacts on them are conducted in Step 1 and 2 in Figure 2.
2.4 Biodiversity in Swedish coastal waters affected by TBT (Paper I)

Biodiversity in Swedish coastal waters are affected by several substances, leading to toxicological effects on species. Beginning in the 1960s, tributyltin (TBT) was used in anti-fouling boat paint, until 2001, when TBT was banned in Europe (EC 2003). In 2008 ships painted with TBT coating were prohibited from entering EU ports. Despite these policy actions, TBT can still be found in sediments (Eklund et al. 2008) and causes a number of negative effects on biota (Alzieu et al. 1986; Beaumont and Budd 1984; Bryan et al. 1987; Fischer et al. 1990).

The Marine Strategy Framework Directive (MSFD) is a policy document aiming at achieving or maintaining, good environmental status (GES) in marine waters by 2020 (EC 2008). GES is achieved when the level of contaminants do not cause negative consequences. Swedish coastal waters have not achieved this status. TBT has been found to such an extent that it forms a significant risk to the marine environment (Helcom 2010), causing a number of anomalies in biota. Oyster shells are deformed (Alzieu et al. 1986) and their immune functions are negatively affected (Fischer et al. 1990), mussel larvae suffer from mortality (Beaumont and Budd 1984) and the reproductive capability of species is affected by imposex and intersex (Bryan et al. 1987). All these effects lead to a risk that populations will not be able to reproduce and will thus face extinction.

2.5 Ecosystem services at risk from an oil spill in Lofoten (Paper II)

In the Arctic, a number of ecosystem services are identified as being under threat from factors such as climate change, infrastructure development (transport routes etc.) and oil and gas spills (CAFF 2010; OECD/IEA 2008). Lofoten-Vesterålen was chosen as a representative case to illustrate a conflict between different actors with varying interests in natural resources and ecosystem services. The risk of oil spills associated with maritime transport is expected to increase in the area (von Quillfeldt (red) 2010). The area provides a number of ecosystem services, such as primary production, food, non-edible goods, genetic resources, habitat and biodiversity, chemical resources, space and waterways, recreation, scenery, science and education, cultural heritage and the legacy of the sea. Some of these services are under more severe risk: for example, habitat, diversity, recreation and scenery (Magnussen et al. 2010). In the case study presented in Paper II habitat and diversity are represented by different species of fish, birds and mammals, which are some of the species groups at greatest risk from oil spills (Brude et al. 2011). The status of beaches and coastline is used as an example of quality of recreation and scenery. Also, potential restrictions in fishing illustrate the possibility to use the ecosystem service for recreation.

A well-functioning Arctic is of concern for several actors, but it is also important for humans in general, given that this area contains a large number of migrating fish and mammals and also plays a role in climate regulation. It is also important for transport routes and for the provision of natural resources (AGP, 2010). The Arctic Council argues for a well-governed region (e.g. Ministry of Foreign Affairs 2011) and has decided to take preventive measures against marine pollution (EPPR, 2013). A number of agreements to reduce environmental risk in the Arctic have been made (see, e.g., Meszaros, 2003).
3 Methods

3.1 Contingent valuation

The CV method was used in the two valuation studies (paper I and II). It involves asking individuals about their WTP or willingness to accept (WTA) different changes in a good or (ecosystem) service. The method has become increasingly popular since the 1960s (Davis 1963), and it is now the most commonly used stated preference method (Champ et al. 2003). Perman et al. (2011) presents some common steps when conducting a CV:

1. Questionnaire design (discussed in more detail below).
2. Choice of survey technique (e.g. face-to-face interviews, mail surveys, telephone interviews).
3. Identification of the relevant population and sampling.
5. Aggregation of WTP responses across the population.
6. Study evaluation.

The method is not free from critiques. Kahneman et al. (1999) criticise the CV method for collecting attitudes rather than preferences regarding certain goods, as values are affected by framing effects, sensitive to scope and context dependent. Others have found the opposite. Carson, Flores and Meade (2001) describe an approach where respondents are asked to value different levels of a good to test their responsiveness to scope. They conclude that poor survey design often is the cause to insensitivity to scope, a finding also strengthened by Veisten et al. (2004), who concluded that the elicitation format is important, rather than the method itself being weak. Carson et al. (2003) estimated non-use values in connection with the Exxon Valdez spill and found that respondents were sensitive to scope. Jacobsen et al. (2007) also arrived at this conclusion.

There are a number of ways to decrease uncertainty in results, such as questionnaire design, elicitation format, or handling of valuation uncertainty and protest bids. The next section looks at uncertainties and how they can be handled in the valuation process. In particular, protest bids had to be handled in the Lofoten case study.

3.1.1 Questionnaire design

The creation of a survey instrument includes for example constructing hypothetical scenario(s), determining payment vehicle and elicitation method. In general, the questionnaire starts by asking a number of questions regarding the topic, such as which environmental issues are of greatest concern to the respondent, how often they visit the area and why, and so on. The problem is then described in detail, sometimes with maps, figures and pictures. It is attempted to describe the problem in a comprehensible manner. After this the questionnaire presents certain credible projects, or measures, aiming at changing environmental status, either by improving its quality or by preventing further degradation. The experienced change is highlighted to the respondent. The payment vehicle describes how the payment will be designed. Some sort of national tax increase is a commonly used means, as are increased fees, either one-time payment or a yearly recurring payment. The WTP part usually ends by asking follow-up questions that elicit the reasoning behind certain answers, especially so-called protest bids. The last part of the survey gathers information about respondents, such as age, gender and income, and to what extent the survey was understood (Perman et al. 2011).

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2 The WTA concept will not be further elaborated here.
3.1.2 Elicitation format

The elicitation format describes how the WTP question is asked (Perman et al. 2011) and will have an impact on the information revealed, both to and from the respondents. The format influences incentives to strategic behaviour, as well as the amount of information needed to be revealed to the respondents. Therefore a number of aspects must be considered when choosing the format (Riera & Signorello n.d.). There are two main formats for eliciting WTP responses in CV studies: close ended and open ended. The difference between these two is the way the respondent is to choose the value of the good. In close-ended formats, respondents must accept or reject a given bid. The open-ended format, on the other hand gives the respondent the opportunity to state his or her own bid.

The close-ended format is the traditional elicitation format in that sense that it was recommended by the NOAA (National Oceanic and Atmospheric Administration) panel (Arrow et al. 1993). The method is very common and asks the respondent if he or she is willing to support or oppose a project at a certain cost, which is varied across the sample (Perman et al. 2011). The main arguments for close-ended formats are that people are more used to dichotomous choices, and that the open-ended format would encourage free-riding, which implies strategic overstatements (Arrow et al. 1993). When compared to dichotomous choice results, the open-ended WTP is usually lower. This has two implications: the biases in dichotomous choice (such as anchoring, rounding-up, or strategic over-bidding) inflate WTP, and the open-ended format underestimates WTP, or both situations occur (Bateman et al. 1995).

On the other hand, open-ended questions usually give many zeroes, a very small amount of very high bids, and few small bid (Carson et al. 2001). Further, an open-ended question can be difficult to answer and might lead to higher non-response rates (Perman et al. 2011). A version of the open-ended format is the interval open-ended format: here the respondent is asked to state his/her WTP as a self-selected interval, not a point. The format captures respondents’ uncertainty. It also makes it possible to interpret uncertainty, since the interval indicates how large the uncertainty is in relation to predetermined intervals, or brackets, this creates bias and forces respondents to choose an interval “somewhere in the middle”. Interval open-ended avoids this form of bias (Belyaev and Kriström 2010). In the Classic and Interval Open Ended (CIOE) format used by Håkansson (2008), the respondent can choose either to state an exact value (indicating low uncertainty) or to give an interval in which the WTP lies. The CIOE was used in paper I and II, and an example of this method is presented in Figure 3.

```
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.
```

Figure 3. An example of the CIOE elicitation format used in papers I and II.

3.1.3 Protest bids

Handling protest bids is a part of the analysis (step 4 in the process). When conducting a CV study, some respondents do not state their true WTP. They might state a very high bid, a zero bid, or refuse to bid at all. Not expressing their true value, of course, affects the results of the
study. If those who are categorised as protesters and state a zero bid are excluded from the calculations but in fact these same respondents have a WTP corresponding to the average WTP, then the value would be underestimated. If respondents with a zero WTP are wrongly categorised as protesters, then the value would be overestimated, since their zero bids should be included. The main reasons (Boyle in Champ et al. 2003) that protest bidders do not express their true WTP value are a) they do not understand the question but answer anyway, b) they are acting strategically in the sense that they are hoping that other people will pay so they do not have to reveal their true value, and c) they object to something in the survey. There is no general agreement on how to separate true zeroes from protest responses. But a common approach is to ask a number of debriefing questions for those respondents not willing to pay. One idea is that respondents state a value because they get satisfaction from giving—a warm glow—an idea put forward by Kahneman and Knetsch (1992). Also, environmental concerns have been found to affect WTP but in different directions. Dilemma concern is related to the prisoner’s dilemma, where respondents are tempted to let others pay for the good. Meyerhoff and Liebe (2006) have also found that these factors explain some of the variance. They also find that even though a respondent is willing to pay, he or she may hold protest beliefs affecting the decision. Their conclusion is that zeroes should be kept within the sample and regarded as true, even if there is a risk of underestimating the results. This theory was also tested by Bohn (1972) in the 1970s; no significant differences in WTP were found, despite different incentives for under- or overstating the bid.

In paper I, protest reasons such as “the polluter should pay”, “the measures are not reliable” and “the costs should be covered by existing taxes” were excluded from the analysis. Also, outliers above 10,000 Euros were excluded. In paper II, the share of protesters was high during the pre-studies. Therefore, actions were taken to reduce the protest bids. In the final survey, only budgetary reasons and that there are other, more important, issues, were provided to the respondent. All other reasons were to be stated as an open question. During analysis, those answering something like “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”, and “I have received too little information” were considered protesters. In both studies, only those refusing to pay were regarded as protesters, if the above conditions were filled.

3.1.4 Other biases

Despite these efforts to reduce uncertainty in the valuation process, there are problems with valuation. Environmental valuation involves uncertainties such as whether people actually would pay what they say they would. Another issue is part-whole bias, meaning that aggregating values for smaller habitats, for example, might lead to overestimating the total value. Also related to stated preference methods is the problem of how much information should be provided to the respondent. More information seems to increase WTP values. Ecosystem thresholds are generally not accounted for in valuations. Rather, ecosystem values are seen as linear. The reason is simply that there is a lack of data about marginal losses related to changes in nonlinear settings (Hanley & Barbier 2009).

3.1.5 Valuing risk

Objective risk can be said to consist of probability and consequence, often based on statistical findings on something actually occurring. The opposite is subjective risk, which is based on people’s opinion that there is a certain probability that something will occur. Rekola and Pouta (2005) find in a CV study that respondents are risk averse when the outcome is uncertain, and that they tend to overestimate small probabilities. This has implications for where to target measures and policy. Bickerstaff (2004) discusses what factors influence risk perception. Dread,
or the feeling of not having control, is one factor. Yet another is when the incident will occur, temporarily as well as how familiar one is with the risk.

Risk can be valued *ex ante* or *ex post*. Freeman defines an *ex ante* study as predicting “the physical and economic consequences of policies” while an *ex post* study “involves measuring the actual consequences of the policy” (Freeman III 1993 p.14). An *ex ante* approach includes valuing changed provision of ecosystem services based on the probability that an accident will occur and taking individuals’ risk preferences into account. An *ex post* approach, on the other hand, values the actual damage caused by, for example, an accidental oil spill. Although both types of approaches are challenging to undertake due to the uncertainty of measuring impacts on ecological impact, and its impact on human wellbeing, *ex ante* studies are particularly important for policymaking: for example, to support decisions on whether or not to explore oil and gas discoveries, or to estimate the resources needed for precautionary measures (Carson and Walsh 2006). When people are risk averse, the distinction between *ex ante* and *ex post* studies is more clear, since the option price, or the value of maintaining an ecosystem service, is higher in the *ex ante* approach than in the *ex post* (Freeman 2003).

*Ex ante* studies valuing environmental risk are scarce, and there is a particular dearth of studies that look at the valuation of oil spills. Valuation studies have been made for other areas, such as health risks due to hazardous waste (Smith & Desvousges 1987), radiation versus car accidents (Choi et al. 2001), or climate (Lorenzoni et al. 2005). These studies indicate that the marginal value of a change in risk decreases with increased risk (Smith & Desvousges 1987), that the perception of risk influences WTP depending on the object (Choi et al. 2001), and that risk is context specific and hence meaningful to look at only in that specific context (Lorenzoni et al. 2005). The latter conclusion implies that values of different risk-reduction measures cannot be compared in a CBA, since the values differ according to the proposed policies (Hanley & Shogren 2005).

Sometimes the concept of expected value of damage when valuing ecosystem services is used. This concept can be defined as the probability ($p$) that something will occur multiplied by the value ($D$) of the damage caused by that occurrence. In transport studies the concept has been used to some extent, where the value of “expected delay” has been analysed. Here, a probability $p$ that a train is delayed by $x$ minutes is presented to the respondent. A shortcoming with such an analysis is the assumption that the value of both the probability and the length of the delay are linear for low probabilities, an assumption that does not hold. The conclusion is that for delays with a low probability, respondents find that the length of the delay is not crucial, since any delay at all will destroy the whole travel plan (Börjesson & Elisasson 2008). These non-linear relations found in transport studies can also hold for other areas. The main problem with valuing risk is that objective risk differs from subjective risk, and thus that what the analyst seeks to value—a change in welfare due to scientific measures of risk (the objective risk)—does not correspond to what the respondents actually value, which is a perceived feeling of risk (the subjective risk). This implies a risk of suggesting wrong measures (Hanley & Barbier 2009).

We have now moved beyond Step 2 in the framework and have entered the third circle “Values” in Figure 2. Sections 3.2 and 3.3 present the valuation methods used for the ecosystem services identified in Section 2.

In both paper I and paper II, the handling of intervals when estimating mean WTP uses the approach suggested by Håkansson (2008) and Mahieu, Riera, & Giergiczny (2012). This approach uses the mean of the interval as the true WTP (WTP$_{MP}$). But in order to capture uncertainty, a lower and upper bound was also estimated by using the lowest and highest values in the interval
stated by the respondents, forming the mean $WTP_L$ (lower) and mean $WTP_U$ (upper), respectively.

### 3.2 Valuing ecotoxicological impacts on biodiversity in Swedish coastal waters affected by TBT (Paper I)

#### 3.2.1 The quantitative environmental assessment framework

An environmental assessment framework was developed in order to effectively illustrate improvement in environmental quality due to different policy actions (Table 1); this framework describes the impacts on different biota under certain levels of TBT. The level *Good* is based on the definition on good environmental status (GES). The other levels are developed so that natural degradation over 100 years will result in visible impact on the biota.

**Table 1.** The quantitative environmental assessment framework developed, consisting of environmental impacts of TBT on different groups of organisms.

<table>
<thead>
<tr>
<th>TBT concentrations</th>
<th>Environmental status</th>
<th>Growth disorders</th>
<th>Physical defects</th>
<th>Decreased reproduction</th>
<th>Increased mortality</th>
<th>Acute poisoning</th>
</tr>
</thead>
<tbody>
<tr>
<td>&lt;0.02 µg kg(^{-1})</td>
<td>Good</td>
<td>Snails and clams</td>
<td>Snails and clams</td>
<td>Snails, clams and crustaceans</td>
<td>Snails, clams and crustaceans</td>
<td></td>
</tr>
<tr>
<td>0.02-1.5 µg kg(^{-1})</td>
<td>Moderate</td>
<td>Snails, clams and fish</td>
<td>Snails, clams and crustaceans and fish</td>
<td>Snails, clams and crustaceans</td>
<td>Snails, clams, crustaceans and fish</td>
<td></td>
</tr>
<tr>
<td>1.5-100 µg kg(^{-1})</td>
<td>Poor</td>
<td>Snails, clams and fish, worms, insects and plants</td>
<td>Snails, clams, crustaceans and fish</td>
<td>Snails, clams, crustaceans and fish</td>
<td>Snails, clams, crustaceans, fish and worms</td>
<td></td>
</tr>
<tr>
<td>&gt;100 µg/kg</td>
<td>Bad</td>
<td>Snails, clams, fish, worms, insects and plants</td>
<td>Snails, clams, crustaceans, fish and plants</td>
<td>Snails, clams, crustaceans, fish and plants</td>
<td>Snails, clams, crustaceans, fish, worms, plants and amphibians</td>
<td></td>
</tr>
</tbody>
</table>

#### 3.2.2 Valuation scenarios

In the survey described in paper I, shells, clams, fish, crustaceans and plants living in and close to seabed sediments are used as examples to describe the overall impact on the coastal ecosystem service of *biodiversity*. Even though TBT is prohibited on all types of boats, there is still an inflow
into the seabed due to paint still attached to boat hulls. By doing nothing except for letting the toxic substance degrade, GES will not be reached even within the next 100 years due to the half-life of TBT. In order to make the survey both realistic and policy relevant, two scenarios are used: one where GES is reached in 2020, and one where the pace of natural degradation will go on until year 2100. Respondents were asked to state their WTP for the two scenarios, both of which would decrease TBT levels but over different time scales and to different levels.

The One-level scenario includes natural degradation of existing TBT substances and prevention of further additions by collection at boat-washing sites and boat dry dock areas. This assumes that the TBT levels would reach an improved level on the framework scale in year 2100.

The GES scenario, on the other hand, is more abrupt. It includes preventing further additions, as in the One-level scenario, as well as remediation of all polluted sites. By doing this, GES would be reached by 2020 and the environmental objective would be fulfilled.

In both scenarios funding was assumed to be a yearly tax per household, levied between 2013 and 2020. The scenarios are both described in Table 2.

The questionnaire was sent to an internet panel, representing a random sample of the Swedish population between 18 and 80 years old. In the end, 507 respondents finalised the questionnaire. It was preceded by a pilot study, indicating that no major changes were needed.

Table 2. The scenarios used in paper I.

<table>
<thead>
<tr>
<th></th>
<th>One-level scenario</th>
<th>GES scenario</th>
</tr>
</thead>
<tbody>
<tr>
<td>Status quo</td>
<td>Bad, poor and moderate status in all locations</td>
<td>Bad, poor and moderate status in all locations</td>
</tr>
<tr>
<td>Policy action</td>
<td>Only actions to prevent paint flakes from reaching the waters.</td>
<td>Remediation of sediment in combination with actions to prevent paint flakes from reaching the waters</td>
</tr>
<tr>
<td>After policy action</td>
<td>One level of improvement at every location by 2100</td>
<td>All locations GES by 2020</td>
</tr>
</tbody>
</table>

3.3 Valuing ecosystem services at risk from an oil spill in Lofoten (Paper II)

In this ex ante study of ecosystem services at risk from an oil spill, two scenarios (presented in Table 3) were developed that describe a future oil spill of about 60,000 tonnes due to a ship accident outside Lofoten-Vesterålen in northern Norway. The scenarios are based on a hypothetical but representative case in which a 283 m ship starts drifting and finally strikes some smaller islands. This causes the hull to break and 20,000 tonnes of crude oil to leak during the first day and 40,000 tonnes during over the following 3–4 days. The oil then drifts towards Lofoten and Vesterålen. The objective, statistically based, probability for such a spill is once every 350 years. If no measures are undertaken, increased transport will increase the probability to once every 150 years.
In the survey, the respondents were asked to state their WTP for limiting the risk of an oil spill in two different scenarios. The payment vehicle used was a yearly payment (tax) per household to a national fund earmarked solely for the purpose of reducing the risk of ship accidents causing oil spills in the particular area. The first scenario, the *Probability* scenario, asked for the respondents’ WTP for reducing the probability of an oil spill. The second scenario, the *Total risk* scenario, asked for the WTP for reducing the total risk, which consists of the probability that an accident will take place as well as the consequences on ecosystem services.

In the *Probability* scenario, it is assumed that the probability of an oil spill will increase from the current likelihood of once every 350 years to once every 150 years if no new measures are implemented. With certain new measures however—such as improved shipping routes—the probability will stay at once every 350 years despite increased sea traffic. Furthermore, the scenario implies that the consequences of an oil spill on ecosystem services will stay unchanged despite the new measures.

The *Total risk* scenario suggests measures for decreasing the probability and the consequences of an oil spill. In this scenario, measures to reduce probability are supplemented with measures targeted at reducing the negative consequences of an oil spill on ecosystem services. The change in probability is the same as in the first scenario: i.e., the probability of an oil spill will increase from the current figure of once every 350 years to once every 150 years if no new measures are implemented. With certain new measures, however, the probability will stay unchanged and the consequences on wildlife, fisheries and coastline would last only half as long as they would with no new measures.

Table 3. Impacts of the implementation of measures on probability of an oil spill and consequences to ecosystem services, as presented in the *Total risk* scenario. If Wildlife, Fisheries and Coastline are excluded the table also presents the *Probability* scenario.

<table>
<thead>
<tr>
<th>Total risk scenario</th>
<th>Without additional measures</th>
<th>With additional measures</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Probability of an oil spill</strong></td>
<td>Once every 150 years</td>
<td>Once every 350 years</td>
</tr>
<tr>
<td><strong>Wildlife</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Birds</td>
<td>Impacted share of population</td>
<td>Length of impact</td>
</tr>
<tr>
<td></td>
<td>50%</td>
<td>Some months to some years</td>
</tr>
<tr>
<td>Mammals</td>
<td>30%</td>
<td>Some months to some years</td>
</tr>
<tr>
<td>Fish</td>
<td>10%</td>
<td>Some months to some years</td>
</tr>
<tr>
<td>Fisheries</td>
<td>Restrictions for 2–3 years</td>
<td></td>
</tr>
<tr>
<td>Coastline</td>
<td>1000 km, 4 months</td>
<td>500 km, 2 months</td>
</tr>
</tbody>
</table>

The questionnaire was sent out, using an internet panel. Since one of the purposes of the valuation study in Norway was to analyse distributional effects, the survey was sent to a random sample of the population, older than 18 years, in the three northernmost counties in Norway: Nordland, Finnmark and Troms. 400 respondents finalised the questionnaire. The respondents were also, among a number of things, asked to rate ecosystem services identified in the area on a five-graded scale.
3.4 Cost-benefit analysis

CBA is widely applied to provide decision-makers with advice when evaluating projects and policies. There are several EU directives requiring the use of CBA, such as the Marine Strategy Framework Directive (MSFD) (EC 2008) and the Energy Efficiency Directive (EED) (EC 2012). On a more local level, the implementation of the EU Water Framework Directive (WFD) in Sweden requires that costs and benefits be quantified when measures are developed (SFS 2004).

Several authors have described CBA (see, e.g., Perman et al. 2011; de Rus, 2010; Johansson, 1993; Sugden & Williams, 1978). In a CBA, changes in welfare due to some project (or policy or measure) are estimated. According to CBA theory, the project should be recommended for implementation if it improves the welfare of society, i.e. if the benefits exceed the costs. Welfare is difficult to estimate, and often the consequences associated with the project are quite varied. Therefore, money is used as a common measure. All costs and benefits arising from the project are relevant, i.e. all social costs and benefits. This means that not only are consequences for private actors considered but also consequences for people not involved in any payments for the project. A project, in this case may be a physical undertaking, such as a construction project, or it may be a policy or measure.

de Rus (2010) describes a number of steps that need to be included in CBA. First, the objective (What is the problem to be solved?) of the project must be defined, together with relevant alternatives for solving the problem. Relevant alternatives aim at achieving the same objective, and the reason for their inclusion is important. According to de Rus, it is not enough to simply present a project with a positive net present value (NPV). One must also show that the net benefit is the greatest among the different solutions. Except for relevant policy alternatives, the base case, or business-as-usual (BAU), must also be specified. BAU describes what will happen if no project is undertaken but circumstances, such as changes in demand, change in the same way in both the BAU and the proposed scenario(s). When the project is defined, all costs and benefits (or consequences) deriving from it must be defined. Consequences may be direct and indirect, where direct effects happen on the market and indirect ones occur outside the market. Finding direct costs and benefits is usually the easiest part. It is important to study several categories of costs and benefits. Carson et al. (2003) suggest that non-market values can be substantial. (Kinell et al. (2012) stress that it is important to include ecological expertise as well, so as to ensure that the views of various stakeholders are considered and that the analysis includes environmental thresholds and effects on ecosystem services.

The sizes of costs and benefits, or values, are then estimated, i.e. measured. In some cases values can be found using market prices. In other cases value must be derived by asking people about their willingness to pay (see section 3.1). Costs and benefits are then aggregated over the project horizon to get the NPV. Usually this is performed using a discount rate greater than zero, implying that present consumption is given more weight than future consumption. It is also possible to put more weight on aspects such as consequences to poor people. The choice of discount rate, as well as weighting of different groups in society, is a subjective policy decision. Finally, the NPV is interpreted. If the NPV is greater than zero compared to BAU, social welfare is increased. But it is important to present the financial part of the analysis as well: i.e. the costs and benefits imposed on the private sector. This is because the actual budget might not cover the project’s expenses even though it is beneficial for society, and because different policy designs affect distributional effects differently. An example of the latter is the construction of a toll road where a number of price structures exist, each affecting different types of travellers (de Rus 2010).
The main strength of CBA is that it tests for economic efficiency in a world of scarce resources. The process sheds light on all impacts (positive or negative) associated with the project and that are able to be included. All information is presented in a structured way, and it provides a basis for further discussions among decision-makers. Social values, such as preferences of the general public, are included, as are behavioural predictions, helping to avoid future surprises. Such values might not be considered if no CBA were undertaken (Hanley 2001).

CBA has been the subject of critique, however. In addition to the problems related to environmental valuation and risk already discussed, the choice of a discount rate implies uncertainties in the estimates. A correct rate cannot be found, as political and personal principles guide the choice. CBAs should always include a sensitivity analysis that presents different outcomes depending on the discount rate. Some critiques examine the underlying principles of CBA, such as the assumption that gainers theoretically can compensate losers and still be better off if social wellbeing is positive (The Kaldor-Hicks test), or the assumption that people’s preferences are a proper way of guiding societal decisions (Hanley & Barbier 2009). These critiques are discussed in more detail below.

The Kaldor-Hicks compensation principle (a loose form of Pareto optimality) means that gainers theoretically could compensate losers such that in the end no one is worse off after a project is implemented. In CBA, this is met by only calculating a positive net present value. A problem with the Kaldor-Hicks assumption is that in some cases it might not be possible to compensate losers. Some people might even refuse any compensation. This could theoretically obstruct a whole project, even though the rest of the society would benefit from it (Hanley & Barbier 2009; Hanely 2001). Hansson (2007) discusses this principle and concludes that this approach is used differently in different contexts: e.g. in clinical trials the case of a patient exposed to risks could be outweighed by the expected social benefit if a CBA approach was used. This is not how it is done in practice. When it comes to decisions such as recommendations on the intake of polluted fish, recommendations are not based on a full CBA either (Hansson 2007). Another problem is the assumption that changes in utility can be compared across places and generations and that the marginal utility is the same for each person. Also, one objection concerns what sorts of decision-making CBA is relevant to. Can policies that are ethically complicated be assessed using a CBA, for example? Examples include ethical concerns such as banning slavery or child labour. The goal of sustainable development is another example. A CBA assesses economic efficiency over time. Development can be defined as sustainable if total wealth does not decline over time. Depending on how the Kaldor-Hicks criterion is interpreted and whether weak or strong sustainability is assumed, CBA may or may not be compatible with sustainable development (Hanley & Barbier 2009; Hanley 2001). This is also related to the choice of discount rate mentioned earlier. Hanley and Barbier (2009) and Hanley (2001) further argue for that CBA cannot guarantee the life support on which humans depend. CBA should instead be used to assessing alternatives set by politicians where the ethical concerns are already accounted for and moral limits are set. Recall also the discussion on marginal changes. Thresholds and points where life support cannot be guaranteed are hardly marginal changes.

Sections 3.5 and 3.6 present the CBAs in paper I and III. The CBAs could provide input to policy decisions and are in the framework included in Figure 2’s Step 4, Policy implications. Also the generic weighting set (section 3.8) and LCC (section 3.9) provide input into decision-making and thus have policy implications.
3.5 Costs and benefits of reducing TBT levels in Swedish coastal waters (Paper I)

In paper I, a CBA compared remediation costs with the benefits of reducing TBT-levels in Swedish marine waters. The scenarios in the valuation study and the CBA are connected to both the ecological preconditions and the political goal of reaching good environmental status (GES).

Two scenarios were compared against BAU, i.e. doing nothing. The first scenario, GES scenario, was in line with the policy goal of reaching GES in Swedish waters with respect to toxic TBTs by 2020. In it, measures are implemented aiming at removing TBT to harmless levels from the sediments, as well as preventing further spread of TBT. The One-level scenario only involves preventing further spread of TBT and letting natural degradation decrease pollutant levels in Swedish waters. In the One-level scenario, each location would improve its environmental status in the quantitative assessment framework by one level at 2100 (Table 1).

The mean WTP estimates elicited in the valuation study were aggregated for the whole population of Sweden to illustrate the benefits. Respondents were asked to answer as a household; hence, estimates were multiplied by the number of households to arrive at a national estimate. Estimates on costs for boat washers were used in both scenarios, and the GES scenario also used estimates of remediation costs. Thus, the two scenarios aimed to demonstrate the consequences of political goals with the help of the framework presented in Table 1 rather than by comparing different measures. In the CBA, all estimates—both costs and benefits—were discounted by 0 and 6% over 8 years, illustrating the timeframe for GES to be reached according to the MSFD (EC 2008). Eight years was the time horizon in the GES scenario, and the same period was used for payments in the One-level scenario.

Dredging costs were estimated based on the assumption that the amount of material to be treated was about 1.5 million m$^3$ for 40 sites (Holm et al. 2007). Estimates were also calculated for 1,000 sites, as the literature shows that the number of harbours may approach that figure (if.se 2012). Dredging cost estimates varied depending on the method used. Thus, a lower estimate was based on USD 42 per m$^3$ and a higher estimate was based on USD 315 per m$^3$ (Holm et al. 2007; Magnusson et al. 2006). The cost range for small boat washers was USD 0.12–0.16 million, and the cost range for large boat washers was USD 0.15–0.19 million. In addition to these costs, costs for washing facilities including filters and tanks (but no maintenance), which were estimated at between USD 22,400 and 49,000 per year, were included (Havs- och vattenmyndigheten 2012).

3.6 Costs and benefits from decreasing the probability of oil spills in Lofoten (Paper III)

The CBA in paper III describes a situation where maritime traffic outside Lofoten-Vesterålen in northern Norway increase, and along with it the risk of oil spills. Risk is divided into probability and effect, where only a change in probability is presented in the CBA. If no measures are taken, probability of a spill from a ship accident is expected to increase from .003 to .007 (one in 150 years). This is the BAU, if we also consider no changes in existing equipment. This scenario is compared with the case where certain measures keep probability at .003 (once in 350 years). These long-term perspectives are based on current knowledge of the probability of a 20,000 tonne spill in northern Norway. The study estimated costs for improved tugboats. The potential impact of the measures on ecosystem services was not addressed in the analysis; focus was on a change in probability of an oil spill and the impacts on ecosystem services were kept constant.
The conflict here concerns the choice between using the sea for shipping and the risk that an oil spill would entail for affected ecosystems services such as fishing and recreation. The costs and benefits to different actors of reducing the probability of an oil spill illustrate the conflict in terms of differing preferences. Different actors are affected in different ways by a reduction in probability, both in terms of what they gain (benefits) and what they have to pay (costs). Finding these costs and benefits for different actors illustrates their preferences and the distribution of such preferences. Since the analysis concerns a possible future incident, the concept of expected values has been used: i.e., costs are defined as costs for measures that reduce the probability of a spill, while benefits are used in two ways. The first is connected to the valuation study in paper II, where the value of avoiding loss of ecosystem services is presented. The second is called expected avoided loss of income. This is based on a study in the literature that presents loss of income due to a spill. In order to relate these losses to their probability of occurring, the figures have been multiplied by the change in probability. The separation of benefits into different groups implies a risk of double counting, as does adding them together. This is because in theory, results from a CV study include all values related to the scenario presented—both use and non-use values. The identified costs are the costs of the measures—including, for example improved equipment, such as tugboats. When identifying costs for policy instruments and measures we note that in many cases they reduce both the probability of an oil spill and its effects on ecosystem services. This means that it is likely that the costs presented in this study will also have other, positive effects beyond those that this case study examines, such as limiting the spread of oil and thus shortening recovery times.

Many industries and stakeholders would be affected by an oil spill. However, in paper III the number of stakeholders that gain from policy measures that reduce the probability of a spill is limited to the Norwegian population engaged in the fishing and tourism industries in the three northernmost counties. Since only the benefits for the three northernmost counties in Norway are included, the total benefit will be an underestimation of the total value of the ecosystem services compared to the outcome if a larger geographical area had been considered. Fisheries and tourism are important sectors in the area (e.g. Midtgard et al. 2012), and it is assumed that their inclusion will make the analysis representative for a conflict in the Arctic. The purpose is to illustrate how different industries are affected by a spill in relation to the total value (avoided loss of ecosystem service values), not to present exact values. All estimates were discounted by 0 and 6% over 15 years, illustrating the length of visible effects from the measures. The costs were assumed to be evenly distributed over the first three years.

### 3.6.1 Costs of reducing probability of an oil spill

The cost analysis in paper III is based on an investigation by Kystverket (2012) assessing costs and benefits of an increased number of state-owned tugboats in Norway compared to today’s level. According to their analysis, the probability of a spill will decrease nationwide from 1/20 to 1/30, (appr. 30%) annually. Other measures and policy instruments aiming at reducing the probability of a spill can be expected to exist, but cost estimates for those have not been found. Neither have costs for reducing only low-probability spills, better corresponding to the scenario in paper II, been found. In Kystverker’s analysis no extra tugboats would be operated in northern Norway. On the other hand, the three existing tugboats would be replaced by larger vessels with extra capacity (also decreasing the impacts of a spill). The estimated cost of one ship is NOK 280 million (2011, less VAT). The ships would be delivered during 2014–2018. Operational costs are excluded from the analysis, since they are also included in the BAU option and can be cancelled out. It is assumed that annual maintenance costs for the new tugboats would be the same as those for the existing vessels, since they are of similar size.
3.6.2 Avoided loss of ecosystem service values

In order to find values for the benefits produced by the measures, estimates from the valuation study in paper II were used. The mean WTP estimates were aggregated to the population of the three northernmost counties\(^3\) and used as the values for avoided loss of ecosystem services due to the measures.

3.6.2.1 Expected avoided loss of income

Since the CBA seeks to analyse distributional effects, estimates of benefits for different industries were presented. Figures for avoided loss of income in the fishing and tourism sectors in the relevant counties were identified. The figures have been adjusted to 2012 price levels (SSN 2014). The figures were further adjusted by the change in probability, i.e. –57.14\%, to illustrate the expected value, evenly distributed over 15 years, discounted and then summed.

Fisheries

The most important commercial fishing species in the area are cod and herring, and the effects on these two species are used to illustrate the impact on fishing industry revenues. Faugstad (2010) estimates NOK 3.2 billion to be cost of possible loss of cod landings due to a spill from an underwater source lasting for 28 days, accumulated over 15 years (assuming 2010 price levels and a price per kilo of NOK 16.8). These costs are directly linked to the fisheries sector; processing industries are not accounted for. The daily spill in Faugstad (2010) was 4,500 tonnes which resulted in a total spill of 126,000 tonnes; this is a much larger spill than the one used in paper III analysis. The damage cost is thus likely to be smaller if adjusted to the scenario used in paper III. There are also other uncertainties, such as season, weather and location, which may raise or lower the cost of a spill. Ibenholt et al. (2010) presents estimates for the reduction in herring landings. A 28-day blowout (4,500 tonnes per day) is calculated to result in a reduction in landings valued at NOK 230 million, accumulated over 15 years (assuming 2007 price levels and a price per kilo of NOK 2.50). As was the case in the cod example above, the processing industry is not included in the analysis. In both studies, effects from the blowout will be entirely dissipated after 15 years.

This analysis uses figures from Faugstad (2010) and Ibenholt et al. (2010) to illustrate impacts on the fishing sector.

Tourism

The reduction in tourism based on an actual spill, has been discussed in earlier literature. Forland (2003) found that the reduction in tourism might be between 20 to 50\%, depending on how close the tourism zone is to the source of the spill. For specific severely affected groups, the loss might be as high as 80\%. The effect would be worst in the beginning, but still be noticeable after four or five years. Not only are the physical aspects important for the reduction but also psychological factors, depending on the amount of information disseminated about the spill. Clean-up activities will have a short term positive impact due to demand for beds and food during the work. Also, the spill might attract curious individuals, environmental organisations and journalists to the area. Furthermore, Forland has found that the specific loss to Lofoten due to one spill (of undefined volume) is estimated at over NOK 110 million (1995) for five years, with hotels, restaurants and

\(^3\) In 2012 the population of Nordland, Troms and Finnmark was in 474,563, the average household size was 2.2 persons (SSN 2013a), and the number of households was 215,710 (SSN 2013b).
transport providers being the most severely affected. For Nordkapp\textsuperscript{4} the figure would be NOK 172 million (2002) (Førland 2003). Haugberg (2010) finds that the negative consequences to tourism of a ship accident would last for about five years and amount to approximately NOK 677 million in the affected areas of Lofoten, Vesterålen and Senja. The figures from Førland (2003) and Haugberg (2010) have been used to illustrate the impacts on the tourism sector in the analysis.

3.7 Value transfer and generic weighting sets

Often it is too time-consuming or expensive to be practically feasible to find new values. Ahlroth et al. (2011) find that weighting and valuation is used in several environmental systems analysis tools, and that monetary weighting is useful. They further conclude that there is a need for generic weighting sets.

One solution to this dilemma of needing values but not having the resources to find them is value transfer\textsuperscript{5}, which transfers values from one study site to another policy site. In a simplified explanation, there are two main approaches of transferring a value: either a single value is transferred, or else a function. There are several problems with value transfer: measurement errors (i.e. uncertainty within the original study); generalisation or transfer errors; and publication selection bias, which is the result when the publication process steers the available publications towards certain types of ecosystem values and results (Pascual et al. 2010; Johnston & Rosenberger 2009). Lack of correspondence between the study site and policy site is a transfer error and can be seen when the characteristics of the population in one site do not necessarily correspond to those in the policy site. For example, income and education levels, religion or ethnicity might not be the same. Also, the sites are usually not identical (Pascual et al. 2010; Navrud & Ready 2007). When eliciting the WTP in a valuation study, a specific discrete change in welfare is used and not a general marginal change. This makes generalisations difficult. Also, when there is a difference in both the initial level of welfare and in the direction of change, unit value transfer is often not feasible. Despite these shortcomings, value transfer is commonly used (Johnston & Rosenberger 2009), as are generic weighting sets (Ahlroth et al. 2011). There are differences in the use of generic weights and values between different applications. In CBA, absolute values are of importance, while in tools such as LCA it is instead the relation between weights that is important. The characteristics of the particular pollutant also have an impact on whether generic values are relevant or not. When the study object is geographically limited, site-specific values are more important than if the problem is more general and measures can be taken anywhere (Ahlroth et al. 2011).

A recent review by Pizzol et al. (2014) analyses different monetary weighting sets for LCA. They underscore the importance of applying the monetary weighting at relevant points in time (i.e. endpoint or midpoint). They do not, however, recommend CV as a method for weighting LCA, since the focus of a CV study is rather narrow compared to the impact categories which are complex and includes many variables. Despite this, the weighting set Ecovalue08 (Ahlroth & Finnveden 2011), partly using CV results, is considered to have a rather high relevance for weighting LCA (Pizzol et al. 2014). Value transfer is common in CBA, despite the obvious problems. Generic values are useful for LCC. But external effects are generally not included in LCC analyses, since the costs generally should be costs directly linked to the company (Ahlroth et al. 2011). Ahlroth et al. further list a number of advantages of generic weighting sets.

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\textsuperscript{4} Defined as the municipalities of Nordkapp, Måsøy, Hammerfest, Kvalsund, Porsanger and Alta (Førland 2003).

\textsuperscript{5} Another common term is benefit transfer. But as Navrud and Ready (2007) suggest, costs can also be transferred and thus they propose the term value transfer instead.
Communication is simplified, and studies might be more readily accepted when the sets are well known and frequently used. Transparency increases and makes comparisons possible. Finally, the approach saves time and money. Johnston and Rosenberger (2009) identify the need for an increased number of streamlined studies providing empirical estimates making the process more transparent.

### 3.8 An updated weighting set (Paper IV)

Paper IV is based on Ahlroth and Finnveden's work (2011), which develops a weighting set, Ecovalue08, for the impact categories of eutrophication, acidification, global warming, forming of tropospherical ozone, human health and depletion of resources. Paper IV adds two more impact categories: particulates and marine toxicity, and updates several of the values in Ahlroth and Finnveden (2011). All the values are adjusted to 2012 price levels and presented in Euros. Single value transfer based on more recent studies was used in paper IV to update the values. The value for marine toxicity is based on the results from the valuation study presented in paper I. The mean WTP for the One-level scenario is multiplied by the number of Swedish households and discounted over eight years (2013–2020). The One-level scenario was chosen over the GES scenario, since the former scenario was regarded as practically feasible. The total WTP was then divided by the total amount of TBT released between 1965 and 2001 (Kemikalieinspektionen 2012; Naturvårdsverket 2012). The estimate for SEK/released amount of TBT was finally divided by the characterisation factor for TBT in the Recipe method, allowing for compatibility with LCA data. The final step implies that the original value for TBT can be used for a number of other substances, since it is transformed into an equivalent.

### 3.9 Interviews on environmental costs in LCC (Paper V)

Paper V administered 13 interviews and seven questionnaires to representatives from different industry sectors in Sweden, such as real estate companies, manufacturing companies, vehicle industry and Swedish authorities within defence and transport, to collect information regarding the use of LCC. Open-ended questions were used in both the interviews and the questionnaires. The interviewees were all familiar with LCC in some way. The questions concerned the LCC method used, whether environmental costs are considered, the purpose of using LCC, how the results are used and the importance of the results for decision-making within the organisations.

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6 A discount rate of 3.5% was used, recommended by the Swedish Transport Administration (Trafikverket 2012), and 2012 was the base year.

7 The characterisation factor is a measure of the potential environmental impact of a specific substance. The factor for each impact category is defined as a common denominator (e.g. the release of H+ causing acidification from SO2, NOx etc.), and the factor indicates the relation between different substances causing the same impact (Baumann & Tillman 2004).

8 8390 kg 1,4-di-chloro-benzene equivalents per kg TBT (Goedkoop et al., 2009).
4 Results

The section begins with results from the valuation study and CBA regarding ecosystem services in the Swedish coastal area (paper I), followed by results from the valuation study of ecosystem services in northern Norway (paper II). The findings in paper II constitute the basis for the CBA concerning measures for decreasing probability of an oil spill (paper III) in the same region. In paper IV the weighting set, partly based on paper I, is presented. The chapter is finalised by the results from paper V describing how practitioners consider the use of environmental costs in LCC.

4.1 The values of ecotoxicological impacts on biodiversity in Swedish coastal waters (Paper I)

4.1.1 The valuation study

In paper I respondents were asked to state their WTP for measures that would improve environmental quality in two scenarios: One-level and GES. The different proportions of respondents’ willingness to pay for the measures, as well as the share of protest bids, are presented in Table 4. A simple t-test showed that there is no significant difference between the two scenarios regarding protest bids (within a 95% confidence interval), implying that the scenarios did not have an effect on the number of protests. The rates of protesters were not remarkably high; therefore no specific attention was paid to the formulation of the WTP question when it comes to provoking respondents in some way or their reasons for protesting. Those who had some objections to the scenarios (e.g., “I don’t believe the measures would work,” “the polluter should pay,” “there is too little information”, etc.) were considered protesters.

<table>
<thead>
<tr>
<th></th>
<th>WTP &gt;0</th>
<th>Zero WTP</th>
<th>Protest bids</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>One-level scenario</td>
<td>68.3 (366)</td>
<td>10.4 (56)</td>
<td>21.3 (114)</td>
<td>100 (536)</td>
</tr>
<tr>
<td>GES scenario</td>
<td>71.6 (384)</td>
<td>9.1 (49)</td>
<td>19.2 (103)</td>
<td>100 (384)</td>
</tr>
</tbody>
</table>

The elicitation format (CIOE) gave respondents the chance to answer either with an exact bid or with interval range. In this case there was no significant difference between the scenarios, indicating that the uncertainty perceived was of similar magnitude. In both cases the share answering with an exact bid was smaller than the share answering with a range. The shares are presented in Table 5.

<table>
<thead>
<tr>
<th></th>
<th>Exact</th>
<th>Range</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>One-level scenario</td>
<td>17.5 (64)</td>
<td>82.5 (302)</td>
<td>100 (366)</td>
</tr>
<tr>
<td>GES scenario</td>
<td>17.7 (68)</td>
<td>82.3 (316)</td>
<td>100 (384)</td>
</tr>
</tbody>
</table>
The mean WTP estimates (WTP$_{MP}$, WTP$_{L}$ and WTP$_{U}$) show a tendency to be higher for the GES scenario than for the One-level scenario, but a Mann-Whitney test on all mean WTP estimates shows that the differences are not significant. The results are presented in Table 6.

Table 6. Mean and median WTP estimates for the scenarios. Per year and household (in USD, confidence interval p < 0.05 in parentheses).

<table>
<thead>
<tr>
<th></th>
<th>One-level scenario</th>
<th>GES scenario</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Mean WTP$_{MP}$</strong></td>
<td>107.79 (89.12–126.45)</td>
<td>119.31 (100.58–138.04)</td>
</tr>
<tr>
<td><strong>Mean WTP$_{L}$</strong></td>
<td>73.65 (58.11–89.19)</td>
<td>82.84 (66.39–99.29)</td>
</tr>
<tr>
<td><strong>Mean WTP$_{U}$</strong></td>
<td>129.26 (108.98–149.56)</td>
<td>156.74 (132.51–180.99)</td>
</tr>
</tbody>
</table>

### 4.1.2 Costs and benefits of reducing TBT levels

All estimates are presented in Table 7. The net benefit varies depending on the discount rate, the number of sites in need of remediation and whether higher or lower values for costs and benefits are used. According to the sensitivity analysis, the measures described in the One-level scenario are always beneficial even when lower WTP and higher costs are used, independent of the discount rate used. When 40 sites are considered the net benefits are even higher. The GES scenario, on the other hand, is always beneficial except when high costs estimates are considered for 1,000 sites.

Given that the approach in the GES scenario was designed to lead to the political goal of good status being achieved in Swedish waters by 2020, it is more reasonable to assume that 1,000 (rather than 40) sites needs to be cleaned up. This means that the installation of washing facilities—in this example one boat washer and one cleaning facility with collection in each site, combined with dredging and treatment of polluted sediment—might be beneficial for the Swedish society depending on remediation prices and WTP estimates. These measures would also lead to achieving GES in 2020.

Table 7. Benefits, costs and net benefits for both scenarios in USD with different discount rates. The upper and lower WTP estimates are combined with the higher and lower cost estimates. Numbers for 40 sites within parentheses.

<table>
<thead>
<tr>
<th></th>
<th>Benefits</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>One-level scenario</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>%</strong></td>
<td>WTP$_{L}$</td>
<td>WTP$_{U}$</td>
<td>WTP$_{L}$</td>
</tr>
<tr>
<td>0</td>
<td>2.81*10$^7$</td>
<td>4.93*10$^8$</td>
<td>3.16*10$^9$</td>
</tr>
<tr>
<td>6</td>
<td>2.31*10$^7$</td>
<td>4.06*10$^8$</td>
<td>2.60*10$^9$</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th></th>
<th>Costs</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Low</td>
<td>High</td>
<td>Low</td>
</tr>
<tr>
<td><strong>%</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>0</td>
<td>1.41*10$^7$</td>
<td>2.39*10$^8$</td>
<td>1.41*10$^9$</td>
</tr>
<tr>
<td></td>
<td>(7.92*10$^6$)</td>
<td>(1.17*10$^7$)</td>
<td>(7.92*10$^6$)</td>
</tr>
<tr>
<td>6</td>
<td>1.16*10$^7$</td>
<td>1.96*10$^8$</td>
<td>1.16*10$^9$</td>
</tr>
<tr>
<td></td>
<td>(6.52*10$^6$)</td>
<td>(9.72*10$^6$)</td>
<td>(6.52*10$^6$)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th></th>
<th>Net benefits</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Low</td>
<td>High</td>
<td>Low</td>
</tr>
<tr>
<td><strong>%</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>0</td>
<td>2.67*10$^7$</td>
<td>2.57*10$^8$</td>
<td>4.79*10$^9$</td>
</tr>
<tr>
<td></td>
<td>(2.80*10$^6$)</td>
<td>(2.80*10$^7$)</td>
<td>(4.92*10$^8$)</td>
</tr>
<tr>
<td>6</td>
<td>2.20*10$^7$</td>
<td>2.12*10$^8$</td>
<td>3.94*10$^9$</td>
</tr>
<tr>
<td></td>
<td>(2.31*10$^6$)</td>
<td>(2.30*10$^7$)</td>
<td>(4.05*10$^8$)</td>
</tr>
</tbody>
</table>
4.2 The values of ecosystem services at risk from oil spill in Lofoten (Paper II)

4.2.1 The valuation study

Respondents rated ecosystem services in the area on a scale from 1 to 5, where 1 = ‘no importance at all’ and 5 = ‘of crucial importance’. The respondents could also choose ‘Don’t know’ as an alternative.

A paired sample t-test was used to test differences between the different categories. The lower grades (1–2) were grouped for each ecosystem service into a new category called ‘Less important’, and the higher grades (4–5) were grouped in a similar manner to form a new category called ‘More important’ (see Table 8). A pairwise comparison between the category ‘Less important’/’More important’ for one ecosystem service and the corresponding category for each other ecosystem service was carried out. For example it was tested whether or not there was a statistically significant difference between recreation and cultural values for the category ‘Less important’. In this case no significant difference was found (within a 95% confidence interval).

‘Clean and non-polluted sea areas’ was found to be more important compared to all other services at a statistically significant level. In addition, ‘biodiversity’ and ‘aesthetic values’ were significantly more important than most other ecosystem services. ‘Open sea areas, transport routes’ was found to be less important at a statistically significant level (within a 95% confidence interval) compared to all services except for ‘Recreation’ and ‘Cultural values’.

The findings are in line with earlier studies showing that existence and recreation values in aquatic ecosystems are important for respondents (Eggert & Olsson 2009; Kosenius 2010; Johnston et al. 2011; Östberg et al. 2012). When comparing the ‘More important’ ecosystem services, the significant differences are more common than when comparing the ‘Less important’ ecosystem services.

### Table 8. Less important’ and ‘More important’ ecosystem services (in percent, frequency within parentheses). The lines do not add up to 100%, as the middle alternative is not included.

<table>
<thead>
<tr>
<th>Ecosystem service</th>
<th>Less important</th>
<th>More important</th>
<th>Don’t know</th>
</tr>
</thead>
<tbody>
<tr>
<td>Recreation</td>
<td>13.3 (53)</td>
<td>56.0 (224)</td>
<td>6.0 (24)</td>
</tr>
<tr>
<td>Cultural values</td>
<td>11.3 (45)</td>
<td>58.0 (232)</td>
<td>4.8 (19)</td>
</tr>
<tr>
<td>Food</td>
<td>8.8 (35)</td>
<td>70.0 (280)</td>
<td>3.8 (15)</td>
</tr>
<tr>
<td>Aesthetical values</td>
<td>8.5 (34)</td>
<td>69.3 (277)</td>
<td>3.5 (14)</td>
</tr>
<tr>
<td>Biodiversity</td>
<td>7.8 (31)</td>
<td>70.3 (281)</td>
<td>4.3 (17)</td>
</tr>
<tr>
<td>Clean and unpolluted areas</td>
<td>5.8 (23)</td>
<td>76.5 (306)</td>
<td>4.8 (19)</td>
</tr>
<tr>
<td>Open sea areas</td>
<td>12.0 (48)</td>
<td>52.5 (210)</td>
<td>7.0 (28)</td>
</tr>
</tbody>
</table>

For both scenarios Table 9 presents median WTP\textsubscript{MP} and mean WTP (WTP\textsubscript{MP}, WTP\textsubscript{L}, WTP\textsubscript{U}) estimates. When protesters are excluded from the analysis, the median is NOK 250/year/household for the Probability scenario and NOK 300/year/household for the Total risk scenario. The mean WTP\textsubscript{MP} estimates vary between NOK 851 for the Probability scenario and NOK 888 for the Total risk scenario. A Mann-Whitney test shows that the differences in median WTP, mean WTP\textsubscript{MP}, or mean WTP\textsubscript{L} and WTP\textsubscript{U} are not statistically significant (within a 95% confidence interval). That is, even though respondents present a higher WTP if both probability and consequences are changed, compared to if only probability is changed, this difference cannot be confirmed statistically. The fact that there are no statistical differences in mean WTP between
the scenarios is not uncommon in CV-studies (e.g. (Ahtiainen et al. 2012; Håkansson 2009). The lack of significant differences between the scenarios may be accounted for by closeness to the valued good, or the presentation of the good in the survey (Ahtiainen et al. 2012). It also may be that marginal WTP is low and quickly diminishing (Håkansson 2009).

The pilot studies showed high protest shares (around 50%). This was reduced to about 30% by underscoring the importance of consumption of the general public as a cause for increased transport and by reducing the number of possible reasons for protest available in the survey. In the final version only budgetary reasons were suggested in advance and were not defined as protest reasons. For all other reasons, respondents were given the opportunity to freely state their thoughts. Socio-economic factors among protesters were analysed using a simple t-test. It seems that gender (men protest significantly more, at a 95% confidence interval) and working area yield significant differences (at a 95% confidence interval) in protesting. People working in the fishing, oil, maritime transport, tourism and farming sectors are significantly more prone to protesting and to state zero WTP (95% confidence interval) than those working outside these sectors. The importance that people placed on nature had a positive effect on the tendency to protest. This was tested by dividing respondents into two groups and merging respondents stating more environmental concern, and respondents stating less environmental concern. The two groups were compared using a t-test on a 95% significance level.

Table 9. Median WTPMP and Mean WTP (mean WTP_{MP}, WTP_{L}, WTP_{U}) when protest bids have been excluded (NOK/year/household) (confidence interval within parentheses).

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Median WTP_{MP}</th>
<th>Mean WTP_{MP}</th>
<th>Mean WTP_{L}</th>
<th>Mean WTP_{U}</th>
</tr>
</thead>
<tbody>
<tr>
<td>Probability scenario</td>
<td>250</td>
<td>851 (637.5–1065.0)</td>
<td>672 (461.2–882.3)</td>
<td>1022 (784.0–1259.8)</td>
</tr>
<tr>
<td>Total risk scenario</td>
<td>300</td>
<td>888 (669.8–1105.4)</td>
<td>714 (498.8–930.1)</td>
<td>1094 (834.9–1344.1)</td>
</tr>
</tbody>
</table>

4.2.2 The perception of risk of an oil spill

In order to investigate whether perceived risk had any influence on respondents’ WTP, early on in the study respondents were asked to estimate the probability of an oil spill in the Lofoten-Vesterålen area. This question was asked before information on objective probability (once every 350 years) was provided.

The answers ranged from 0 to 5,000 years. In order to compare willingness to pay depending on perceived level of risk, respondents were divided into three groups based on their subjective judgement of probability: once every 0–20 years (n = 114), 21–100 years (n = 123) and 101–5,000 years (n = 24). The third group is rather small, meaning few people have an accurate belief about the low probability of an oil spill. Only 6% of respondents think that the probability is once every 100 years or less, and out of these only 5% think that the probability is once every 350 years or less. Respondents therefore have a tendency to overestimate the probability of an oil spill. These findings are in line with earlier studies on perceived risk (e.g. Rekola & Pouta 2005). Table 11 compares WTP estimates for the first two groups, since the third group was regarded as too small.

For each scenario (Probability and Total risk) Table 10 presents the mean WTP_{MP} for the two groups of respondents with different perceptions of the probability of an oil spill: once in 0–20 years or once in 21–100 years. A Mann-Whitney test was conducted to find out whether the perceived probability affects the stated WTP. No statistically significant differences (within a 95
% confidence interval) were found between the groups. That is, the perceived probability of an oil accident does not seem to have an impact on WTP.

Table 10. Mean WTP\text{MP} (in NOK/year/household) related to perceived probability for an oil spill to occur (once per X years (confidence interval within parentheses).

<table>
<thead>
<tr>
<th>Group</th>
<th>WTP\text{MP} estimates</th>
</tr>
</thead>
<tbody>
<tr>
<td>Perceived probability (once per X years)</td>
<td></td>
</tr>
<tr>
<td>0-20</td>
<td>755.2 (433.3-1067.1)</td>
</tr>
<tr>
<td>21-100</td>
<td>899.2 (584.3-1214.1)</td>
</tr>
</tbody>
</table>

4.3 Costs and benefits of measures decreasing the probability of an oil spill in Lofoten (paper III)

4.3.1 Costs from reducing probability of an oil spill

Based on the assumptions made in section 3.6.1, the estimated total cost to reduce the probability of an oil spill by approximately 30% is NOK 857 to 907 million. In the scenarios in paper II the reduction in probability is on the order of 60% (from .007 to .003) for large spills; i.e., about twice as high as that in Kystverket (2012). But, since the marginal cost for decreased probability is likely to be increasing, it can be assumed that the cost to reduce the probability of an oil spill by 60% is greater than the cost to achieve a 30% reduction based on Kystverket (2012). Besides the uncertainty in the costs related to change in probability, there might also be measures besides the ones presented that are more cost efficient. This would imply that the cost to reach the same goal may be lower.

4.3.2 Benefits from reducing probability of an oil spill

4.3.2.1 Avoided loss of ecosystem service values

The results, presented in Table 11, range between NOK 1.5 billion and 3.3 billion and are based on the assumptions made in chapter 3.6.2.

Table 11. Total net value of ecosystem services using different discount rates and different mean WTP estimates over 15 years. Adjusted to 2012 prices. In millions of NOK.

<table>
<thead>
<tr>
<th>Discount rate</th>
<th>Mean WTP\text{L}</th>
<th>Mean WTP\text{MP}</th>
<th>Mean WTP\text{U}</th>
</tr>
</thead>
<tbody>
<tr>
<td>0%</td>
<td>2,170</td>
<td>2,750</td>
<td>3,310</td>
</tr>
<tr>
<td>6%</td>
<td>1,490</td>
<td>1,890</td>
<td>2,270</td>
</tr>
</tbody>
</table>

4.3.2.2 Expected avoided loss of income

Fisheries

The discounted yearly values are NOK 8.53–12.43 million for cod (based on Faugstad 2010) and NOK 0.67–0.97 million for herring (based on Ibenholt et al. 2010). Summed over 15 years, the total expected avoided loss of income, in terms of cod and herring landings over a 15-year period, is estimated to be between NOK 138 and 201 million.
Tourism

Based on Førland (2013), including impacts on both Lofoten and Nordkapp, the estimated benefit for the tourism sector is between NOK 14 and 20 million. The estimate based on Haugberg (2010) gives an range of NOK 27–39 million.

4.3.2.3 Summary of costs and benefits

The figures presented above show a tendency that the expected benefits of measures to reduce the probability of a major oil spill from shipping in northern Norway far exceed the costs. The costs of measures are estimated at NOK 857–907 million, and the benefits, based on WTP estimates for the three northernmost counties of Norway, are NOK 1.5–3.3 billion. The figures are uncertain, and this is underscored by the effort to specifically study the fishing and tourism sectors. The financial value to the fishing and tourism industries from the reduced risk of oil spills are small but should still not be added to the WTP estimates due to a risk of double counting, since the financial values may be reflected in the WTP estimates. The financial values should instead be seen as a tool to analyse distributional effects between different actors. The expected avoided loss of income used for estimating the value to the fishing industry is NOK 138–201 million. The corresponding value for the tourism industry lies in the range of NOK 14–39 million.

4.4 A generic weighting set (Paper IV)

The updated values are presented in Table 12.


<table>
<thead>
<tr>
<th>Impact category</th>
<th>Weighting: mean value</th>
<th>Weighting: interval</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abiotic resources</td>
<td>0.013/MJ</td>
<td>0.00044-0.027/MJ</td>
</tr>
<tr>
<td>Global warming</td>
<td>0.317/kg CO$_2$-eq</td>
<td>0.011-0.62/kg CO$_2$-eq</td>
</tr>
<tr>
<td>Photochemical oxidation</td>
<td>1.8/kg NMVOC-eq</td>
<td>0.89-2.67/kg NMVOC-eq</td>
</tr>
<tr>
<td>Acidification</td>
<td>3.33/kg SO$_2$-eq</td>
<td>3.33/kg SO$_2$-eq</td>
</tr>
<tr>
<td>Eutrophication, marine</td>
<td>10/kg N</td>
<td>10/kg N</td>
</tr>
<tr>
<td>Eutrophication, fresh water</td>
<td>74.4/kg P</td>
<td>74.4/kg P</td>
</tr>
<tr>
<td>Human toxicity</td>
<td>0.312/kg 1,4 DB-eq</td>
<td>0.0022-0.543/kg 1,4 DB-eq</td>
</tr>
<tr>
<td>Marine water toxicity</td>
<td>1/kg 1,4 DB-eq</td>
<td>1/kg 1,4 DB-eq</td>
</tr>
<tr>
<td>Particles (regional impacts)</td>
<td>30/kg PM10-eq</td>
<td>30/kg PM10-eq</td>
</tr>
</tbody>
</table>

4.5 Environmental costs in LCC (Paper V)

The use of environmental costs (external and internal) in LCC varies among Swedish companies, and the use of external costs in relation to LCC is not common. Environmental costs are most commonly employed at public organisations, or when the customer so requests. External costs were only included in analyses made in one company in the automotive industry, and in that case it is mainly CO$_2$ costs that were used. The reason they state is that there is a risk of the values being internalised in the future and that it is important to show their magnitude. One interviewee
emphasised that the inclusion of environmental costs in an analysis makes it more likely that that cost will have an impact on the final decision. In some cases environmental aspects are considered, e.g. in the form of LCA, but not monetised. But although several respondents emphasised that it is important to express environmental impacts in monetary terms, they were doubtful whether including them in an LCC is the best approach. Environmental costs were nevertheless considered to have an impact on the final decision. In general, the LCC method used varies among organisations and is often company-specific. Different aspects are often excluded, such as different parts of the lifecycle. Its use depends on customer demand. This result also holds for environmental costs: when internal environmental costs are included, not all are necessarily included in the analysis.
5 Discussion

5.1 The ecosystem framework

The concept of ecosystem services is popular in the academic literature and has gained increasing prominence during recent decades. The concept provides a framework where ecological conditions are translated into functions and services that provide benefits to humans. These benefits can be valued using different methods and approaches, and these values can be used in analyses that weight costs and benefits against each other. The results of these analyses can provide input into decision-making regarding the management of ecosystem services and policy making. Policy decisions, in turn, have an effect on the functioning of ecosystem services. It is clear that human activity has an impact on the health of the ecosystems. We are totally dependent on properly functioning ecosystems for our survival. It is important that ecology, economy and policy be linked. This implies good cooperation between, for example, ecologists and economists as they provide economic assessments of policy alternatives that incorporate in-depth ecological knowledge.

This thesis follows the conceptual framework presented in chapter 2 (see also Figure 2). The subsequent papers contribute to different parts of this circle. For two different areas, both concerning water resources, ecosystem services have been identified through literature studies and valued. The values have been included in CBAs and a weighting set has been used to provide a basis for future decision-making. The thesis also presents findings the extent to which environmental costs are included in LCC. The valuation studies look at how policy decisions affect ecosystem services, although a specific discussion of governance perspective on actual policy-making is not included.

Voosen (2013) argues that the two perspectives—that something has an intrinsic value as well as providing benefit to humans—can coexist, but that the ecosystem services concept does not take this into consideration. It is an interesting perspective. The ecosystem service framework is anthropocentric, in the sense that its purpose is to show how ecosystems are important to humans. In valuation theory it is also usually argued that non-use values are included in people’s stated preferences. But applying the ecosystem services concept will at least raise awareness that there are values not included in market prices, and it provides a way of illustrating the benefits that humans derive from nature, as well as our dependence on it. This has not always been the case and is certainly a step forward.

5.2 Valuation and CBA

By providing decision-makers with figures, value is not only given a specific magnitude but is also highlighted in a way that would not be possible without presenting values. The generated value is important, but it is also important to remember that the results of a valuation study or a cost-benefit analysis are just one contribution to the foundations that decision-makers need to soundly weight various options. A CBA is supposed to include the monetary values of all consequences and present the option with the highest NPV. Of course, there may be other values that are not identified in the CBA. Ethical considerations, for example whether or not it is justified to implement the suggested measures or not, are not included in an analysis. Some consequences are identified without being monetarily valued. But integrating ecological conditions in the analysis increases our understanding of how to manage ecosystems. The CBA methodology does not stipulate how to weigh its results against those from other methods. This assessment is left to the decision-maker. CBA is generally used to analyse previously defined policy alternatives. This was partly the case in the two studies. In the TBT study (paper I) the GES’s scenario is a political goal,
while the One-level scenario was constructed to fit ecologically and technically practical conditions. The scenarios in the Lofoten study (paper II) are based on ecological and technical scenarios presented in earlier literature. It is important to describe the benefits that people gain from ecosystem services and their change due to the analysed policies.

Stated preference methods can provide insights into non-market values, but the results are uncertain. There are a number of strategies to overcome or limit these uncertainties, and these were presented in section 3. One example is how to reduce the number of protest bids. Despite these strategies, the results should still be taken with caution. One specific issue is how far in the future effects can be valued. In the weighting set presented in paper IV, values of climate impacts are included. The effects on the climate are long-term and they are highly uncertain. Furthermore, the preferences of future generations are also uncertain. The discount rate can be used to increase or decrease the importance of future changes in the analysis. Paper II raises the issue of whether the impact of risk preferences also influences results when using stated-preference methods.

Despite the uncertainties in CV studies and in CBA, the results provide new inputs into the management of our shared resources. In both papers a CBA shows that the benefits are likely to be higher than the costs of achieving the desired change. This provides a basis for discussing environmental management, and at least indicates that people in general find these issues to be important. Future research on economic assessments of policy scenarios and measures should include ecological knowledge. How to handle long-term changes and thresholds in valuation studies need to be further developed. We need empirical results for values in order to equip decision-makers with generic values that enable better decision-making and also enhance comparisons between methods and results.

Thresholds are important to consider but are usually not included in economic analyses. It is a challenge to include thresholds and other non-marginal changes in economic analyses. Ecological knowledge, however, is a precondition for sound analyses of these thresholds.

5.3 Discussions of the papers

5.3.1 Biodiversity in Swedish coastal waters affected by TBT (Paper I)

The quantitative assessment framework that has been developed and presented in this thesis links ecological knowledge (steps 1 and 2 in the framework) with economical assessment (steps 2 and 3) and technological possibilities. It also starts on the relation between step 5 and step 1, showing the impact of policy on ecosystem services (drivers). This is a very practical example of the links in the ecosystem service framework.

The issue of impacts on biodiversity seems to be of high concern to the Swedish population, as both the response rate and the importance of ecosystem services were high. What have not been included in the study are the impacts of other substances on biota and on humans. When dredging, other substances are likely to be removed, decreasing negative effects. On the other hand, dredging might also release hazardous substances causing negative effects in turn.

The results from the valuation study regarding TBT in Swedish coastal waters not only show a high WTP for the suggested changes. It also presents a framework for the levels and consequences of TBT. In the case of TBT the situation will not be improved for a very long time if measures are not adopted. The framework enables scrutiny of implemented measures and can thus be of help in evaluating remediation methods. The framework also allows a visual representation of current environmental status. A precondition is that ecological data on the different levels of hazardous substances are available and that the effects of those levels are
known. A framework like this also has the ability to show threshold levels (if they are known) indicating when the situation is getting closer to ecological collapse. More research is definitely needed on other substances, as are effects on humans. The framework presents a way of analysing the effects that political decisions can have on ecosystems and the services they provide. It also shows how previously established political ambitions are targeted. In the case of TBT, the study clearly shows that the goal of achieving GES in Swedish coastal waters by 2020 probably is beneficial for society, under the assumptions made in the study, but we might question whether that is the case in practice. The study only considers a number of harbours and no fairways. Pollutants are also likely to exist in other places where boats have been present. Other pollutants might also be removed in such a measure, but the effects of dredging on the spread of hazardous substances are unknown. More research connecting people’s preferences and political/environmental objectives and policy documents is needed. Expanding the analysis on TBT to other water pollutants, the MSFD and the WFD policy documents aiming at achieving good environmental status would be more easily assessed if a multilevel framework existed for additional hazardous substances.

5.3.2 Ecosystem services at risk from oil spills in Lofoten (Paper II)

Ecosystems and the services they provide are rather well investigated in the Arctic, and in particular in northern Norway. There are also a number of policy documents regarding the management of resources and ecosystem services in the area. Paper II contributes by linking a valuation study (step 3) of the values of ecosystem services (step 1 and 2) in the Lofoten area with policy (step 4 and 5). Because the study is an ex ante study, the results yield implications for policy decisions for the region. Two scenarios were used: one reducing total risk or probability and impact, and one reducing only the probability of an oil spill. Although respondents believed that probability was larger than it is, their beliefs on probability did not affect their WTP. Perhaps this is because respondents accepted the information provided in the scenarios. There was no significant difference in mean WTP between the scenarios. Non-linear preferences for small probabilities and large consequences might help explain the results. In this survey the number of protest bids was large—about one third. Efforts were made to reduce the number of protesters, as it was already a concern in the pilot studies. In addition, shares of protest bids did not differ significantly between the scenarios. The interpretation is that people are mainly concerned with preventing oil spills from happening at all, since a small spill might imply that the Lofoten area would lose the pristine state currently associated with the region. Even though some ecosystem services are regarded as more important than others, it was found that respondents have a holistic view of ecosystem services and believe that measures taken should focus on ecosystem services in general rather than on specific ones.

It seems that people want to prevent spills from happening at all. If a spill, of any size, were to occur, the pristine qualities of the area would be lost and that should be avoided. It could be argued that it is difficult to value such small changes in probability, and that respondents cannot differentiate between the different magnitudes of a spill. In this particular case there is also a debate concerning the extraction of fossil fuels that affects the issue. This most likely has an impact on the high protest rates seen. People working close to, and dependent upon ecosystem services for their livelihood, protested more than other categories of respondents. Little research has been done on ecosystem services at risk. This is definitely an area in need of future research. We need both empirical results and methodological developments in the handling of risk in valuation studies. We also need ex ante studies on the effects of oil spills and other pollutants in the Arctic in general.
5.3.3 Costs and benefits from decreasing the probability of an oil spill in Lofoten (Paper III)

The benefits derived are based on the valuation study in paper II (step 3 in the framework) and a literature study, with the aim of finding values directly linked to different actors in the Lofoten area. The CBA summarises the costs and benefits associated with some measures and is part of the framework, specifically step 4. Compared to total benefits, the benefit to industry sectors is small. WTP estimates have been aggregated for the three northernmost counties. The purpose was to be able to make a distributional analysis between different actors in the area. If a larger geographical area was used as the reference point, the total benefit would consequently have been larger.

The costs presented follow the same reasoning. Only costs related to reducing probability in the northernmost counties are presented. The figures found were on measures that also reduced the impact of an oil spill. This implies that the estimates might be higher than actual costs. Even when using different discount rates and different WTP and cost estimates, the benefits derived from the measures were larger than their costs. This gives a positive net present value, and according to the CBA it is recommended that the measures are implemented.

The conclusions concerning the results elicited in the CBA in northern Norway are quite straightforward. Given the assumptions made, it would be beneficial for society to decrease the probability of an oil spill, even if the initial probability is small. Empirical studies on the costs and benefits of different interventions in the Arctic are needed to verify the results and create a flora of references for management decisions.

The Norwegian government has already decided to improve precautionary measures. Limited analyses regarding the costs and benefits preceding that decision have been made. This decision was not based on any valuation of the ecosystem services that would be improved or kept intact. This could perhaps be an example of where the values of ecosystem services have not been valued monetarily but are still a part of decision-makers’ awareness. From this it is perhaps not farfetched to conclude that democracy works, since the importance that people place on ecosystem services as expressed in the valuation study is reflected in politicians’ decisions. It could also be the other way around, that the valuation method and CBA reflect the opinion of the people, and thus work.

5.3.4 An updated weighting set (Paper IV)

The generic weighting set presented in Paper IV is an important contribution to the assessment and understanding of the environmental consequences of choices (step 4 and 5 in the framework). When several environmental aspects are considered, a weighting of the results simplifies the analysis. The decision-maker is provided with support in making trade-offs between different aspects, and larger and smaller impacts can be visualised through the weighting set. This does not mean that decision-makers do not have to make trade-offs between different aspects regardless. Of course, a generic weighting set raises a number of issues as well. Usability is increased at the expense of transparency. Transparency decreases, since there are more steps and calculations that distance analysis from the actual environmental impact, and the decision-maker in most cases has no time to achieve a deeper understanding of these impacts. On the other hand, decision-makers, who might not make an analysis on their own, have access to more information. It is probably better to present generic values than nothing at all, if that is the only other option, as is the case in general for ecosystem values discussed above. By using generic weightings one can say which environmental aspects are the most important in a specific case. It also enables comparisons against other monetised costs and benefits.
The weighting set should be seen as a generic set of values for different environmental issues, such as climate change or air and water pollution, to be used within different environmental systems analysis tools. The updated set has so far been used to weigh LCA results (Paper IV) and climate change in a CBA (Naturvårdsverket 2013).

5.3.5 Environmental costs in LCC (Paper V)

The inclusion of external costs in LCCs was not very common when the study was conducted. At that time, external costs could only be included in public procurement procedures when defining the subject matter of a contract; they could not be used to evaluate procurement bids. The EU Directive (EC 2009) provides for the possibility of including external costs when evaluating tenders in the transport sector. A common obstacle for not using environmental costs in LCC analyses seems to be that there are often internal (LCC) methods used, which is a barrier to the entry of new methods. For public organisations, legislation provides sufficient barriers against including external costs in procurement decisions. If the public sector is unable to or does not have demand to include external costs, there are also no clear incentives for the private sector to do so either. As an interviewee from the automotive industry stated, the only reason is to anticipate future costs or to use such analyses for marketing. Since 2014, a new EU directive has opened up the possibility of including characteristics not related to market prices in public procurement, such as external costs (EU 2014). This could have an impact on the use of external costs in procurement decision-making.

This is also closely related to the discussion of generic weighting sets above. A generic weighting set could improve the possibility of including external costs in decision-making using LCC. But the way that external costs can be incorporated in company-specific decision-making requires further investigation.
6 Conclusions

6.1 The ecosystem service framework

The thesis has presented a number of studies and results in relation to a conceptual ecosystem service framework. The framework includes two valuation studies on ecosystem services; two CBAs comparing costs and benefits from measures affecting the ecosystem services; a generic weighting set where the values of ecosystem services are included and aim at providing decision-makers with knowledge; and finally results from how practitioners include internal and external environmental costs in LCC use. The studies and their results all contribute to rounding out the framework.

The main contributions of this thesis are the new values of ecosystem services (section 6.2 and 6.3) and their use in different tools: CBA (section 6.2 and 6.4) and a weighting set (section 6.5). Furthermore, a quantitative assessment framework was developed for TBT (see section 6.2) including ecological, economic and technical conditions. The framework provides possibilities for decision-makers to more easily follow changes in the environment, and it is recommended that such an approach also be used for other substances. The weighting set has been updated with new values. Its characteristic is its focus on Swedish applications and its use of stated preferences as a basis in most cases. This ensures, theoretically, that non-use values are also included in the estimates. The thesis also presents findings on how environmental costs are included in LCCs used by Swedish companies (section 6.6). One finding is that environmental costs are considered important but are seldom included in LCCs or in decision-making in general. Real-world implementation is regarded as too complicated. The generic weighting set has the possibility to simplify the inclusion of (external) environmental costs in decision-making among Swedish organisations. How this can be done in practice is a question for future research.

The thesis presents all these findings in an ecosystem service framework, providing a pedagogical overview of how everything is interlinked. The approach has been to link ecology, economics and policy. The valuation studies value ecological assets monetarily, and the starting point has been policy goals and recommendations. In paper I this is particularly clear where the framework in itself included all these parts. The values of ecosystem services and natural resources have additionally been related to other economic and financial costs. Valuation results were included in a weighting set with the goal of being included in policymaking and decision-making. Problems with this approach are illustrated in paper V where practitioners describe their experiences.

6.2 The papers

In paper I a quantitative assessment framework was developed linking ecological knowledge, practical feasibility and valuation. The framework was used in a valuation study providing monetary estimates of the changes in biodiversity due to different measures that would mitigate the harm caused by TBT pollution. The results from the valuation study were used in a CBA that compared costs and benefits of different measures to achieve GES, on the one hand, and to improve the environmental status by one level through natural degradation. The study shows that people put significant value on biodiversity in Swedish coastal waters and that the values were likely to exceed the costs for measures.

Paper II investigates how subjective estimates of the probability of an oil spill steers the WTP of a reduced probability. It further analyses the differences in WTP for reduced probability and for a reduction in both probability and consequences from an oil spill causing negative impacts on different ecosystem services, and also analyses respondents’ preferences for different ecosystem
services in the area. The findings show that the subjective judgement of probability does not drive WTP. No significant differences between WTP in the two scenarios have been found. Another finding is that even though people tend to regard some ecosystem services as more important than others, the differences are not large enough to imply that policies should be focused on specific ecosystem services but rather on environmental improvement in general.

The third paper sought to analyse a case study based on a hypothetical future oil spill event in the Lofoten area in northern Norway using CBA. This study compared a) a BAU scenario with b) a policy option implying decreased probability of a marine oil spill accident thanks to certain measures. The study finds that benefits of the analysed measures would exceed the costs and that it is socially beneficial to spend resources on reducing the probability of an oil spill.

Paper IV presents an updated weighting set to be used in environmental assessment tools such as LCC and LCA.

The last paper discusses how environmental costs are considered in LCC. The findings show that practitioners think that environmental aspects are important but are generally not included in LCC tools. External costs are included only in one case. Respondents also doubted whether the best approach is to include environmental costs in LCC.
References


TEEB, 2010. The Economics of Ecosystems and Biodiversity: Mainstreaming the economics of nature: A synthesis of the approach, conclusions and recommendations of TEEB.


