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Application of economic tools for informing Atlantic salmon management in Norway

Opportunity cost of protection measures, valuation of ecosystem services and environmental cost-benefit analysis

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Summary

The research presented in this thesis concerns the application of economic analysis for informing Atlantic salmon management in Norway. The questions investigated here are largely related to the problem of impacts of the salmon aquaculture industry on wild salmon populations. The results have important implications from both scientific and management perspectives.

The first issue examined in the thesis concerns the opportunity cost of wild salmon protection under the new regulations in aquaculture sector – “Traffic light” system (TS). As the TS requires to reduce production of the farmed salmon in areas with a high risk of negative impact on the wild salmon populations, the forgone economic benefit is the main cost of environmental protection in this case. A general model for the opportunity cost in the presence of externalities is proposed and challenges in its estimation, mainly related to the complexity of ecological-economic links, are discussed. The results suggest that the effect of a marginal cut in aquaculture production, given its already large scale, is likely to be negligible, which renders the TS a protection strategy that is both economically and ecologically inefficient. It is argued that an analysis of costs and benefits would reveal potential inefficiency at an early stage.

The second research question refers to the nonmarket valuation of ecosystem services as a tool for pricing environmental effects of public projects and policies. As the current “wild-farmed salmon” controversy has led to increased awareness of problems related to salmon management in the general public, the answers given by respondents in stated preference surveys may be biased, which makes the results less reliable for informing management. The experiment reported in the thesis, indicates that the affective valuation bias is higher for wild salmon than for other environmental goods. The bias is suggested to be associated with both the high awareness and prevalence of nonmarket ecosystem services of wild salmon. This effect should be corrected for in survey design if affect is not regarded as a reflection of preferences. However, the influence of affect should be minimized in the project appraisal process by, for example, using cost-benefit analysis (CBA) framework and encouraging joint representation of the management problem.

The third part of the research turns to the issue of economic analysis application in practice. Examples from Norwegian wild salmon management are used to indicate knowledge gaps that may hinder the use of environmental CBA. The results confirm the previous findings given in this thesis on the importance of the quantification of ecological-economic links, especially with application to damage functions. Identified research gaps are also placed in the context of ethical, political and technical constraints of CBA use. The results emphasize a need for improvements to the existing guidelines on when CBA is required and on how it can be better combined with analyses of nonmarket values and distributional issues.

List of papers

Paper 1

Nikitina, E. (2019). Opportunity cost of environmental conservation in the presence of externalities: Application to the farmed and wild salmon trade-off in Norway. *Environmental and resource economics*, *73*(2), 679-696.

Paper 2

Nikitina, E. (2019). Policy context as a factor of bias in the valuation of environmental goods–a dual-process theories perspective. *Journal of Environmental Planning and Management*, *62*(5), 779-796.

Paper 3

Nikitina, E., Aanesen M. (2020). What knowledge is needed to improve applicability of environmental cost-benefit analysis? Insights from a comparative study of two cases in the Atlantic salmon management in Norway. Manuscript submitted for publication.

Acronyms

CBA	Cost-benefit analysis
DF	Directorate of Fisheries
DFØ	Norwegian Government Agency for Financial Management
EBM	Ecosystem-based management
ES	Ecosystem services
GS	*Gyrodactylus salaris*
MAB	Maximum allowable biomass
MEA	Millennium Ecosystem Assessment
NASCO	North Atlantic Salmon Conservation Organization
NEA	Norwegian Environmental Agency
NFSA	Norwegian Food Safety Authority
OECD	Organization for Economic Cooperation and Development
TS	"Traffic light" system
WEI	Wider economic impacts
WTP	Willingness-to-pay

Part I Introduction

1. Research problem

The loss of biodiversity, depletion of natural resources and deterioration of the environment have been accelerating in the last few decades (Millennium Ecosystem Assessment [MEA], 2005; Ripple et al., 2017). The negative impacts on nature are largely associated with economic activities, and its protection and conservation imply an opportunity cost for the economy. Therefore, environmental-economic trade-offs are important aspects of environmental management. Efficient interventions from regulators in such settings require the application of economic analysis.

The economic discipline has become part of environmental management practice along with the natural sciences (Polasky et al., 2019). Theoretical concepts such as ecosystem services (ES) and natural capital are applied in the research and management of a wide range of ecosystems, including diverse marine environments (Costanza et al., 2017). In practice, the increasing role of economics has been reflected in a wider application of analytical tools such as ES economic valuation, cost-benefit analysis (CBA) and various methods for impacts evaluation to environmental management problems (Organization for Economic Cooperation and Development [OECD], 2018; Tinch et al., 2019).

The scholarly literature developing and improving such tools is abundant and growing (Kube et al., 2018). It has seen a new expansion since the late 1990s, when the concepts of natural capital and ES were first put on the research agenda. Considerable advances have been achieved in the field of ES valuation and environmental CBA since then (Atkinson et al., 2012). However, scholars and practitioners have recently been concerned over a lack of research directly relevant to current real-world management issues (Laurans et al., 2013; Segerson, 2015; Olander et al., 2017). In this respect, the present research seeks to make a contribution which is not only intended to improve the economic toolbox, but which is also relevant for the management of a particular resource.

In this thesis, I study different aspects of the application of economic analysis to the management of wild Atlantic salmon (*Salmo salar*) in Norway. Numerous fjords and fresh water systems in Norway are habitat to approximately one-third of the existing stock of wild Atlantic salmon, which means that the country plays an important role in the protection and conservation of the species. This role is formalized through the country's membership in the North Atlantic Salmon Conservation Organization (NASCO, 1988).

Issues related to Atlantic salmon have long been important in the country due to the charismatic nature of the species and its role in Norwegian economy and culture, which is mostly related to recreational fisheries. In contrast to what is observed in other countries, Atlantic salmon angling in Norway is not only a privilege for wealthy tourists, but an important traditional recreational activity accessible to most inhabitants. It is economically important for many local communities that have developed infrastructure and established businesses around this activity. Moreover, salmon fishing is a traditional subsistence and cultural activity of the indigenous Sami people.

In recent years, as the stock has declined, the need for the conservation and protection of wild Atlantic salmon has become urgent. One of the main threats to wild salmon in Norway is related to the negative effects of aquaculture, such as the spread of sea lice and farmed fish escapees (Forseth et al., 2017). As the government plans to expand the aquaculture sector considerably, the trade-off between wild and farmed salmon has become a subject of heated debate in Norwegian society. An economic analysis of this trade-off is therefore an important input to the sustainable management of wild salmon.

Salmon management in Norway is characterized by high levels of complexity, dynamics and scale. Such a system is difficult to govern (Jentoft & Chuenpagdee, 2013) and it is also difficult to analyze with the existing economic tools. The following aspects related to complexity are worth mentioning. First, the species provides a wide range of ES, where practically all categories (of the MEA, 2005 classification) are represented, and the number of Norwegian people who benefit from these ES is quite large. Second, the prevalence of nonmarket ES poses challenges for economic analysis. Third, the decline of the wild salmon populations has partly been caused by externalities, including aquaculture effects that impact different interest groups, including the aquaculture industry itself. Finally, the complexity of farmed and wild salmon ecological interactions has created significant challenges for both analysts and regulators. These issues are found in the political and social contexts as well. By addressing certain aspects of the economic analysis of wild salmon management, this thesis intends to contribute to a more efficient governance of wild salmon, particularly in relation to aquaculture impacts.

Although the problem of aquaculture-related externalities is not the only one creating an environmental-economic trade-off in the field of wild salmon management, it currently appears to be the most pressing. Thus, all research presented in the thesis is directly or indirectly linked to this problem. Aquaculture-related aspects of salmon management provide rich material for studying application of economic analysis to environmental management due to the abovementioned complexities. Therefore, from an academic perspective, it serves as an illustrative example from which valuable lessons for the discipline can be gained. The thesis therefore represents a case study that contributes to knowledge in the field of environmental economics. Among the number of potential research questions, three issues were chosen for this thesis and they are addressed in the three papers. These issues concern the opportunity costs of wild salmon protection, the valuation of ES related to wild salmon by stated preference methods and the application of CBA for informing salmon management.

The remainder of the Introduction proceeds with description of Atlantic salmon management in Norway where relevant economic aspects are emphasized. This is followed by a contextualization of research questions and a presentation of the research methods used. I conclude with a discussion of the findings and a summary of lessons drawn from this research with regard to salmon management practice and academic contributions.

2. Background

2.1. Atlantic salmon in Norway

With over 400 salmon rivers, Norwegian freshwater systems and fjords are habitat to approximately one-third of the existing stock of Atlantic salmon. In recent years, roughly 500 000 fish have been migrating annually from the sea to the Norwegian rivers to breed (Anon., 2019).

The benefits of wild salmon conservation in Norway are primarily associated with recreational fishing in rivers, which has significant economic value. Approximately 75-85 000 anglers participate in salmon fishing every season. About 20% are tourists from other countries (Stensland et al., 2015). This activity is profitable for the landowners (who own fishing rights) and supports local businesses that provide infrastructure and services related to fishing, such as transport and accommodation.

According to Statistics Norway (www.ssb.no), 82 000 salmon and 35 000 trout were caught in the Norwegian rivers in 2018 where 20 600 and 11 500 respectively were released, and the rest consumed. Despite not constituting a major source of food, wild salmonids are still important from the perspective of provisioning ES.

However, it is cultural ES that make salmon fishing activity attractive. The value of salmon fishing as a recreational activity has been central reason for the development of angling in Norway since the 1830s, when the first British tourists visited Norwegian rivers. For many Norwegians, wild salmon is part of their lifestyle and culture, even if they do not fish themselves. Salmon rivers define the places in which people live and their activities. These values are particularly important for the indigenous people of Norway, the Sami, for whom wild salmon serves as a material base for their culture.

Sea fishing for salmon is still present in Norway, but at a much smaller scale than before. Less than 1 000 individuals are involved in this activity. While the economic benefits are much lower per fish compared to those of river fishing, the activity is still both economically and culturally important for sea fishers (Rybråten et al., 2020).

As to supporting and regulating ES, the research is not abundant, but the species has been shown to play a role in the transfer of nutrients (Jonsson & Jonsson, 2003; Nislow et al., 2004). The role of the species in supporting freshwater and marine water ecosystems is presumably less significant than that of pacific salmonids. Other ES and values can be defined, for example using wild salmon as genetic material in aquaculture, its contribution to education and research, and option values, that is, values related to the possible future use of salmon.

2.2. Management system

The stock of Atlantic salmon in Norway has been declining over the past decades (Anon., 2019). The average number of salmon returning to the rivers for spawning is currently about a half of the average return numbers recorded in the 1980s (Figure 1). Moreover, a recent assessment of 148 populations found 80% of them to not meet the required minimum quality levels in terms of abundance and genetic effects of farmed salmon (Anon., 2017). As stock numbers and quality show negative trend, so does the provision of all ES the stock delivers to people (Limburg & Waldman, 2009), necessitating the implementation of policies and measures that contribute to the species conservation.

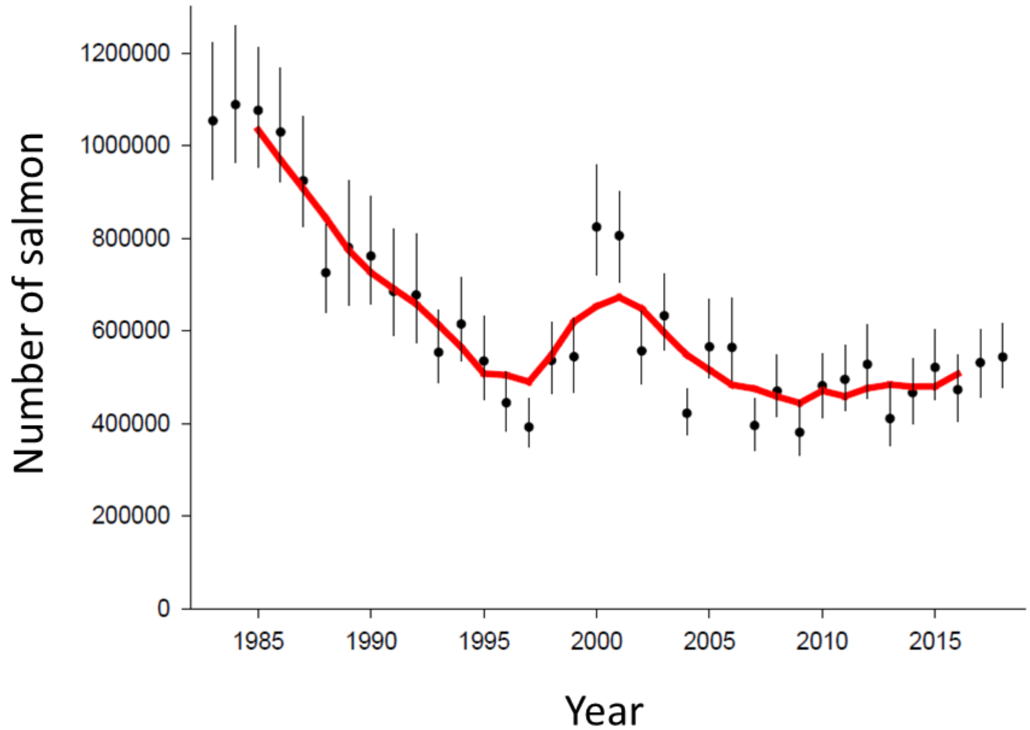

Figure 1. Estimated number of salmon migrating to the Norwegian coast in the period 1983-2018. Dots denote median values, vertical lines illustrate intervals between the lowest and highest values in the simulations, the red curve is the moving average estimated based on 5 years. Source: Anon. 2019.

Conservation measures for Atlantic salmon in Norway address causes of populations' decline, which are numerous and complex. Forseth et al. (2017) classify sea lice and escaped fish from salmon farms as "expanding" population threats that can cause further decline or loss of wild salmon populations. The introduced parasite *Gyrodactylus salaris*, river acidification, hydropower regulation and other habitat invasions are classified as "stabilized" threats that have harmed wild salmon populations at some point, but that are unlikely to cause further damage. Other threats such as pollution, overexploitation and climate change, are indicated as less influential.

The legal basis for salmon conservation is given in an Act on Salmonids and Freshwater Fish (Lakse- og innlandsfiskloven, 1992). Interestingly, the Act establishes the protection of wild fish populations as a first priority, but also aims at providing conditions for their management "in the interest of landowners and recreational fishers." The government agencies most involved in salmon management are the Norwegian Environmental Agency (NEA), Directorate of Fisheries (DF) and Norwegian Food Safety Authority (NFSA).

The NEA's mandate is limited to fresh water systems. It is the central authority for freshwater fish management and makes decisions related to fishing rules, river regulation, river limings, and measures to eradicate salmon parasite *G. salaris*.

The Aquaculture and Coast Management Department of the DF is responsible for management under the Aquaculture Act (Akvakulturloven, 2006). It is involved in the conservation of wild Atlantic salmon where it concerns the impact of aquaculture on the marine environment.

The NFSA's mandate on wild salmon conservation concerns fish health. In cooperation with other agencies, it formulates and implements measures to control fish diseases found in aquaculture and in the natural environment.

The decision-making role of the agencies varies depending on the case. For example, the DF adheres to political objectives related to aquaculture, even if these objectives are questioned within the DF itself. In this case, the DF acts as a stakeholder who may influence policies proposed by the ministries. In individual cases, the agencies have more authority. Decisions about the financing of measures, however, are made by the Ministry of Finance.

The decline of salmon populations and related management actions has impacted a wide range of stakeholders: recreational and commercial fishers, landowners, the aquaculture industry and the public in general. These interest groups are represented in the management system through organizations such as Norwegian Salmon Rivers (Norske Lakseelver), the Norwegian Association of Hunters and Anglers (NJFF), associations of salmon sea fishers, the Norwegian Seafood Federation (Sjømat Norge), as well as municipalities and counties. Environmental organizations are also important stakeholders. Research institutions influence management by providing scientific knowledge and advice.

2.2.1. Aquaculture regulations

The negative impacts of the aquaculture industry on wild salmon populations is a highly debated issue in Norway. Due to the industry's large scale and use of open sea cage technology, these effects are widespread. Today, roughly 800 active fish farming sites along the Norwegian coast produce Atlantic salmon and rainbow trout (www.fiskeridir.no). This corresponds to 3500 sea cages or to 400 million farmed fish at sea at any given time. Notably, the number of wild salmon returning annually for spawning is about half a million, and all of these fish can be placed in 2-3 standard sea cages.

Threats associated with aquaculture include the spread of sea lice and other diseases, genetic impacts due to farmed fish escapement and chemical pollution (Taranger et al., 2014). Sea lice (mainly *Lepeophtheirus salmonis* in Norway) is a salmon parasite commonly present in the marine environment. The number and density of hosts provided by aquaculture creates favorable conditions for the spread of sea lice. When attached to salmon, sea lice feed on their skin and blood, affecting their survival by reducing fitness (Anon., 2012; Torrissen et al., 2013). Sea lice is a major problem for farmed fish producers, with annual costs to the industry estimated at about NOK 5 billion (Iversen et al., 2017). The parasite also affects wild salmon, contributing to the reduction in the number of spawners (Shephard & Gargan, 2017). While a number of methods for combating sea lice are being developed, none appears to provide a definitive solution. Medicinal treatment remains the most commonly used method. The problem with this treatment, however, is that sea lice develop resistance to drugs (Aaen et al., 2015; Fjørtoft et al., 2020).

Despite its environmental impacts, the growth of the salmon farming sector is supported by the government with the ambition for a fivefold increase in production by 2050 (Ministry of Trade, Industry and Fisheries [MTIF], 2014b). Aquaculture development in Norway over the last 50 years has already been unprecedented, however, the aquaculture sector is estimated to have significant potential for further expansion (Olafsen et al., 2012). High profitability, efficiency, innovation and growing demand contribute to this goal (Asche et al., 2018). According to a recent report (Richardsen et al., 2018), the aquaculture industry accounted for an export value of NOK 71 billion in 2018 and contributed NOK 68 billion to GDP including wider economic impacts (WEI). Industries of the aquaculture value chain employ approximately 12 000 people, and these work places are among the most productive in the country in terms of GDP contribution.

Environmental impacts, however, pose a major constraint to further growth, especially with increasing public concern and involvement of stakeholders in communicating the sustainability challenges and associated social impacts (Chu et al., 2010; De Silva, 2012; Olesen et al., 2011; Orstavik, 2017).

The trade-off between industry growth and negative impacts on ecosystems, primarily on wild salmon populations, called for a shift from technical regulations such as sea lice level limits and farm operation standards toward incentive-based policies. The principle of sustainable growth in aquaculture was first placed on the agenda in 2013 with the allocation of "green" aquaculture licenses[1]. A new license allocation process began when 45 licenses were issued, some with a fixed price and some auctioned. The general

[1] Hersoug, 2015 provides a detailed review of the "greening" of Norwegian aquaculture sector.

requirement for all applicants was an obligation to apply a new technological solution or production method that reduces risks of sea lice spread or fish escape.

Beginning with "green" licenses, the environmental regulation of the aquaculture sector became embedded in production growth policy. Shortly after "green" license distribution, a new proposal for production growth was initiated by the government. An increase in maximum allowable biomass (MAB)[2] by 5% at a price of 1.5 million NOK was offered to all producers based on their existing licenses. Firms willing to expand their production would have to ensure lower sea lice levels. The proposed system of sanctions included fines and eventually the withdrawal of the given additional capacity for cases of requirement violation. This proposal, however, has never been fully implemented. In 2014, it was replaced with the proposal of "traffic light" system (TS) as a new strategy for sustainable growth in the sector.

The TS comprises territorial organization of all salmon farms in 13 production areas and managing the production capacity of farms within each area according to environmental indicators which are currently based solely on sea lice levels. Each area is assigned a color code ("traffic light") under regular examinations with the aid of a model predicting the risk of wild population infection. Green, yellow and red colors assigned lead respectively to a 6% increase, no change, or a 6% reduction in MAB within a given area (Figure 2). Additional capacity is offered to producers in green areas partly at a fixed price and partly auctioned. Most of the income (80%) is directed to the recently established Aquaculture Fund that distributes the income between municipalities in which production takes place.

According to a report by the Institute of Marine Research and the Veterinary Institute (Karlsen et al., 2016), which are responsible for the model development, some major knowledge gaps create substantial model uncertainty. Despite these concerns, a white paper on sustainable growth in aquaculture (MTIF, 2014a) introduced the new system, which came into force with a separate act in 2017 (Produksjonsområdeforskriften, 2017). Since the publication of the white paper, several changes have been made to the TS. For example, the possibility of exception from rules in "yellow" and "red" production areas is provided under certain conditions. The new rule implies that those producers who can document low sea lice levels on their farms can be offered growth regardless of the status of their area.

[2] MAB, maximum allowable biomass (maksimalt tillatt biomasse, MTB), is the amount of fish in tons that can be present in sea cages at any given time. A standard aquaculture license is limited to 780 t (945 t in Troms and Finnmark county).

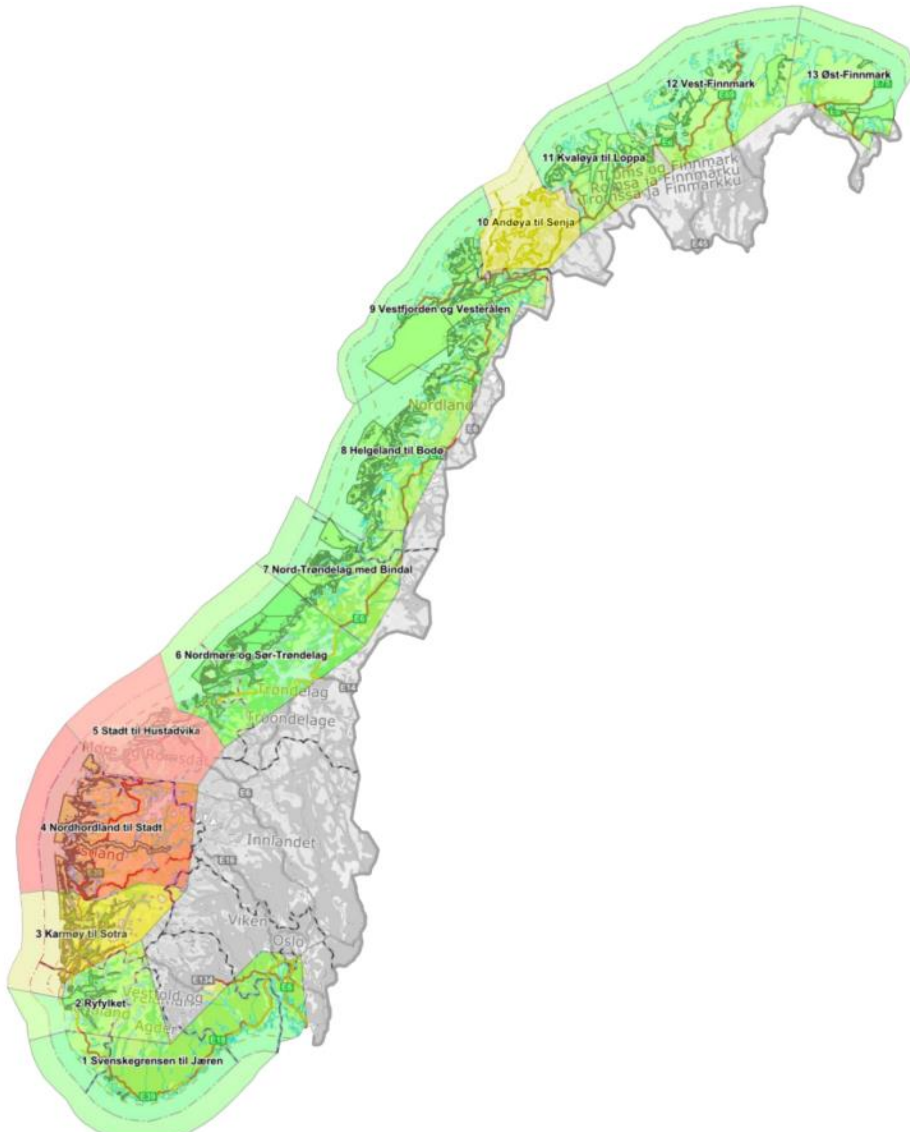

Figure 2. The 13 production areas and their assigned statuses in 2020. Source: regjeringen.no.

The first assessment conducted according to the sea lice indicator and the color-coding of areas took place in 2017. Two areas were designated as red zones, and three were designated as yellow zones. The government decided to hold production at the same level in both groups and to not reduce MAB in the two areas where unacceptable sea lice levels were found. In the eight “green” areas, 6% growth was offered. A 2% capacity increase was allowed to 47 companies based on their existing licenses, resulting in a 7897 tons increase. The price per ton increase was set at NOK 120 000. The remaining growth potential (4%) was introduced via new licenses auctioned and sold for NOK 4 billion in total.

In 2020, two production areas were designated as red zones, two areas were designated as yellow zones and nine were given green status. This time, production capacity in the red areas was to be reduced. Interestingly, while three of the areas improved their status, two others showed negative development in terms of sea lice relative to the previous assessment.

2.3. Research questions

Norwegian salmon management is illustrative for studying the application of economic analysis to environmental issues. Ecological, economic and social complexity yields a number of potential research questions to investigate. In this section, I outline research questions as they emerged in the course of studying the economics of salmon management.

At the start of the “greening” processes in the Norwegian aquaculture, when radical changes to regulations were introduced, the question of efficiency arose. The “green” license initiative prompted a debate on the ecological and economic efficiency of the proposed measures. In a personal conversation, one stakeholder representing anglers’ interests pointed to a lack of evaluation of several measures recently undertaken to regulate the aquaculture sector. Indeed, after “green” license allocation, the new management system, the TS, was introduced without consideration of effects of the previous system and with no evidence of the efficiency of the TS itself.

The problem of evaluation underlies the research questions of the first paper of this thesis, which focuses on the TS. What is striking about this new policy is the possibility (for the first time) to force a reduction in aquaculture biomass to protect wild salmon. However, there was no sound scientific argument for the proposed scale of the reduction. A 6% reduction or increase in production capacity (MAB) has been assumed to have a proportional impact on the environment. Whether this assumption holds, was unknown. In its hearing statement regarding the TS, the NEA expressed doubts that a small reduction in farmed salmon production can lead to a noticeable improvement in wild salmon stocks in the short term and argues for a much larger reduction of MAB in “red” areas (NEA, 2015). However, a lack of knowledge on the relationship between aquaculture biomass and wild salmon makes the premise of such arguments uncertain.

Knowledge gaps have considerable implications for the application of economic analysis of the TS. For example, the benefits and costs of environmental protection under the TS will depend on actual ecological effects of the regulations. If a 6% reduction in biomass leads to no significant effect on wild salmon populations, this will mean that costs of protection are not justified. If, however, the regulations do have an effect, it would be possible to estimate the opportunity cost and consider it in relation to environmental improvements. It would also be possible to compare the TS to other options in terms of cost-efficiency. These issues are examined in Paper 1, which intends to estimate how much is to be given up of benefits as measured in monetary terms to protect one wild salmon under the TS regulations.

The research question investigated in Paper 2 originates from observations of the debate surrounding wild and farmed salmon issues in the Norwegian media and in other arenas of public discourse. There has been increased interest in these issues across the country in recent years with a further polarization of opinions and an increasing prevalence of emotional arguments. As emotions are known to have an influence on economic behavior (e.g., Hanley et al., 2017; Rick & Loewenstein, 2007), the research question that arose then was whether the present grade of emotions would have an impact on the economic choices regarding wild salmon. In Paper 2, these questions are discussed in relation to ES valuation with stated preference methods.

Estimates of ES value should be reliable and accurate enough to provide sound information for decision makers. Stated preference surveys are commonly used for the valuation of environmental benefits that are not traded in the market. As the current “wild-farmed salmon” controversy has led to an increased awareness of the problem in the general public, answers given by respondents in such surveys may be influenced in ways that bias results. In Paper 2, the possibility of affective bias in the valuation of Atlantic salmon is examined in an experiment, where the species is presented for valuation together with other marine species. Based on dual-process theories of judgment (Hsee & Rottenstreich, 2004), the survey is designed to measure the difference between “valuation by feeling” and “valuation by calculation” for each species and to link the size of bias with their characteristics.

The third research question emerged mainly from studying materials related to the TS. It was surprising that with regulations that are so radically new and significant for the whole industry and for other stakeholders, the economic analysis was of such a modest scope. Instructions for public policy appraisal clearly suggest the use of CBA for measures with large impacts (Utredningsinstruksen, 2016). However, the analysis commissioned by the MTIF (Winther et al., 2015) focused solely on impacts on the aquaculture industry and state budget. Environmental effects were not assessed from an economic standpoint despite the fact that these effects were the reason behind the regulation.

On the other hand, CBA has been applied elsewhere to informing environmental measures, including measures for Atlantic salmon protection. For example, the program for the eradication of invasive salmon parasite *G. salaris* was extensively analyzed in a CBA framework (Krokan & Mørkved, 1994; Magnussen, 2011; Andersen et al., 2019). Another example is the liming of rivers to reduce acidification, where the environmental effects were quantified and priced (Navrud, 2001). A natural question is then why economic instruments are applied in the analysis of one measure, and not for the other.

Factors that influence CBA application have not been widely studied. The economic literature focuses mainly on the use of ES valuation as part of CBA, but not on the entire CBA framework (e.g., Laurans et al., 2013). In discussing the application of ES valuation, economists define factors of administrative and technical character, political factors, as well as ethical constraints (e.g., Atkinson et al., 2018; Dick et al., 2017). Some scholars emphasize a lack of scientific knowledge as one limitation (Carmen et al., 2018). In particular, the need for easily applicable economic analytical methods, including nonmarket valuation, is underlined (Guo & Kildow, 2015; Maczka et al., 2016), as well as a lack of relevant ecological data (Daily et al., 2009; de Groot et al., 2010; Drakou et al., 2017; Ruckelshaus et al., 2015). Concerns are expressed regarding disconnection between ecological and social sciences, creating an inability to translate ecological data to economic settings (Collins et al., 2011; Sagoff, 2011). In this respect, challenges in the quantification of ecosystem benefits are central (Olander et al., 2017). The factors facilitating or preventing CBA use in practice are presumably of the same nature. However, as a wider concept, CBA relies on a broader knowledge base that is not limited to ES quantification and pricing.

The issue of knowledge gaps in the use of CBA is the focus of Paper 3. Although based on the previous work (Paper 1 and 2) some ideas on what kinds of knowledge are most important were already present, the ambition was to examine the problem in a systematic way, possibly revealing more research needs. The paper intends to define those knowledge gaps that are most important from the user's perspective. As the paper's research question stems from observing contrasting outcomes of economic analysis application in different cases, Paper 3 is designed as a comparative study. We compare two measures in the wild Atlantic salmon management: the TS and the eradication of *G. salaris*. The paper compares the knowledge base for CBA for the two cases and identifies those knowledge gaps that were more pronounced in the TS case thus partly explaining the absence of CBA in the TS appraisal.

Since knowledge gaps are not the only factors that influence the use of economic analysis in practice, the research addressing knowledge needs must ideally take other factors into account. Therefore, Paper 3 places the knowledge gaps in the context of ethical, political and technical constraints of the CBA use.

Specifying these factors is useful to adjust the research agenda to limitations and opportunities of the management system.

The three papers and their research questions do not represent an exhaustive account of economic analysis application to wild salmon management. However, they can be seen as a building block for scientific knowledge in the field. The research questions discussed here explore important aspects of salmon management and of environmental management in general.

3. Methodological framework

The research questions explored in this project require the application of diverse research methods. While Paper 1 applies a statistical analysis of panel data, Paper 2 is an example of mixed methods research, and Paper 3 is purely qualitative. The dissertation is then written in the mixed methods paradigm, which applies a pragmatic approach and advocates for the use of whatever methodological tools are deemed useful for answering research questions (Teddlie & Tashakkori, 2009). The pragmatic approach allows one to conduct research without regard for the "positivism-constructivism" duality and without dedication to one school of thought, academic tradition, method or technique (Feilzer, 2009).

In embracing flexibility in the choice of methods, the papers presented in this thesis "borrow" extensively from other research fields and apply methods developed elsewhere in the context of the study. For example, in Paper 1, a model of natural capital value (Fenichel & Abbott, 2014) is used as a point of departure to produce a conservation opportunity cost model. In Paper 2, a qualitative analysis is used to supplement experimental data, which are usually analyzed with statistical methods. Paper 3 applies a comparative study method to examine the use of economic analysis in two cases, "borrowing" the method from policy implementation research. In terms of the nonconventional application of research methods, the papers make methodological contributions, which I emphasize in this section. The overview presented here is inevitably general, as there is a trade-off between the scope and depth of methodologic discussion. However, the papers present more details on each method used.

3.1. Natural capital value and opportunity cost of conservation

In Paper 1, the natural capital valuation model developed by Fenichel and Abbott (2014) is applied to produce a general model for the opportunity cost of conservation. While abatement cost models are developed for various types of externalities (Huang et al., 2016), there are currently no established standard models for opportunity costs related to the protection of environmental goods in the economic literature. There is a need for a general model that links the value of market good production to affected environmental condition rather than to pollutant itself. To produce such a model, the value of an environmental good is taken as a point of departure (1). Here, the natural stock condition at time t is denoted s and the economic behavior is denoted as x. To separate the model for benefit estimation from the cost estimation model, I use x_b. The harvest function is then given as $x_b(s)$.

Function $x_b(s)$ describes economic behavior (in this case harvest) which depends on the stock availability. As argued by Fenichel and Abbott (2014), this behavior is unlikely to be optimal, as it is influenced by societal constraints. Thus, the function should be estimated under the current management regime. In addition to market value related to stock exploitation, nonuse values are included in the model. Thus, the total benefit Wacquired from the natural stock is defined as $W(s, x_b(s))$. Within a static framework, we can derive the value of the environmental good as its marginal benefit W_s:

$$p^* = W_s\big(s, x_b(s)\big) = \frac{\partial W}{\partial s} + \frac{\partial W}{\partial x_b}\frac{dx_b}{ds} \quad (1)$$

The value estimated by (1) is the benefit of conservation, which can be compared to costs in the economic assessment of conservation options. Applying the logic of the model to the opportunity cost of conservation, the model is transformed to (2):

$$p^* = W_s(x_c) = \frac{\partial W}{\partial x_c}\frac{1}{ds/dx_c} \quad (2)$$

Here, p^* is the marginal value of a unit of environmental good, W_s is the marginal benefit of the polluters' production, and x_c is the polluters' production as a function of an input factor that causes the degradation of natural stock. A nonoptimal framework is applied to quantify slope ds/dx_c. An empirical estimation of the effect of x_c on stock s is therefore necessary. Expression (2) describes the opportunity cost of natural stock conservation, which can be used on its own to determine the least-cost conservation strategy or in combination with benefits (1) in a CBA of a policy.

3.2. Mixed methods in experimental research

Mixed methods research is defined as "research in which the investigator collects and analyzes data, integrates the findings, and draws inferences using both qualitative and quantitative approaches or methods in a single study or program of inquiry" (Tashakkori & Creswell, 2007, p.4). The data and results in a mixed methods study can be presented in both numerical and narrative form. Despite the apparent increased interest in qualitative and mixed methods (Starr, 2014), their role in economic research remains limited.

One area of mixed methods application in environmental economics involves informing the design and interpretation of results of ES valuation studies. In one study, Hattam et al. (2015) use qualitative and quantitative methods to assess and value the ES of a large marine area. Here, a CE and citizens' jury workshop were set up to price ES. The authors conclude that the use of mixed methods approach affords a better understanding of ES value. In another example, Aanesen et al. (2020) used a focus group discussion that informed a CE study of the valuation of Norwegian coastal ES. The CE included an open-ended question and was followed by interviews that contributed to the interpretation of results. The mixed-method approach proved useful for the elicitation of noncommercial uses of the coastal zone from a management perspective.

While CE is primarily designed for studying preferences, other types of experiments are widely used to study behavioral motivations behind these preferences and the actions of economic agents. Experiments provide information on the behavioral mechanisms that connect incentives, values, and choices, and can therefore contribute to a better understanding of the effects of environmental policy (Ehmke & Shogren, 2009).

A variety of experimental designs have been developed in economics and used for studying environmental behavior (Schram & Ule, 2019). Most behavioral experiments apply statistical methods for structuring studies and for data analysis, while their combination with qualitative data and methods is uncommon. The application of mixed methods can be useful in such experiments, as demonstrated in Paper 2.

Paper 2 describes an experiment conducted in the form of an online survey, where participants were randomly assigned to 5 groups (including one control group). The purpose was to study affective valuation bias. I applied simple statistical analysis to quantify the bias. However, the confirmed difference in the size of bias across groups was not sufficient to claim a cause-effect relationship between the size of bias and the characteristics of the goods, such as awareness about the good and the relative importance of its nonmarket ES. As there is no established theory in economics that could guide the categorization of the goods according to these characteristics, the construction of groups and treatments was based on certain assumptions. To verify these assumptions and strengthen internal validity, additional evidence was required. I used an open-ended question and a multiple choice question as part of the survey to get the participants' own motivation for answering the valuation question. In addition, a quiz was included in the questionnaire to collect information on the awareness about the goods and assess how answers given vary between topics (see Appendix 1). The information obtained from these sources was combined and analyzed qualitatively lending additional support to the hypothesis that differences in the size of bias were indeed driven by variation in the assumed degree of awareness and the prevalence of nonmarket ES.

The paper demonstrates how within a randomized experimental settings, one can enrich a dataset with information that facilitates interpretation of the results. Moreover, the use of qualitative data does not necessarily increase the cognitive load of an experiment. Participants were presented with only one open-ended question, to which they could provide a short answer.

A limitation of the strategy used in Paper 2 concerns the inevitable subjectivity of qualitative analysis and therefore the possibility of confirmation bias. Different interpretations of the same information that may support or contradict the hypothesis can be offered. However, contradictions and differences in interpretation can also be a source of new research questions. The use of qualitative material in any case provides additional insight and improves our understanding of the studied processes.

3.3. Comparative analysis of environmental measures

Comparative research has been applied in political science to study the implementation of policies or, how a policy is put into practice (Hupe & Sætren, 2015). Such studies seek to explain different results of similar policies implementation and to generalize findings in order to develop an implementation theory. Here, cross-national comparisons have been particularly useful, as they present more variability in contexts (Saetren, 2014). An example of a comparative study conducted in the domain of environmental management is a recent analysis exploring ecosystem-based management (EBM) implementation in Norway and Canada (Sander, 2018). The study evaluates elements of an implementation framework

(Winter, 2012) to explain factors responsible for the successful practical application of the EBM concept in one case and failure in the other.

There are similarities between research problems addressed in the field of policy implementation and the issue of economic analysis application in public management. Analogous to many policy concepts such as EBM, economic concepts and analytical tools related to environmental management have not been widely applied in practice despite initial expectations and a remarkable international effort. Thus, in parallel with the policy research literature, discussion is emerging among economic scholars on the extent to which economic tools such as CBA and ES valuation contribute to environmental management (Atkinson et al., 2018; Laurans & Mermet, 2014; Segerson, 2015). Giving the similarity of the research questions in the two fields, comparative studies can explore applications of economic concepts and methods in the same way as they explore policy implementation. However, such studies are not abundant in the economic literature studying the operationalization of economic analysis.

The methodological contribution of Paper 3 lies in its use of focused structured comparison (George, 2005) to identify factors explaining CBA application (or a lack thereof) in environmental management. It is a standard method for conducting multiple case studies in social science, and as noted by George (2005), its principles are straightforward. The method is "structured" in the sense that it examines the same questions in each case. It is "focused" because it is selective in the range of variables examined.

Four types of variables are compared in the paper and are the types of factors that facilitate or hinder CBA use: knowledge gaps, ethical, political and technical constraints. As my main focus is knowledge gaps, this factor is studied in-depth, while the other factors are compared more generally. To study knowledge gaps, a standard CBA framework is used. The cases were compared through each step of the CBA procedure. The research goal here was to define what kinds of knowledge gaps may have reduced the likelihood of CBA application in one of the two studied cases. In the comparison of cases in terms of ethical, political and technical factors of CBA use, no standard framework is found in the literature, but a growing body of work defines groups of factors important for economic toolbox application (e.g., Carmen et al., 2018; Dick et al., 2017). The development of a framework that accounts for interactions between factors, may be a subject for future research.

4. Results and discussion

The results of this research have implications from both scientific and management perspectives. The value of the findings for the economic literature is in identifying and addressing some research needs related to the application of economic analysis to inform environmental management. The following points in particular constitute the main contributions.

First, this research contributes to widely discussed problem of quantification of ecological-economic links, which is an essential element of measuring effects of environmental change. The comparative analysis of two environmental measures provided in Paper 3 highlights the importance of measuring conservation benefits relative to corresponding costs. The cost dimension is largely overlooked in the current literature, which mostly focuses on linking management interventions to ecosystem benefits. Yet, when the main costs associated with environmental conservation and protection objectives are not direct, the cost aspect becomes important, as illustrated by the analysis of the TS. Under TS rules, the main costs of conservation are opportunity costs incurred due to a reduction of aquaculture production in "red" areas. These costs are not easily linked to effects of the TS policy on wild salmon and its ES, which makes the application of analytical tools such as CBA and cost-efficiency analysis problematic. Because this type of environmental-economic links involving externalities, opportunity cost of polluters and conservation benefits has not been widely studied, there is currently no established definition for it in the literature. Although the term "damage function" used in Paper 1 seems appropriate, it is not widely accepted. There is also a lack of modelling frameworks that link polluters' behavior and their costs to the improvement of the affected ES. Such a framework is proposed in Paper 1 based on the natural capital value model developed by Fenichel and Abbott (2014). More research is needed to conceptualize the cost dimension of environmental-economic trade-offs and to develop models (damage functions) for different types of environments and externalities.

Among other research needs identified and to an extent addressed in this thesis is the development of nonmarket valuation techniques. In particular, a contribution is made to the issue of the influence of affective valuation on the results of stated preference surveys. The experiment reported in Paper 2 demonstrates that the average WTP for the same environmental good is significantly higher in groups that performed "valuation by feeling" compared to those performed "valuation by calculation". This difference is explained by the influence of affect. Affect is in turn linked to nonmarket values and awareness about a good. Whether the demonstrated affective valuation is a bias that should be eliminated or part of individual preferences is not a trivial question. The answer to this question defines what kinds of results are used as input for environmental decision making. Despite a potential impact on environmental management, the need to correct for affective bias has not been a focus of the scientific debate. In one of the few publications discussing this question explicitly, Wilson (2008) suggests that there should ideally be a technique that balances affect and reasoning to produce informative values, as both processes are important for making judgments.

In current practice, it is implicitly assumed that stated WTP, even if resulted from affect, should not be corrected unless it is clearly unrealistic or a common anomaly is detected. The literature does not focus specifically on the affect issue in recommendations for conducting WTP surveys according to best practices (e.g., Johnston et al., 2017). There is, however, a discussion of anomalies in the stated preferences method in general (e.g., Hanley & Shogren, 2005; Carlsson, 2010; Poe, 2016). Difference in WTP between the two modes of valuation is definitely an anomaly, as such a result contradicts the assumption of standard economic theory on the stable nature of preferences. Strategies of coping with anomalies vary and do not always include bias correction (Sugden, 2005). Therefore, more discussion regarding this particular form of bias is needed to develop a standard practice for valuation surveys.

The estimation of conservation benefits with stated preference methods and linking such benefits to costs are among the research needs critical for the application of CBA in environmental management. The discussion of the CBA tool in Paper 3 yields some important points for the economic discourse. In particular, alternatives to CBA and a possible use of complimentary tools merit attention. The scholarly literature and management practice have been favoring CBA as the main tool for public project analyses (OECD, 2006; Hanley & Barbier, 2009; U.S. Environmental Protection Agency, 2010; HM Treasury, 2018; Norwegian Government Agency for Financial Management [DFØ], 2018). However, as demonstrated by the study of CBA application in the management of Atlantic salmon (Paper 3), there is a need for a more flexible approach that recognizes the limitations of CBA in capturing distributional effects, wider economic impacts (WEI) and changes in nonmarket values. Although the standard CBA framework includes an assessment of such effects, other tools may also be used to better integrate them into analysis. Multicriteria analysis (Saarikoski et al., 2016), integrated valuation (Jacobs et al., 2018) and other analytical frameworks are proposed in the literature. However, to improve the quality of environmental project appraisal, such tools should be formalized in official guidelines as complimentary or alternative to CBA.

Good guidelines, however, do not improve the quality of decision-making process if they are not followed. Despite the requirement to apply CBA in the appraisal of public projects with significant impacts, the analysis of TS did not consider environmental values and other important effects. Yet, a CBA in accordance to Norwegian guidelines (Utredningsinstruksen, 2016; DFØ, 2018) could reveal ecological and economic inefficiency of the TS at an early stage. The inefficiency of the TS due to the extremely high opportunity cost of conservation is the main finding of this research relevant specifically for the management of Atlantic salmon.

According to results of the panel data analysis given in Paper 1, the marginal effect of a change in aquaculture biomass on the abundance of wild salmon is not significantly different from zero (p = 0.11). This means that a small reduction in biomass in "red" production areas will not necessarily lead to an improvement in wild salmon populations, and therefore the conservation goals of the TS will not be achieved in the short term. Thus, wild salmon conservation under the TS will have an extremely high opportunity cost. On the other hand, a small increase in farmed salmon biomass is unlikely to cause further significant harm to wild salmon. However, if the environmental strategy is targeted on the improvement of the environment rather than preserving its poor condition, such an argument may not be sufficient for approving production growth in the aquaculture sector. Moreover, the form of the damage function beyond the data range is unknown. It may have flat regions demonstrating a buffer effect, but can also have tipping points, where the cumulative effect of aquaculture growth over the years will result in the deterioration of wild salmon condition.

Since a full CBA was not performed, approval of the TS policy was made with incomplete information and based on unverified assumptions. Apart from errors resulting from the omission of relevant facts, decisions made without a systematic assessment of benefits and costs can be prone to heuristics and biases such as affect (Sunstein, 2000, 2017). This is relevant for the management of Atlantic salmon since according to the findings of this research, ethical and political aspects of Atlantic salmon management are influential. Apart from hindering CBA use, these aspects of decision-making environment also increase the emotional load of the debate due to increased awareness of the "wild-farmed salmon" trade-off. The influence of affect goes beyond the valuation issues discussed above, as valuation is only one form of making judgments in general. Taking into account that decision-makers and stakeholders influencing decisions are not immune to heuristics and biases (Hallsworth et al., 2018), their judgments related to the environmental management can be made "by feeling" instead of "by calculation". While it is unclear whether affective valuation should be corrected for in a WTP study, public decision making is typically expected to be rational and not based on intuition and emotions. Although the application of CBA in itself contributes to a rational assessment of information, the dual-process theories used in this thesis suggest that analyzing the issue in a comparative framework with similar issues can also enhance the rationality of judgments made. Joint representation of management problems may be encouraged in guidelines for public project appraisal. For example, such guidelines may require a comparison of effects to those observed in other economic sectors and environments. The use of CBA procedure together with other tools and the joint representation of information can prevent omission bias and increase the rationality of judgments made by officials. However, as noted above, when the procedure is not applied or is applied only partly as in the TS case, it has little value for decision making. The question is then as follows. With good instruments in the economic toolbox, how can we encourage its use in practice?

The results of the comparative study of two measures for protecting wild salmon suggest that although knowledge gaps play a certain role, the use of CBA (and presumably other analytical tools) is also influenced by political, ethical and technical factors. Political and ethical issues were apparently substantial in the TS case and the vagueness of guidelines for the appraisal of public projects has created a technical possibility to omit some important aspects from analysis. Although a CBA is required whenever a project imposes significant costs and benefits, what is meant under "significant" is not clear. One way to increase the use of economic tools where they are needed is to provide detailed and unambiguous requirements and criteria regarding their application. Under the Norwegian guidelines, such improvements are needed in documents defining when and what type of economic analysis is required. Most importantly, the term "significant effects" upon which the thoroughness of analysis depends should be further clarified. Exemplifying the types of effects to be considered in the analysis can also limit the possibility to disregard, for instance, environmental effects and opportunity costs of environmental protection.

Concluding the work on this project, I would like to highlight a point made by Sunstein (2018) in his discussion of CBA, but also relevant to the application of economic analysis in public policy in general. He argues that the CBA approach "has the advantage of forcing officials to ask the right questions in the domains in which it has been on the ascendency, including environmental protection…" He further suggests that "whether or not an analysis of costs and benefits tells us everything we need to know, at least it tells us a great deal that we need to know. We cannot safely proceed without that knowledge." The present project demonstrates that economic analysis with its different tools is useful for asking the right questions, which contribute to more sustainable environmental management. Therefore, it is important to improve the economic toolbox and facilitate its use in practice.

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Part II Papers

Paper 1

"Is this the price of fish, the price of pollution, the price of mitigation, or the price of Mars?"

Anonymous reviewer

Environmental and Resource Economics
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Opportunity Cost of Environmental Conservation in the Presence of Externalities: Application to the Farmed and Wild Salmon Trade-Off in Norway

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Abstract
Estimation of the opportunity cost is necessary for the economic assessment of environmental conservation policies. This paper considers the case where an environmental good is negatively affected in the process of the production of marketable goods. In the presence of externalities, conservation implies the undertaking of abatement measures by polluters. A relevant measure of opportunity cost in these settings is the abatement cost required to preserve or restore a unit of the environmental good in question. Current economic literature lacks an established methodology for deriving such measures, as it commonly focuses on pricing pollutants rather than the natural stocks affected by them. This paper uses a non-optimal valuation approach suggested by Fenichel and Abbott (JAERE 1(1/2):1–27, 2014) as a point of departure and develops a framework for deriving the opportunity cost of conservation as the cost of externality abatement per unit of environmental good. An empirical illustration is provided for the case of the farmed and wild salmon trade-off in Norway in light of the current pollution control regulations.

Keywords Cost–benefit analysis · Damage function · Ecosystem services · Optimality · Policy efficiency · Production frontier · Production function · Shadow price · Valuation

1 Introduction

Environmental protection and conservation policies often involve mitigation of negative effects of economic activities on the natural stock of interest. This challenge is common in the management of systems where the production of marketable goods is happening in close interaction with the natural environment, as in fisheries, aquaculture and agriculture. For example, restoration of the native fish populations and their habitats in many regions around the world requires the reduction of pollution and other negative effects from the aquaculture industry (Naylor et al. 1998; De Silva 2012; Taranger et al. 2014).

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The opportunity cost of conservation in the presence of externalities is the forgone benefit of producers of marketable goods (e.g., aquaculture producers) resulting from their abatement efforts.[1] An opportunity cost estimate is applicable in conservation policy assessment in two ways: either alone in defining the least-cost conservation (abatement) strategy or in combination with the valuation of ecosystem goods and services in the cost–benefit analysis. The former is referred to as cost-effectiveness criterion, which defines whether the expected result could be achieved at lower price (Mickwitz 2003; Nyborg 2014). Here, the valuation of cost is equally important and in some cases is more informative than the valuation of the benefits of protecting the environment.

A relevant measure of conservation cost for a social planner in the described settings is the private abatement costs needed to preserve or restore a unit of the affected natural stock. Thus, the cost per unit reduction of pollutant or harmful effect itself (e.g., the costs needed to reduce pathogens and organic waste emission from aquaculture farms) does not provide sufficient information to a decision-maker for conservation policy appraisal, as the level of emission might not be a perfect substitute to the measure of environmental stock condition.

Despite the relevance for environmental management, abatement cost studies usually price pollutants rather than affected environmental goods. For example, in aquaculture economic literature, the estimations of the costs of reducing parasites and organic waste emissions are common (Costello 2009; Liu and Sumaila 2010). This gap in the literature can be explained by the established theoretical tradition in environmental economics, where the marginal abatement cost is defined as the necessary cost undertaken per additional unit of emissions reduction (McKitrick 1999). From this tradition follows the methodological practice: methods and techniques in the abatement cost literature have been developed for deriving prices of pollutants and, therefore, are not flexible enough to incorporate another level of complexity of ecological-economic relationships that conservation issues involve.

For example, the distance function method that prevails in the literature on the opportunity cost of abatement (Huang et al. 2016; Zhou et al. 2014) assumes the optimal behavior of polluters. It constructs a production frontier that quantifies the relationship between "good" and "bad" outputs (value of marketable goods produced and emission of pollutants, respectively), where the output in the interior of the production frontier is treated as inefficient. This optimized relationship between economic and environmental variables determines the derivation of the opportunity cost of the undesirable output reduction. As noticed in one of the few studies that extended this approach to pricing the affected environmental good, the distance function may ignore the biophysical relationships in the ecological-economic system (Bostian and Herlihy 2014).

In other words, the third level in the relationship "production—pollutant emission—environmental good condition" makes the damage function more complex. While the function explaining the emission level from the production variables is relatively straightforward given the information about production technology, the relationship between environmental good and emission level might involve a range of exogenous environmental factors. Consequently, the assumption on the optimal behavior of the economic agents might no longer hold. Polluters may optimize their emissions in relation to produced output (although, as discussed by Picazo-Tadeo and Prior (2009), the largest producer is not always the largest polluter), but they are less likely to optimize production in relation to the affected environmental good, as

[1] Here, conservation and abatement cost are the opportunity cost: conservation is achieved due to externality abatement, while abatement cost is measured as the forgone net profit of polluters. This is different from the direct costs concept. The direct costs of conservation usually include the costs of conservation activities and management costs. The direct abatement costs are associated with the use of a specific technology (Huang et al. 2016).

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they do not have control over the ecological factors influencing this good. Thus, traditional methods such as distance function result in unrealistic pricing and are not perfectly suitable for estimating the opportunity cost of conservation.

The purpose of this paper is to contribute to the development of a methodological framework for the valuation of conservation opportunity cost as the cost of externality abatement per unit of affected natural stock. This framework should satisfy two requirements. First, it should derive the value per unit of the affected environmental good rather than the pollutant. Second, it should take into account non-optimality in the relationship between the polluters' behavior and the environmental good condition determined by ecological complexity.

The first requirement makes the approach akin to valuation of ecosystem services, as it is the natural stock itself that values are derived for. Therefore, a convenient way of forming the methodology for pricing environmental goods from the cost side is to use models from the ecosystem services valuation literature, where they are priced from the benefits side. Like abatement cost models, valuation models analyze the relationship between environmental condition and the economic activity in order to derive prices, as in the production function valuation approach (Barbier 2007). Therefore, a parallel can be drawn between the two methodologies. A general model of ecosystem services valuation would satisfy the first requirement named above by definition. Considering the second requirement, a non-optimized valuation framework suggested by Fenichel and Abbott (2014) is particularly relevant for the given problem, as it seeks to incorporate a real-world empirically derived relationship between economic activity and the natural stock condition.

This paper uses the natural stock pricing model suggested by Fenichel and Abbott (2014) as a point of departure. Since their model prices ecosystem services, which are the benefits people obtain from ecosystems (Millennium Ecosystem Assessment 2005), it needs to be modified for the purpose of valuation of the opportunity cost people undertake in order to preserve the environment. First, the value of the conservation opportunity cost is associated with the marketable good produced by the polluter and not with the consumption of market and non-market goods provided by the stock itself. The second difference lies in the nature of the relationship between the natural stock (s) and the economic behavior (x). In the benefit valuation model, x is a function that characterizes extraction or other use of the natural resource, and it therefore depends on the condition of this resource. Non-optimality in the relationship $x(s)$ results from the fact that the decisions of the economic agents are influenced by the current management regime (Fenichel and Abbott 2014). In the cost valuation model, natural stock s is a function of the polluters' production input x that causes a harmful effect. The non-optimality of the function $s(x)$ is explained by both ecological and behavioral complexity.

Application of the framework is illustrated with an empirical case of the wild and farmed salmon trade-off in Norway. I use estimates of the wild salmon populations abundance and farmed salmon production data in a panel regression model to quantify the relationship between aquaculture production and wild fish abundance. The stochastic model provides non-optimal empirically estimated coefficients and incorporates environmental variables according to the evidence from ecological studies. The results are discussed in relation to a recently implemented policy that includes regulation of the aquaculture production for the purpose of wild fish stock conservation. The results suggest that the opportunity cost of conservation given the assumptions embedded in the regulatory framework may be infinitely high, which makes the policy inefficient as a conservation strategy. Data limitations, however, prevent drawing this conclusion with certainty.

The paper proceeds with the development of the general methodological framework for valuation of the opportunity cost of conservation as a marginal abatement cost. Section 3

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presents the case of aquaculture externalities in Norway and the empirical application of the proposed methodology. Section 4 describes the data. The econometric treatment and the results are presented in Sect. 5. Section 6 discusses the case study results in relation to the conservation policy efficiency.

2 Valuation of Benefits Versus Opportunity Cost of Natural Stock Conservation

I begin with the general model of valuing the benefits of environmental goods, where the value of a natural stock is associated with the consumption of market and non-market goods through its extraction or other use. Within the context of an exploited fish stock, harvest is the economic activity that generates the value by converting natural stock to a marketable product. Using notations applied in Fenichel and Abbott (2014), I denote the natural stock condition at time t as s and the economic behavior as x. To separate this model for benefits estimation from the cost estimation model, I use x_b in this case. The harvest function is then given as $x_b(s)$. This function describes the economic behavior, in this case harvest, which depends on the availability of the stock. As argued by Fenichel and Abbott (2014), this behavior is unlikely to be optimal, as it is influenced by societal constraints. Thus, the function should be estimated under the current management regime.

In addition to the market value related to the stock exploitation, non-use values are also included in the model. Thus, the total benefit W acquired from the natural stock is defined as $W(s, x_b(s))$. Within a static framework, we can derive the value of the environmental good as its marginal benefit W_s:

$$p^* = W_s(s, x_b(s)) = \frac{\partial W}{\partial s} + \frac{\partial W}{\partial x_b}\frac{dx_b}{ds} \tag{1}$$

The value estimated by (1) is the benefit of conservation, which can be compared with cost in the economic assessment of conservation options.

Now consider a fish stock with its condition s being affected by aquaculture externalities. In this case, conservation implies abatement cost. We define the polluters' behavior x_c as the use of input factor that causes degradation of the natural stock. Consider the polluters' production function with two input factors:

$$Q = Q(x_c, y), \tag{2}$$

where Q is the production output.

Note that the natural stock is not used in the production and, therefore, does not enter the production function. Factor x_c here might be the use of materials that emit harmful by-products, the factor of location or technology. For example, in aquaculture, the main input factor, biomass of the farmed fish, is linked to parasite emissions, number of escapees, organic load, use of various chemicals and other harmful effects (De Silva 2012; Asche and Bjørndal 2011). The same effects, however, can be attributed to the technological factor, as a cleaner technology may reduce the negative impact, keeping the biomass input unchanged. Depending on the abatement strategy (whether the abatement is achieved via reduction of biomass or improving the current technology), a reduction of the factor x_c will influence the polluters' profit in different ways. For example, a reduction of the biomass in aquaculture will mitigate the negative impact on the wild fish but will naturally limit the output, while the replacement of harmful technologies with cleaner ones will require an increase in capital input. The way of controlling polluting factor x_c is directed by the management regime:

command-and-control measures may define directly which factor has to be reduced and how, while market-based measures such as emissions taxes leave this decision to the polluters themselves.

Let W be the net benefit of the polluters' production. The net benefit of producing the amount of goods Q is

$$W = r * Q(x_c, y) - c(x_c, y) \tag{3}$$

where r is the revenue per unit of marketable product and c is the cost of input factors.

The marginal change of the net benefit due to change in x_c is:

$$\frac{\partial W}{\partial x_c} = r * \frac{\partial Q}{\partial x_c} - \frac{\partial c}{\partial x_c} \tag{4}$$

We can rewrite (4) in the following way:

$$\frac{\partial W}{\partial x_c} = \frac{\partial Q}{\partial x_c}\left(r - \frac{\partial c}{\partial Q}\right) \tag{5}$$

Expression (5) shows that the marginal benefit with respect to x_c is equal to the marginal change in output multiplied by the forgone net benefit of producing an additional unit of output. The forgone net benefit due to the reduction of polluting factor x_c will depend on the assumptions about x_c (what factor is assumed to be changing), the rate of substitution between x_c and y and their prices.

Since factor x_c is the one causing negative environmental effects, it links production value W to the condition of the natural stock. The relationship between economic choices x_c related to the use of the polluting input factor and the condition of the affected natural stock s is described by the function $s(x_c)$, $\frac{ds}{dx_c} < 0$. As in the benefit model above, we derive the marginal value per additional unit of natural stock:

$$p^* = W_s(x_c) = \frac{\partial W}{\partial x_c}\frac{dx_c}{ds} \tag{6}$$

The first term of expression (6) is the marginal cost of abatement with respect to x_c, which is derived according to (5). The second term is the slope of the inverse relationship $s(x_c)$. It does not have meaning in this context, because x_c is not a function of s. Externalities are different from resource extraction behavior, as the producer (polluter) does not utilize the natural stock as a factor of production, as shown by (2). Consequently, x_c is treated as an exogenous variable in model (6). Since $\frac{dx_c}{ds} * \frac{ds}{dx_c} = 1$, we can substitute the term $\frac{dx_c}{ds}$ with $\frac{1}{ds/dx_c}$, where the denominator is the slope of the function $s(x_c)$. A non-optimal framework applies to quantification of the slope, as the producers do not have control over the natural factors influencing the stock condition and do not depend on the stock they pollute. An empirical estimation of the effect of x_c on the stock s is therefore necessary in order to derive realistic opportunity cost of conservation, which after transformation takes the form:

$$p^* = W_s(x_c) = \frac{\partial W}{\partial x_c}\frac{1}{ds/dx_c} \tag{7}$$

The expression describes the opportunity cost of the natural stock conservation, which can be used on its own for determining the least-cost conservation strategy or in combination with benefits (1) in a cost–benefit analysis of a policy. It is clear from (7) that if the slope of the damage function is large, the marginal cost of conservation is relatively low, as the improvement in environmental good condition can be achieved by a minor reduction of the polluting factor. In contrast, if the industry does not cause substantial damage to the

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environment, a greater reduction of harmful inputs needs to be undertaken to achieve the conservation target.

Expressions (1) and (7) are the marginal benefits and opportunity cost, respectively, of the conservation of a unit of natural stock. They are derived using the same principle, by looking at the value of economic activity and the relationship between this activity and the natural stock condition. The non-optimality assumption in both cases implies that the relationships between economic and environmental variables should be estimated while taking into account current social and ecological conditions. The key challenge, then, lies in the estimation of the harvest function $x_b(s)$ or the damage function $s(x_c)$.

It is important to highlight the difference in the values included in the benefit and cost model. For example, non-use values of the natural stock and the value generated by fisheries cannot enter the opportunity cost model (7), while the value generated by aquaculture cannot enter the benefit model (1). One can argue, for example, that the externalities from aquaculture affect fisheries, and the function $x_b(s)$ of the resource extraction should be taken into account in the abatement cost estimation. However, fishery production is the activity that exploits benefits from the resource. Reduction of negative effects from fish farms improves conditions for fisheries, increasing the value generated there. The cost of abatement is then cancelled out by the benefits from fisheries. Thus, instead of estimation of cost, we perform a cost–benefit analysis of conservation. For the same reason, non-use values of the natural stock are not included in the cost model.

In the same way, it is often assumed that the benefit of protecting the natural stock can be estimated as the cost of achieving environmental targets (Mäler 1991), and therefore, it should enter the benefit function. As shown by Heal and Kriström (2005), this leads to a similar problem where the benefits and costs are correlated, and instead of a valuation of ecosystem services, a cost–benefit analysis is performed.

Cost and benefit valuation are thus two distinct approaches to conservation policy assessment in the presence of exploitation and polluting economic activities, and the resulting values can be analyzed separately. In particular, the value of opportunity cost is an indicator of the economic efficiency of conservation strategies, as the remainder of the paper demonstrates.

3 Wild and Farmed Atlantic Salmon in Norway

The Norwegian stock of Atlantic salmon includes approximately 440 populations (Anonymous 2016). About half-a-million wild salmon migrate to the Norwegian rivers for spawning annually, supporting recreational fishing activities that have a significant value for anglers (Navrud 2001). In addition, this species represents a high non-use value locally and internationally (Meeren 2013). The conservation of wild Atlantic salmon is highly prioritized and has been widely discussed in Norway in recent years.

In the discussion on conservation of the species, both in scientific and political settings, the negative effects of salmon aquaculture production on the survival of wild salmon is considered one of the key problems.[2] There are approximately 380 million farmed fish in the Norwegian aquaculture at any time according to the Norwegian Directorate of Fisheries (www.fiskeridir.no), which is about 800 times more than the number of wild salmon returning to the rivers annually. The externalities result from the large scale and the technological aspects

[2] A note should be made regarding the scope of the study. We focus our analysis on Atlantic salmon (*Salmo salar*) only and disregard other wild salmonids affected by aquaculture, such as sea trout (*Salmo trutta*) and Arctic charr (*Salvelinus alpinus*). This is due to the data availability and the higher use and non-use values assigned to Atlantic salmon compared to other species.

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of Norwegian open sea cage salmon farming. The biological literature provides evidence on the effect of escapees, sea lice, pollution and disease outbreaks at salmon farms on the survival and quality of the wild salmon populations (Taranger et al. 2014). Spread of sea lice (mainly *Lepeophtheirus salmonis*) is recognized as one of the non-stabilized threats (Forseth et al. 2017) and is given much attention in the context of the recent aquaculture regulation initiatives.

The sea louse is a parasite that is present naturally in the marine environment. Attaching to salmon, it affects its growth, swimming, reproduction and immunity (Costello 2006). The year-round high density of hosts in the open sea cages provides ideal conditions for sea lice. A huge problem for farms, the spread of sea lice also affects wild salmon (Torrissen et al. 2013). The effect of sea lice originating from fish farms on the wild populations of salmon is well-documented (Thorstad and Finstad 2018).

In the presence of negative externalities, the opportunity cost of conservation is associated with the abatement cost of the aquaculture producers. Since Norway is the largest producer of farmed salmon worldwide (Asche and Bjørndal 2011) and at the same time has certain obligations to preserve wild Atlantic salmon through its membership in the North Atlantic Salmon Conservation Organization (NASCO), the regulation of the Norwegian aquaculture is relevant outside the country as well as inside.

A new approach to the regulation of the salmon farming industry in Norway, namely, the "traffic light" system, was presented as part of a strategy for sustainable growth in aquaculture in the white paper Meld. St. nr. 16 (2014–2015) that has the ambition of a fivefold production increase by 2050.

Briefly, the regulations that came in force in 2017, comprise territorial organization of the salmon farms in production areas (13 areas along the coast) and adjustment of the biomass of the salmon farms inside each area according to certain environmental indicators. The indicators are currently based solely on sea lice levels, and the area is assigned a color code ("traffic light") under regular examinations with the aid of a model predicting the risk of infecting wild populations. The assignment of "green" means permission for a 6% increase in the biomass in the production area in the following period. In areas assigned "yellow", the current level of the biomass should not be exceeded. A "red light" means that the risk for wild salmon mortality due to sea lice is too high and all producers within the area will have to reduce the biomass by 6%. Control of the environmental impact via biomass adjustments has the purpose to ensure an acceptable level of risk of a negative effect on wild salmon populations.[3] Therefore, I consider this policy as a conservation measure in the context of this paper, even though the main goal of this management tool was to provide a framework for sustainable growth in the aquaculture sector.

It should be noted that the policy introduces common responsibility for all producers within the area for the environmental impact. This might be a controversial measure and may raise concerns about possible enforcement and compliance challenges. In this respect, assessment of the economic efficiency of the regulations is particularly important.

To estimate the opportunity cost of conservation according to the approach described above, assumptions about economic behavior x_c under the given conservation strategy should be clarified. As follows from the policy design, the negative impacts are only attributed to the biomass in aquaculture production. This is the pollution factor that is regulated by the "traffic light" system in a command-and-control manner. Biomass in production is a key input factor that has a limited possibility for substitution by other factors. Studies on the

[3] What constitutes an acceptable level of risk is a debated issue. In the final proposal on the "traffic-light" system, the red light, which implies a reduction of the biomass in production, is assigned where there is a risk of 30% decline of the wild salmon population.

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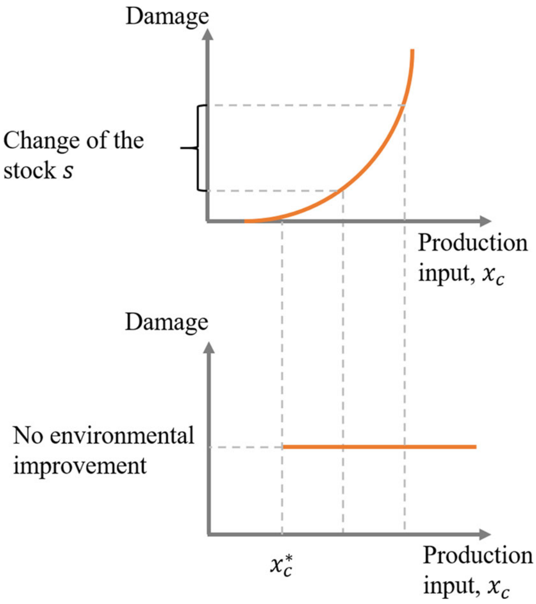

Fig. 1 Different forms of the damage function. The upper graph describes the relationship between the natural stock s and polluting behavior x_c as it is assumed under the "traffic light" system. A continuous damage function implies that an improvement in s can be achieved by a marginal reduction of x_c. The bottom graph illustrates a constant level of damage after some unknown critical level of polluting factor x_c^*

production efficiency of the sector suggest that further significant improvement is unlikely under the present technology (Vassdal and Holst 2011; Asche and Roll 2013; Asche et al. 2013). Thus, with a reduction in the biomass, output Q will be reduced accordingly.

In addition, the "traffic light" system assumes a correlation between the polluting factor (in this case, the biomass in sea cages), the emission of the pollutant (sea lice) and the environmental good condition (wild salmon abundance), which is essentially an assumption about the damage function being continuous (Fig. 1, upper graph). The white paper on sustainable growth in aquaculture (Meld. St. nr. 16 2014–2015: 10) explains it as follows:

"The government seeks to choose an indicator which has a good correlation with the production capacity in a production area. This implies that changes in the biomass at sea cages are associated with the environmental impact in the area, both with increases and reductions of the production capacity."[4]

The slope of the function $s(x_c)$ in (7) under these assumptions is the marginal change of the wild salmon stock per unit change in the aquaculture biomass in the production area. Since substitution for the biomass to any significant extent is unlikely, the marginal abatement cost $\frac{\partial W}{\partial x_c}$ in (5) should be estimated assuming a single-input production function $Q(x_c)$ by multiplying the change of the output by the net benefit of producing a unit of Q. The calculation is straightforward provided price and production data availability.

The quantification of ds/dx_c, on the other hand, is more complex and requires the estimation of an ecological model that links the physical effects of the change in farmed salmon biomass to the abundance of the wild salmon. As emphasized earlier, it should be derived empirically to ensure that current ecological and behavioral factors are taken into account.

I use available ecological and production data in a regression model and analyze the relationship between aquaculture production and salmon populations' abundance in order to derive the slope of the function $s(x_c)$ under the given assumptions. In line with the previous

[4] By capacity, as formulated in the Norwegian policy documents, we mean the maximum allowable biomass of salmon (in kg) in production. If the capacity is fully exploited, it is equal to the biomass.

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ecological literature, I analyze the data on population (river) level and across time, which yields a panel data structure. Hence, the regression model applied to find ds/dx_c takes the general form (the subscript c in x is omitted for clarity):

$$s_{it} = \alpha_i + \beta_0 x_{it} + X'_{it}\beta + \varepsilon_{it}, \quad \varepsilon_{it} \sim IID\left(0, \sigma_\varepsilon^2\right) \tag{8}$$

where s_{it} is the wild salmon abundance in river i in year t, x_{it} is aquaculture biomass that affects population i in the relevant year, and X'_{it} is the matrix of other explanatory variables, that are independent of all ε_{it}. The marginal effect of x_{it} on the abundance s_{it} is represented by the coefficient β_0. The choice of variables and estimation technique are directed primarily by the ecological relationship between wild and farmed salmon and other factors influencing wild salmon abundance.

4 Ecological and Production Data

Figure 2 schematically illustrates the life cycle of a wild salmon cohort and various factors that influence salmon abundance. It begins with smolt migration in year t. At this stage, the smolt is particularly susceptible to sea lice infection (Taranger et al. 2014). Therefore, I use the value of aquaculture biomass in year t as an explanatory variable in the model. Salmon post-smolts spend 1–4 years at sea before they return to the home river for spawning (Jonsson and Jonsson 2011). Depending on the year of return, different year classes are found in each cohort. The number of individuals in each year class returning to the river, before the recreational fishing takes place, is the pre-fishery abundance (PFA) of the year class. The total PFA of the whole cohort is then the sum of PFAs of all year classes.[5]

During the growth at sea, the survival is affected by sea fishing (Anonymous 2016). In the rivers, Atlantic salmon is also harvested, and the survived individuals spawn. Most of these will die after spawning, but there are also individuals that repeat migration. The average size of the population and its dynamics is specific to the home river. The new cohort would normally migrate at the age of 2 or 3 years (Jonsson and Jonsson 2011).

To model the relationship between wild and farmed salmon according to the described ecological processes, we obtained data on wild salmon abundance, standing biomass in aquaculture, and catches in the commercial and recreational fishing of salmon.

The time series of annual PFA estimates were provided by the Norwegian Institute for Nature research (NINA) and the Norwegian Scientific Advisory Committee for Atlantic Salmon Management (VRL). These values represent the sum of the estimated abundances of 3 year classes in each cohort (Fig. 2) and were calculated using methods described in Anonymous (2010) and subsequent reports. According to this description, the PFA estimates are based on the recreational catch data that are corrected for the expected presence of escaped farmed salmon in the catches. The PFA time series were available for the cohorts of salmon from 1992 to 2013 in 154 Norwegian salmon rivers. The total PFA estimates were only available up to 2011, since the return of later cohorts to rivers had not yet been completed (see footnote 5).

In the previous research on the effect of aquaculture externalities on the wild salmon (Otero et al. 2011; Liu and Sumaila 2010), the lack of farm-level production data was identified as

[5] For example, to find the total PFA of the cohort that migrated in 2012, the PFAs of all the returned salmon from this cohort should be estimated. The salmon from this cohort returned to their home rivers in 2013, 2014 and 2015. Since the catch data for 2015 were not yet available, the total PFA could not be calculated for 2012 and later cohorts.

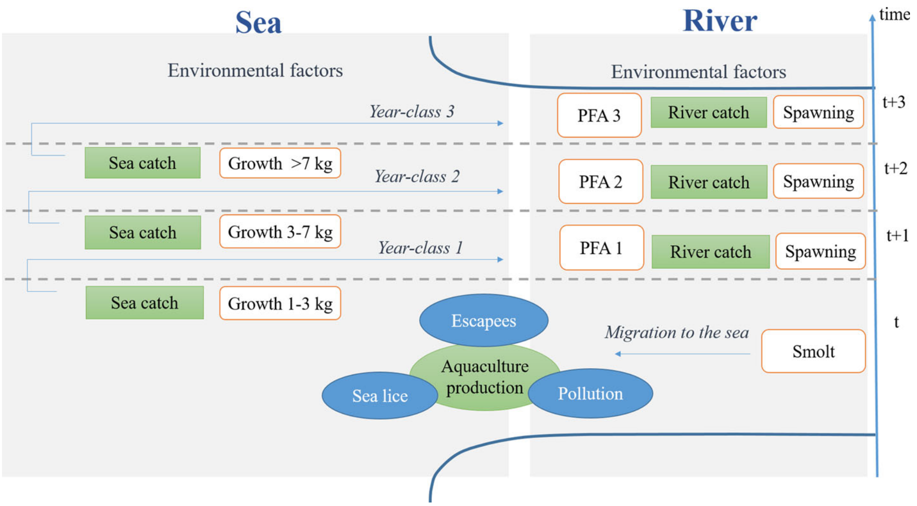

Fig. 2 The life cycle of a wild salmon cohort and factors influencing its abundance. The aquaculture production is assumed to influence the cohort in year t, reducing the number of fish returning to the river in the three following years. The total pre-fishery abundance (PFA) of the whole cohort is the sum of PFAs of all year-classes

an important limitation, making quantification of the damage function challenging. By the time of our study, however, such data became available and allowed a more detailed analysis.

The aquaculture biomass variable x_{it} is formed from the data on monthly reported standing biomass at each farm (in kg farmed fish) provided for this study by the Norwegian Directorate of Fisheries. The time series covered the period from 2005 to 2014. To obtain the values for each river and year, I summed the year average of the standing biomasses of all farms located within 30 km water distance from river outlets.[6] The matrix of water distances between farms and river outlets was provided by VRL. The 30 km distance was chosen following ecological and statistical considerations. The studies on the ecology of sea lice suggest that lice copepodids (infectious stage) spread at a distance of 20–40 km from the source (Asplin et al. 2013). A review by Thorstad et al. (2015) on the effects of lice on sea trout indicates that the probability of infestation is the highest within 30 km from salmon farms; however, this cut-off criterion is uncertain. I compared datasets formed for a range of distances (10–100 km) and found that 28–32 km forms an optimal dataset that provides a sufficiently large number of cross-sectional units for statistical analysis and minimizes the overlap. I proceeded then with the dataset based on a 30 km distance, which included 93 rivers (Fig. 3).

The data on sea catches estimated for each cohort on the regional level provided by VRL are grouped by region, as the sea catches cannot be attributed to a particular population of salmon. Thus, the sea harvest in a particular year has the same value for all rivers located within one of the four regions (East, West, Central and Northern Norway).

Another important variable defining the abundance of a cohort is its initial size. Since the number of migrating smolts is unknown, I use the total river catch time series available at the Statistics Norway database (www.ssb.no) lagged 2 years to account for fishing pressure on the parental stock. The assumption here is that intensive recreational fishing results in less-successful spawning and, therefore, fewer smolts migrating 2 years later.

[6] Water distance is the length of the shortest water route from the river outlet to a salmon farm.

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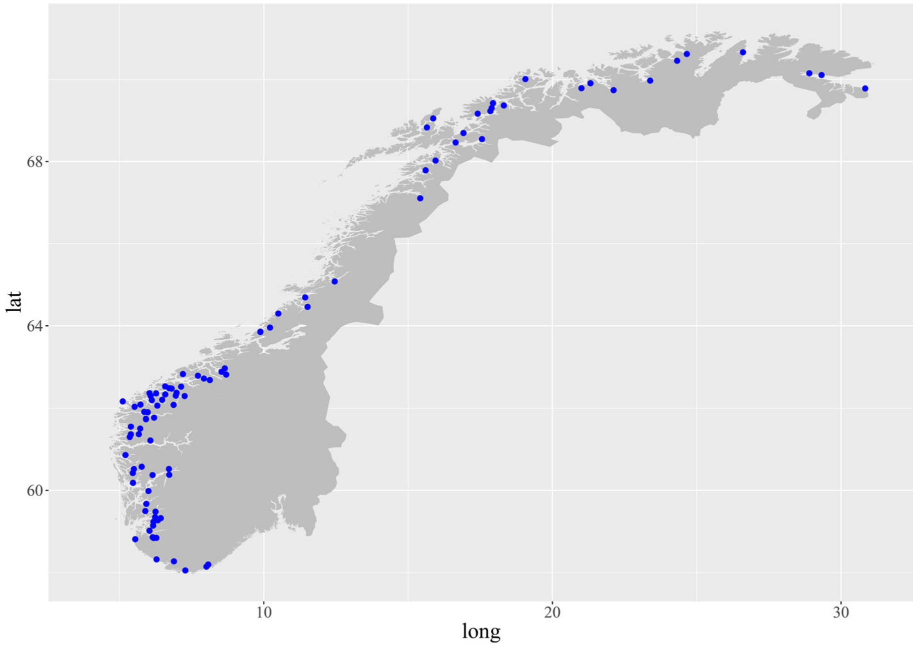

Fig. 3 Locations of the rivers included in the dataset. The dots represent the geographical position of the river outlets

Table 1 Summary of the data

Variable	Units	Minimum value	Maximum value	Mean value	Number of missing values
Total pre-fishery abundance (PFA) of a cohort	Individuals	13	23,540	1167	35
Average standing biomass in aquaculture	kg	0	25,387,188	3,547,187	0
Sea catch in the region	Individuals	766	60,354	18,231	0
River catch	Individuals	0	10,715	495	7

The data represent an unbalanced panel, with the number of cross-sectional units N = 93. The number of years for which the variables are observed varies from 1 to 7 in the period 2005–2011. A summary of the data is presented in Table 1.

5 Opportunity Cost Estimation

Based on the overview of the data and the described ecological relationships, the general model (8) is specified as follows:

$$\ln(s_{it}) = \alpha_i + \beta_0 ln(x_{it}) + \beta_1 \ln(seacatch_{it}) + \beta_2 \ln\left(rivercatch_{i,t-2}\right) + year_{it} + \varepsilon_{it} \quad (9)$$

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where the response variable s_{it} is the total PFA of cohort t in river i. The explanatory variables are the average standing biomass in aquaculture within 30 km from river i in year t, (x_{it}), the total number of individuals of cohort t harvested at sea in region i $(seacatch_{it})$ and the total number of fish harvested in river i in year $(t-2)$, $rivercatch_{i,t-2}$. The intercept α_i varies across locations (rivers), and $year_{it}$ are the year indicator variables.

Since we are interested in the proportional effect of a percentage change in the aquaculture biomass, the log–log function was applied. Taking into account the high relative variability of explanatory variables across locations, the log transformation is also useful for improving the model fit (Gelman and Hill 2007). The functional form used here implies that the coefficient β_0 is the elasticity of the abundance with respect to the aquaculture biomass. The slope of the parameter x_{it} and, therefore, the value of ds/dx_c in (7) is:

$$ds/dx_c = \beta_0 \frac{s_{it}}{x_{it}} \tag{10}$$

As follows from the previous section, the factors influencing PFA can be divided into three groups that define the model specification and econometric technique for parameter estimation. These are river-specific effects, time-variant effects and common shocks. The variables in each group can be observed or unobserved.

Constant environmental characteristics unique to each salmon population and its habitat are summarized under the intercept α_i and are eliminated by the within transformation of data. Therefore, these characteristics are accounted for in the fixed effects panel regression without the need to obtain the data.

Time-variant effects, on the other hand, cannot be omitted unless they are uncorrelated with the variable of interest. Uncorrelated variables should also be included where possible in order to increase precision. Among such variables are catches of salmon at sea and in the rivers and a number of environmental factors influencing wild salmon at different stages of its lifecycle.

As noted by Aas et al. (2010), there is an indefinite range of environmental factors that influence the migration, reproduction and survival of the Atlantic salmon, including water discharge and water chemistry, competition, predation, diseases, climate events, food availability, and density-dependent processes.

Unlike fishing and aquaculture activity, most of the environmental factors are unobservable and, therefore, are omitted from the model. However, all these factors are assumed to be uncorrelated with the aquaculture production. Thus, we do not expect the omission of unobserved variables to cause a substantial bias in the coefficient β_0. We do expect, however, that the error term in this model will be large, since the above-named environmental components might explain most of the variation in the wild salmon PFA.

The aquaculture production variable, measured in terms of standing biomass represents a proxy for all farming-related effects influencing wild salmon PFA (see "Appendix"). Thus, this is not only the variable of interest but also one replacing omitted variables, since reliable data on sea lice levels and escapees are unavailable for all rivers and years.

To account for common environmental events that have an impact on all salmon populations from year to year, I add time dummies to the model. Therefore, a two-way estimation procedure is applied.

It was concluded that a feasible and most relevant approach for estimating the model (9) would be a fixed effects (within) estimator. The procedure allows for individual effects α_i that vary from river to river. The model concentrates on differences within the units, explaining the common change in s_{it} due to change in x_{it}, whether this is a change from one period to

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Table 2 Coefficient estimates of the fixed-effects panel regression model

Variable	Coefficient estimate	Standard error	T	$p > \|t\|$
Average standing biomass in aquaculture, x_{it}	− 0.013	0.008	− 1.60	0.11
Total number of fish harvested at sea, $seacatch_{it}$	− 0.128	0.064	− 1.99	0.05
Total number of fish harvested in river, $rivercatch_{it-2}$	− 0.073	0.028	− 2.59	0.01

The estimates of the log–log model (9) quantify the percentage change of the total pre-fishery abundance (PFA) of wild salmon populations due to percentage change in aquaculture biomass, sea catch and river catch

another or one unit to another (Verbeek 2004). The model was estimated using the R package "plm", following procedures described in Croissant and Millo (2008).[7]

I implement the Lagrange multiplier test (Breusch and Pagan 1980) of individual and time effects based on the results of a model where these effects are not included (ordinary least squares). The p value of the test was close to zero, which indicates that these effects are significant in the data. The test for serial correlation (Wooldridge 2002) rejected the null hypothesis of no correlation in idiosyncratic errors. To produce consistent standard errors, I apply a robust estimator of the covariance matrix of coefficients according to Stock and Watson (2008). Table 2 summarizes the model coefficient estimates for the time-variant effects (β_0, β_1 and β_2) with heteroscedasticity-robust standard errors.

All the coefficients have a negative sign, which suggest that an increase in both fishing and aquaculture production reduces the number of survived individuals in a cohort. In addition to these parameters, intercepts α_i and time effects were estimated. The individual intercepts (fixed effects) estimates were statistically significant. The variation in values of α_i corresponds with the variation in the average stock size in rivers. The time-specific effects were also significant and varied greatly within the 7-year period. Higher coefficients are found for 2009–2011, which corresponds with the common increasing trend in the stock abundance in this period (Anonymous 2016).

As expected, the adjusted R^2 is very low, only 0.03 in this model. This means that the aquaculture effects, harvest, time-specific effects and individual effects together explain only 3% of the variation in the abundance of the wild salmon, at least in the short run (from year to year). A wide range of unobserved environmental effects apparently defines the remaining variation. The model, therefore, cannot be used for the prediction of abundance given specific values of explanatory variables. However, the model is useful for testing the hypothesis about the presence of the effect of aquaculture biomass on the abundance of wild salmon. As seen from Table 2, the coefficient of interest, β_0, is negative as expected but is not significantly different from zero. Thus, the slope of the aquaculture parameter is also equal to zero, as follows from (10). This implies that the opportunity cost of the increase in wild salmon abundance by 1 individual is infinite, according to model (7). As the effect of marginal change in aquaculture biomass on wild salmon abundance is zero, the cost of reduction of farmed fish biomass is undertaken with no environmental improvement in return. It is then infinitely costly to preserve wild population by marginal adjustments of aquaculture production.

[7] The authors also address the issue of methodological differences in the estimation of multiple time series (longitudinal data or mixed-effects models) in economics and in ecological studies.

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Since no significant slope of the damage function was found, calculation of the term $\frac{\partial W}{\partial x_c}$ in (7) becomes irrelevant in this particular example. Non-zero slope would imply positive cost, where the term $\frac{\partial W}{\partial x_c}$ can be calculated according to (3)–(5) given the production function $Q(x_c)$, revenue per unit of marketable product r and the cost of the input factor $c(x_c)$.

6 Discussion

Quantification of the relationship between the polluting factor x_c and the condition of the affected natural stock s is the key element in the estimation of the opportunity cost of conservation in the presence of externalities. In the general model, we assumed a negative slope of the function $s(x_c)$, which implies a continuous damage function (Fig. 1, upper graph). However, in the case of farmed and wild salmon interaction studied in this paper, no significant slope was detected. This is an unexpected result considering the amount of evidence of the negative relationship between the aquaculture production and the wild salmon stock condition, especially due to sea lice.

A plausible explanation is that the complexity of ecological interactions, where different factors can cancel each other out, makes it difficult to isolate the effect of the biomass change. Vollset et al. (2018) discusses this problem in relation to sea lice effect. Taking into account data constraints and knowledge gaps in this field of research (ICES 2016), the failure to capture the effect due to the model uncertainty cannot be ruled out. One of the major limitations of the empirical model is the simplified approach to forming the aquaculture biomass variable based solely on the water distance. Furthermore, as mentioned earlier, a number of ecological variables and their interactions were omitted from the regression model. Another possible source of error is related to the model assumptions applied in the estimation of the wild salmon abundance, such as the population structure and dynamics.

These caveats mentioned, it is nevertheless reasonable to suggest that the absence of a significant effect of the farmed fish biomass change on the wild fish abundance is not a result of model limitations, and the two factors are indeed uncorrelated. This hypothesis does not reject the negative impacts of salmon farming on wild salmon. Rather, it points to another functional form of the relationship. Here, association between the variables is not the same as correlation.

Previous ecological studies estimated the difference in survival and abundance of wild salmonids in the presence and absence of salmon farms (Krkošek et al. 2007; Ford and Myers 2008; Otero et al. 2011). The farming factor in these studies is binary, where exposed populations is shown to be affected compared to unexposed ones. The difference of the empirical exercise in the present paper is that it seeks to estimate the effect of a marginal change in existing aquaculture production, according to the assumptions in the considered management rule, where the farming factor is continuous. While the effect of farming presence explains over 50% decline in many affected wild salmon populations, as estimated by Ford and Myers (2008), and up to 97% in the study by Krkošek et al. (2007), increase or reduction of the biomass in already established farms might not have any significant effect on the wild salmon populations.

This type of association between the variables is consistent with the epidemiology of sea lice, which is considered the main aquaculture-related contributor to the decline of wild salmon. Frazer (2009) explains the host-density effect where any amount of farmed fish above some unknown critical level will cause an exponential growth in lice, and consequently, decline in wild fish populations. Krkošek et al. (2007) also points to this effect, assuming

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that the infestations of wild salmon with sea lice were observed years after the beginning of aquaculture development, when the farmed fish biomass reached a host-density threshold. This might explain the result of the meta-analysis by Vollset et al. (2016) that did not find a significant effect of the estimated level of sea lice exposure from salmon farms on the survival of released smolt. As pointed by the authors, additional salmon lice from fish farms might not affect the released groups. In other words, variation in the sea lice infestation pressure (e.g., as a result of increased biomass at fish farms) over that level might not cause a significant change in wild fish survival.

The threshold hypothesis suggests that the damage function in this case can be piecewise constant (Fig. 1, bottom graph), where negative but constant effect is present when fish farming is established, but there is no damage in the areas without farms. The critical level of the biomass x_c^* is likely to be context-specific and is presumably low, corresponding to the farming intensity on the onset of the aquaculture industry development.

The interpretation of the flat damage function is that the opportunity cost of the wild salmon conservation under the "traffic light" system is infinitely high. This means that the current policy is far from being the least-cost strategy for wild salmon conservation as no significant conservation can be achieved by a small reduction in aquaculture biomass. It is important to stress, however, that the conservation of wild salmon was not the primary goal of the policy, which was intended to provide a predictable growth strategy for the Norwegian salmon aquaculture sector.

The empirical study conducted here is an illustration of the conservation cost estimation approach. Compared to the abatement costs per unit pollutant, the cost estimated in relation to the protected environmental good is a more informative measure for decision-makers and provides better grounds for defining the least-cost conservation strategy, where conservation implies regulation of the polluting industries. It is also a convenient measure for use in a cost–benefit analysis, as the benefits are normally measured per unit of environmental good as well. In the case described in this paper, the cost of pollutant (sea lice) is not an informative measure for the assessment of the implemented regulations. Opportunity cost per unit of environmental good, however, appears to be infinitely high, revealing economic inefficiency of the policy.

An advantage of the proposed approach to conservation cost valuation is that it does not assume optimal polluter behavior. In particular, the marginal value per unit environmental good is defined by the function $s(x_c)$, which is estimated empirically in order to account for both societal factors and ecological complexity. On the other hand, relaxing the optimality assumption poses a major challenge in the practical application of the approach. Ecological-economic systems are characterized by multiple interactions and a number of unknown and unobserved factors. Data requirements and the uncertainty level may, in some cases, be prohibitive. In this respect, the empirical part of this paper, with all the limitations explained above, is not only an illustration of opportunities but also an illustration of the challenges of the proposed approach. A similar problem is discussed by Fenichel and Abbott (2014) in relation to measurement of the natural capital. The present study supports their call for "good biophysical science" in the economic analysis related to environmental management.

Acknowledgements I would like to thank the Norwegian Scientific Advisory Committee for Atlantic Salmon Management for the essential data provided for this research and for sharing their knowledge on salmon ecology. I would like to offer my special thanks to Eva B. Thorstad, Torbjørn Forseth and Peder Fiske. I would also like to thank the Norwegian Directorate of Fisheries for providing the detailed data needed for the analysis.

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Appendix: Standing Biomass in Aquaculture as a Proxy Variable

Assume, for simplicity, that the sea lice level is the only omitted variable. Model (8) then takes the form:

$$s_{it} = \alpha_i + X'_{it}\beta + L_{it}\gamma + \varepsilon_{it}$$

where X'_{it} is a matrix of observed variables (not including aquaculture biomass), and L_{it} is the true, but unobserved, level of sea lice infection pressure. Thus, x_{it} is used to replace L_{it}:

$$s_{it} = \alpha_i + X'_{it}\beta + x_{it}\gamma + \varepsilon_{it}$$

It is required that x_{it} be correlated with L_{it}, so that

$$L_{it} = \delta_0 + \delta x_{it} + v$$

where δ_0 is the intercept, which can be positive or negative, and v is an error due to the indirect relationship between the true variable and the proxy. The parameter δ is assumed to be positive. The positive correlation between production intensity and sea lice has been found in a number of studies (Heuch and Mo 2001; Jansen et al. 2012; Stormoen et al. 2013), making this a valid assumption.

Additional assumptions apply in order to provide consistent estimates of β and γ; see, e.g., Wooldridge (2009):

(1) The error ε_{it} is uncorrelated with x_{it}, which means that the proxy becomes irrelevant in the model if the true variable is included. This is considered a reasonable assumption, because the aquaculture biomass itself has no direct impact on the wild salmon.
(2) The error v is not correlated with X'_{it} and x_{it}. In terms of expected values, this means that the expected average of the omitted true variable only changes with a change in the proxy and not with other regressors. As follows from the description of the data, this assumption also holds.

It can be shown that the same rationale is valid when considering x_{it} as a proxy for the number of escapees and aquaculture-related pollution. A correlation between the aquaculture production and the number of escapees in Norwegian rivers has been found by Fiske et al. (2006). It is reasonable to assume that the emission of pollutants from aquaculture also depend on the production scale. Therefore, x_{it} is a common proxy variable for all the externalities, exact measures for which could not be obtained at the time of analysis. It is important to note, however, that not all of the assumptions might be satisfied by the proxy variable. For example, fishing, which is one of the explanatory variables, might have an association with the number of escapees. Another limitation regarding escapees is that they might affect the abundance in other periods than t.

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Paper 2

"Affect and feelings aroused with regard to some object (whether a consumer good, or a species) can be considered to have economic value."

Anonymous reviewer

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Policy context as a factor of bias in the valuation of environmental goods – a dual-process theories perspective

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Paper 3

"How can the Norwegian government get away with pursuing a CBA in only 25% of decisions, if there is a mandate to perform a CBA whenever the decision is significant? Is there no recourse for disgruntled parties through the justice system or other means?"

Anonymous reviewer

What Knowledge is Needed to Improve Applicability of Environmental Cost-Benefit Analysis? Insights from a Comparative Study of Two Cases in the Atlantic Salmon Management in Norway.

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Abstract

The literature on ecosystem services (ES) identifies knowledge gaps that hinder operationalization of the concept in environmental management, where the challenges of benefits quantification are underlined. A similar discussion is lacking with regard to environmental cost-benefit analysis (ECBA). Yet, as a broader tool its knowledge requirements are not limited to the ES quantification and valuation. This paper seeks to enrich our understanding of the research needs that are important for ECBA's practical applicability. We contribute with a study of two cases in Atlantic salmon management in Norway. The cases are similar in many respects, however, ECBA was applied in only one of them. Using the method of structured, focused comparison, we identify knowledge gaps that may have hindered application of ECBA in the other case. Among several knowledge needs, the study highlights the cost dimension in the widely discussed problem of ecological-economic links quantification. Not only the conservation benefits should be quantified, the respective change in costs should be measured in an ECBA. We interview stakeholders involved in the two cases about their experience with ECBA and discuss the research needs in connection to ethical, political and technical constraints of its practical application.

Key words: aquaculture, ecosystem services, externalities, fishery, valuation

1. Introduction

Cost-benefit analysis (CBA) has "revolutionized" the practice of public policy during the recent 50 years (Sunstein 2018). The principle of weighing costs and benefits of the alternative courses of actions has influenced decisions in an increasing number of areas, from public infrastructure and safety regulation to health and natural environment (Turner 2007, Andersson 2018, Organization for Economic Cooperation and Development [OECD], 2018).

Environmental cost-benefit analysis (ECBA) is the application of CBA to projects or policies that have the deliberate aim of improvement of or actions that somehow affect the natural environment (OECD 2018).

The economic literature has been recently focused on the issue of practical application of ECBA and the related concept of ecosystem services (ES) in environmental management in different countries (e.g., Feuillette et al. 2016, Dick et al. 2017, Atkinson et al. 2018). Among the factors hindering the uptake of these analytical tools, the lack of relevant scientific knowledge is recognized (Carmen et al. 2018). In particular, the need for easily applicable economic methods, including non-market valuation is underlined (Guo and Kildow 2015, Maczka et al. 2016), as well as a lack of relevant ecological data (Daily et al. 2009, de Groot et al. 2010, Ruckelshaus et al. 2015, Drakou et al. 2017). Concerns are expressed about the lack of connection between ecological and social sciences which renders the translation of ecological data to economic settings difficult (Collins et al. 2011, Sagoff 2011). In this respect, challenges in quantification of ecosystem benefits is central (Olander et al. 2017).

When outlining the knowledge gaps, the literature cited above is mainly focused on the ES and their valuation as an essential part of ECBA. However, a discussion is lacking on what knowledge is needed for a successful application of ECBA as a whole. Yet, as a broader concept, ECBA requires knowledge that includes but is not limited to the ES quantification and pricing. The issue of knowledge gaps of (E)CBA is mostly treated in general (Boardman 2011, OECD 2018), and case studies are few.

This paper seeks to contribute to the understanding of the knowledge needs that are most important for ECBA application in practice. We present an analysis of two environmental measures in the wild Atlantic salmon management in Norway. The first measure is the eradication of the invasive salmon parasite *Gyrodactylus salaris* from Norwegian rivers (henceforth GS case). The second measure is the recently introduced regulations of the aquaculture sector called "Traffic light" system (henceforth the TS case). The TS aims to control the negative effect of aquaculture on the wild salmon due to the spread of another parasite, sea lice (*Lepeophtheirus salmonis*). In the GS case, ECBA was applied and considered useful by the decision makers and stakeholders involved in the process. For the TS, assessments only considered consequences for the aquaculture industry, and did not provide economic values for a wider set of benefits and costs, despite the importance of the environmental and social impacts.

Using the method of structured, focused comparison (George 2005), we examine the cases on each step of a standard ECBA procedure to identify the knowledge gaps that potentially hindered application of ECBA in the TS case.

Focusing on the issue of scientific knowledge, we recognize the importance of other factors influencing the extent of ECBA application, such as ethical, political and technical issues. It is important to look at the problem of knowledge gaps in light of these factors and take them into account when forming the research agenda. Semi-structured interviews with officials and stakeholders involved in salmon management were conducted to explore and compare these factors of ECBA use in the two cases. We then discuss how the research can adjust to these external factors in order to address the knowledge gaps more efficiently in terms of ECBA applicability.

The paper begins with some background information on the wild Atlantic salmon management in Norway. Section 3 yields a description of the two cases, the research methods and data used to address the objectives. Section 4 presents the results, and section 5 discusses the findings.

2. Study Background

Norwegian rivers are habitat to approximately one-third of the existing stock of wild Atlantic salmon. The benefits of wild salmon in Norway are associated primarily with recreational fishing which has a significant economic value and contributes to the development of economic activities locally. The stock, however, has been declining over decades. The average number of salmon returning to the rivers for spawning is currently about a half of the average return numbers in the 1980s (Anon. 2019). Moreover, the recent assessment of 148 populations according to the quality norm showed that 80% of these populations did not meet the required minimum quality in terms of abundance and genetic integrity (Anon. 2017). As stock numbers and quality show a negative trend, so does the provision of all ES the stock delivers to people (Limburg and Waldman 2009), necessitating the implementation of policies and measures that contribute to the species conservation.

Conservation measures for Atlantic salmon in Norway address the causes of population decline, which are multiple and complex. Forseth et al. (2017) classify salmon lice and escaped fish from salmon aquaculture as “expanding” population threats, which can cause a critical decline or loss of wild salmon populations. The introduced parasite *G. salaris*, river acidification, hydropower regulation and other habitat invasions are classified as “stabilized” threats, that harmed salmon populations at some point, but are not likely to cause further damage.

The decline of the salmon populations and the related management actions impact a wide range of stakeholders: recreational and commercial fishermen, land owners, the aquaculture industry and the general public. Environmental organizations are also important stakeholders.

The use of CBA in public policy is regulated by the Instructions on Official Studies and Reports (Ministry of Finance 2016) and several related documents, including official guidelines for CBA procedure (Norwegian Government Agency for Financial Management [DFØ] 2018). It is required to present a CBA when the measure is considered significant in terms of benefits and costs. However, the formulation of the significance criteria is quite vague which is perhaps the reason for only 25% of decisions being informed by CBA according to the report by DFØ (2017).

3. Materials and Methods

3.1. The Two Cases

3.1.1. *G. salaris* Eradication Program

G. salaris is an ectoparasite that infects Atlantic salmon, rainbow trout, Arctic char and other salmonids in the fresh water (Sandodden et al. 2018). In Atlantic salmon, infection results in up to 99% mortality of the juveniles, bringing the local population to near extinction within 4-6 years (Johnsen et al. 1999). The parasite, which is not native to the Norwegian rivers, was introduced on several occasions during the development of the Norwegian aquaculture industry in 1970s. Since then, the strain of the parasite was registered in 17 fresh water systems, 50 rivers in total (Hindar et al. 2018).

The work on saving the populations of salmon in the affected rivers started in 1986, when Norwegian Environment Agency (NEA) established a live gene bank. The fish was bred and kept in the live bank to be re-established in its native rivers after the eradication of *G. salaris.*

The most frequently applied method to eliminate the parasite has been the treatment of rivers with rotenone, a plant-based chemical that kills both the parasite and the host (salmon). This method is effective due to the strong host-dependency of the parasite. The chemical, however, is toxic for all gill-breathing organisms (Sandodden et al. 2018). Thus, the treatment may affect other fish and invertebrates in the river. In some rivers, physical barriers were installed to prevent spreading of the parasite, which reduced the use of rotenone. After a river is treated, the local population of salmon is re-established using healthy eggs and fish from the gene bank.

A project of parasite elimination from the river takes several years from the decision to the recovery of the fish population to the pre-infection level. In an analysis of the treatment of the river Driva (Magnussen 2011), a 15-year project period was assumed. During this time, fishing for salmon and other fish species is limited or absent.

The work intensified in 2008 with the introduction of the Action Plan against *G. salaris* for the period 2014-2016. By the end of 2017, 11 of 17 river systems were treated. The measures have proved largely successful, although in some rivers the parasite has re-established.

Costs of rotenone treatment of rivers are covered by the state budget, and thus the Ministry of Finance is the decision maker regarding funding this activity. The annual budget for rotenone treatment of Norwegian salmon rivers has been in the range NOK 60-100 million (NEA, 2014). Realizing the substantial costs of this type of treatment, NEA already in the early 1990s produced the first ECBA for *G. salaris* eradication with the aim to estimate the net benefit of the program.

The first report estimated benefits and costs of the measure for the period 1981-1998 for the whole country (Krokan and Mørkved 1994), and it combined ex-ante and ex-post analyses as some of the rivers were already treated by that time. Following this report, Mørkved and Krokan (2000) conducted an ECBA for the parasite eradication in rivers in Trondheimsfjorden area. In both reports, the authors used benefit transfer using previous studies, where the assumptions were chosen to provide the lowest estimate of the benefits. The recreational fishing value in these studies was estimated using contingent valuation (CV) and travel cost (TC) methods that gave a maximum WTP of the anglers in the range NOK 150-500 per day (in 1992-kroner). Non-use values (defined as existence value, option values and bequest value combined) were assumed to be 200 per household (in 1992-kroner) based on two local studies, Carlsen (1985) and Navrud (1993), that applied CV method. The benefits were found to outweigh the costs in both NEA reports. Distributional effects were discussed qualitatively.

A later ECBA (Magnussen 2011) used the result of a CV survey for the river Driva, where both anglers and non-anglers participated (Almhjell 2003). With a WTP for saving salmon populations amounting to NOK 190 per person, the net benefit of the project for Norway was estimated to be between NOK 300 mln and 1,7 bln depending on the chosen time period, which is larger than the costs of the project.

The recent economic analysis of the GS projects (Andersen et al. 2019) is focused on the local direct and indirect economic effects related to the fishing activity and has not taken into account non-use values of the wild salmon. Nevertheless, also this analysis found that the benefits outweigh the costs.

Although the use of ECBA in the GS case can be considered successful and illustrative, some caveats are worth mentioning. For example, the damage to other living organisms than wild salmon was disregarded under the assumption that the ecosystem will recover shortly after the treatment. A number of limitations in the data and methods were present. However, these considerations were included and explained in the reports following the standard ECBA procedure.

3.1.2. "Traffic Light" System

Sea lice (mainly *Lepeophtheirus salmonis* in Norway) is another salmon parasite. It is commonly present in marine environments. The number and density of hosts in aquaculture creates favorable conditions for the spread of sea lice. Attaching to salmon, sea lice is feeding on its skin and blood, affecting survival by reducing its fitness (Anon. 2012, Torrissen et al. 2013). Sea lice is a major problem for salmon farms, with annual costs to the industry equal to about NOK 5 billion (Iversen et al. 2017). It also affects wild salmon, contributing to the reduction in the number of fish returning to the river to spawn (Shephard and Gargan 2017). There is a number of possible methods to combat sea lice, but none of them appears to provide a definitive solution. Medicinal treatment has been commonly applied, but it leads to the development of drug resistance in sea lice (Aaen et al. 2015).

Despite the sea lice problem and other environmental impacts, growth of the salmon farming industry is a goal of the current government, who announced the ambition of fivefold increase in production by 2050 (Ministry of Trade Industry and Fisheries, MTIF 2014b). However, as the environmental issues have become more urgent, the government started embedding conservation goals in the aquaculture production growth policy. The "Traffic light" system (TS) was a new strategy of sustainable growth in aquaculture suggested in 2014.

The TS proposal comprised territorial organization of all salmon farms in 13 production areas and managing the production capacity of the farms inside the areas according to sea lice levels. Each area is assigned a color code ("traffic light") based on regular examinations with the aid of a model predicting the risk of infecting wild salmon populations. Green, yellow or red color code assigned leads respectively to 6% increase, no change, or 6% reduction in the maximum allowable biomass (MAB)[1] within the area. The additional MAB is offered to producers in the "green" areas partly at

[1] Maximum allowable biomass (MAB) is the amount of fish in tons that can be kept in sea cages at any given time. A standard aquaculture license in Norway is limited to 780 t or 945 t in Troms and Finnmark county.

a fixed price, and partly auctioned. Most of the income, 80%, is directed to the Aquaculture Fund and from there distributed between the municipalities where the production takes place.

According to a report by the Institute of Marine Research and the Veterinary Institute (Karlsen et al. 2016), responsible for the model development, there is substantial uncertainty attached to the model results. Despite these concerns, the white paper on sustainable growth in aquaculture (MTIF 2014a) introduced TS with effect from 2017. There was no full ECBA conducted prior the introduction of the TS. Instead, the economic analysis commissioned by MTIF only analyzed economic impacts on the aquaculture industry, municipal and national budgets (Winther et al. 2015). The analysis compared several alternative strategies for aquaculture growth including the TS and did not take into account environmental costs and benefits. It concluded that comparison between the alternatives is difficult and will depend on the formulation of the measures.

3.2. The ECBA Framework and Identified Knowledge Gaps

The standard ECBA-framework often defines five steps of the analysis, as shown in Figure 1 (OECD 2006, Hanley and Barbier 2009, U.S. Environmental Protection Agency 2010, HM Treasury 2018, DFØ 2018). We assume that the content of an ECBA in general, and the procedures at each of these steps in particular, are well-known. There are some general knowledge gaps that can be identified in the literature on (E)CBA and its elements, including the ES literature. Based on our knowledge of this material we identified on a very general level the issues most commonly specified by economists. For each ECBA-step, these knowledge gaps are indicated in the lower (colored) part of the box in Figure 1, which yields five research questions guiding the analysis.

Comparing the two cases with regard to the knowledge gaps mentioned in Figure 1, we intend to find out which of them potentially plaied a role in the application of ECBA.

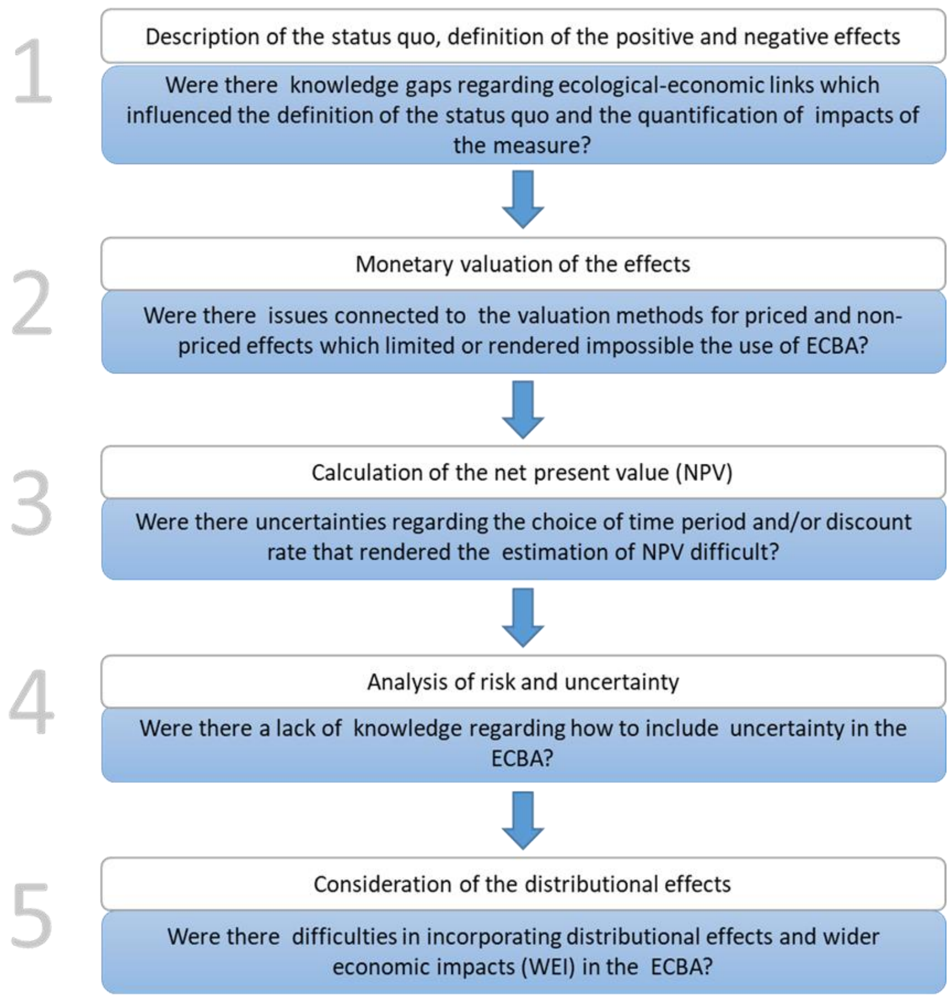

Figure 1. Standard ECBA framework and known knowledge gaps on each step

We apply the method of structured, focused comparison (George 2005), which means that the cases are compared on the same factors (five general types of knowledge gaps) and the comparison is limited to these factors. In the selection of cases, we adopted the Most-Similar-Systems Design (MSSD). This means that the selection is based on cases' similarities rather than differences. The research interest lies in explaining the different results of ECBA application in largely similar circumstances. The two cases are similar with respect to the conservation goals, management system, species of interest, and the type of threat. They are, however, different in terms of scale and time factor, which can be a possible source of bias. The comparison is performed qualitatively by analyzing policy documents related to the two cases (proposals, plans, reports, including scientific evidence) as well as the existing economic analyses of the measures (Krokan and Mørkved 1994, Mørkved and Krokan 2000, Magnussen 2011, Andersen et al. 2019, Winther et al. 2015). We used elements of content and thematic analysis (Bowen 2009). Content analysis comprises identification of categories in the texts which results in a set of codes. In our study, we structured the documents according to the framework presented in Figure 1. Thematic analysis is the process of pattern recognition that results in uncovering themes. In our analysis, these themes are the identified research gaps suggested based on the comparison of structured documents. We compared the knowledge base on each step of ECBA to specify the aspects of knowledge needs that were not satisfied in the TS case, but were fulfilled in the GS case. Based on such comparison we draw conclusions regarding the knowledge gaps that potentially (but not necessarily) hindered the use of ECBA in the TS appraisal.

3.3. Identifying Ethical, Political and Technical Factors

To identify constraints other than knowledge gaps, we collect the data on stakeholder experiences with ECBA in the two cases using semi-structured interviews. We then use these data to compare the two cases in terms of ethical, political and technical factors of ECBA application using the method of structured, focused comparison.

Under ethical constraints, we mean the problem of accepting the ethical grounds of economic approach to environmental issues. In particular, pricing the nature has been controversial since the concept of ES was established (Costanza et al. 2017, Rogers et al. 2015). Political constraints occur, for instance, when ECBA is disregarded because its conclusions conflict the current political agenda (Shapiro 2018, Tinch et al. 2019). Other barriers are of technical nature, as the use of (E)CBA is often problematic due to the lack of financial resources, time restrictions and capacity (Guo and Kildow 2015, Atkinson et al. 2018) or a lack of clear guidelines (Pearce and Seccombe-Hett 2000).

We conducted 12 interviews. Among the participants were representatives of agencies whose mandate concerns salmon management, non-governmental environmental organizations, fishermen interest organizations, local administration representatives, a researcher, and an aquaculture industry representative. We chose informants who are well-known in the field of salmon management and have a long experience and good knowledge of the two cases.

The interviews were semi-structured, where the questions were chosen from the general list (see Appendix), according to the participants' area of expertise.[2] Following the general structure, defined by the research questions, we allowed a certain degree of flexibility, where we discussed related issues that emerged underway. The interviews were recorded with the participants consent, the material was then transcribed and anonymised.[3]

[2] The interviews covered more issues related to salmon management. In this paper, we focus only on the TS and GS cases.

[3] We followed requirements to the processing of personal data by the Norwegian Centre for Research Data: http://www.nsd.uib.no.

The text was analysed in the same way as described in the previous sub-section. We started the content analysis by applying 3 codes summarising ethical, political and technical factors. As we grouped the material under these codes, we added subgroups for specific aspects. We then compared the cases using these groups and subgroups as the base of comparison. As in the analysis of the knowledge gaps, we tried to elicit the barriers and assess their role in the application of ECBA in each case.

4. Results

4.1. Summary of the Knowledge Gaps

The findings related to the knowledge base and possible knowledge gaps are summarized in Table 1. The last column yields information on issues that may have contributed to the lack of an ECBA in the TS case.

Table 1. Summary of the differences in knowledge availability between the two cases on each step of ECBA

ECBA step[4]	GS	TS	Knowledge gaps potentially preventing ECBA use
1	Key ecological-economic links are established and quantified. Main environmental effects are positive. Costs of conservation are direct costs.	Key ecological-economic links are identified but not quantified. Main environmental effects include negative externalities. Costs of conservation are opportunity costs.	Difficult to measure positive and negative effects and the relationship between costs and benefits if ecological-economic links are not quantified, especially when the main environmental effects include negative externalities.
	No feedback effects from the pollution source that can affect the future ecological situation	Feedback effects from the pollution source (the aquaculture industry) that can affect the future ecological situation	Complex dynamics, which reinforces the difficulties describing positive and negative effects
2	Both market and non-market prices are available	Both market and non-market prices are available, but difficulties with non-market prices due to - a lack of updated prices - problems deriving non-market prices for controversial goods and services	Prices on non-market goods and services which are relevant, up-dated and reliable need to be at hand. Special circumstances may make the provision of non-market prices problematic.
3	The measure is one-shot, so NPV for any time horizon can be calculated	The measure is dynamic and is formulated as a decision rule, which makes it difficult to calculate NPV	For more complex measures the NPV is not straightforward

[4] As referred to in Figure 1

4	Uncertainties exist, but can be made tractable by converting them into risk by the use of probability distributions	Uncertainties exist, but as there is little experience with use of this type of measure, including the feedback effects, risk must be assessed qualitatively or semi-qualitatively	When the uncertainties connected to the outfall of the analysis (NPV) cannot be quantified an ECBA will be of less use for decision makers
5	Distributional issues exist, but are not crucial	Distributional issues exist and are crucial both at local, regional and national level	When crucial effects of the measure cannot be quantified an ECBA will be of less use for decision makers

While in the application of ES concept the cause-effect relationship between an intervention and the environmental value is important in terms of quantification of conservation benefits, in ECBA, this relationship is crucial for determining the benefits in relation to costs. The cost dimension is illustrated by the comparison of the two cases on the first step of the ECBA procedure (Table 1). A key difference between the two cases is in the quantification of the main ecological-economic interactions. In the GS case, the impact of the parasite is easily quantified (almost extinction of the wild salmon), and this knowledge is certain. The effect of treatment is also known (full recovery to the pre-infection abundance). Thus, the benefits, mainly related to recreational fishing, could be estimated without major challenges. Moreover, as the main costs of the measure are direct costs, the relationship between benefits and costs is easily measured, making it possible to set up a CBA.

In the TS case, scientific evidence is lacking to allow quantification of the relationship between the biomass in aquaculture and the lice-induced mortality of the wild salmon. Although it is well-known that sea lice from aquaculture farms has a negative effect on wild salmon populations (Finstad et al. 2010, Anon. 2012, Vollset et al. 2017), the effect of 6% reduction or increase of the biomass on the abundance is not yet quantifiable. The main costs of conservation under TS are opportunity cost in terms of forgone benefit of aquaculture producers in “red” areas. The unquantified damage function, that is the function defining the relationship between the change in farmed salmon biomass and wild salmon abundance, renders impossible estimation of costs and benefits relative to each other, which is essential for conducting a CBA.

The fact that the TS considers environmental damage, while GS is about environmental improvement (damage to other species due to rotenone treatment was considered less important and reversible), points to another aspect of knowledge gaps. As the aquaculture industry is the source of increased parasite concentration and can impact the number of lice which again defines the regulation of the biomass under the TS, the polluters' behavior adds another uncertainty to the quantification of the TS effects. Such a feedback effect is absent in the GS case, where e.g. recreational fishers have no influence on the parasite.

Regarding the valuation stage of ECBA, measuring market effects did not represent a major challenge in either of the cases. As to non-market valuation, availability of previous studies facilitated the ECBA in the GS case. Although somewhat outdated, these data could also have been applied in the TS case. Providing new WTP estimates for wild salmon at the time of the introduction of the TS could, however, prove difficult. At that time, awareness about wild salmon and the negative effect of aquaculture was (and still is) a controversial issue in Norway. The debate intensified even more with the introduction of the TS. Such public attention to wild salmon conservation was not present at the time of the first GS analysis. High public awareness and a heated debate can complicate stated preferences studies, which is the only valuation technique providing non-use values for the ecosystem services, by introducing potential bias. In particular, we might expect more protest bids, strategic valuation and affective valuation.

When it comes to the net present value (NPV), the dynamic formulation of the TS measure complicates its calculation. The TS, when implemented will have an almost infinite number of combinations of areas, which have to increase, keep unchanged, or reduce the aquaculture biomass. When, in addition, there are knowledge gaps concerning how to translate ecological effects into economic variables, and effects of reducing or increasing the biomass of farmed salmon are not equally distributed over each of the production areas, it is easy to see how the task of calculating the NPV becomes almost insurmountable.

Treatment of uncertainty (step 4 in Table 1) is also more difficult in the TS case. The uncertainty attached to the implementation of the two measures is of different character. While the uncertainty of effects in the GS case concerned how long it would take until salmon would return to the river and in what quantities, in the TS case the uncertainty rather concerned whether reducing the number of farmed salmon (by 6%) would actually affect the quantity of wild salmon returning to the nearby river. This means that in the GS case the uncertainty could be reduced to risk by the use of a probability distribution. The TS case is different as there is no scientific evidence that a small change in aquaculture biomass will affect wild salmon stocks. Hence, semi-qualitative or purely qualitative methods should be applied, discussing how likely it may be that small reductions in biomass will affect wild salmon stocks. While the former is doable within a traditional ECBA, the latter demands methods that is beyond the ECBA framework (like the development of knowledge enhanced risk matrices, see e.g., Aven, 2017).

Regarding the last step of ECBA, in both cases, relevant studies of wider economic impacts (WEI) were available. Although more specific studies were desirable, the analysis could be done with the present information. Incorporation of distributional effects, on the contrary, represented a challenge in the TS case. Unlike the GS eradication program, where all stakeholders benefited from the measures in the long run (in the short run some would lose while other would be unaffected), the implementation of TS produces winners and losers. Distributional issues are present at various levels in the TS case. First, the measure leads to redistribution of production and thus income generation between producers located in red, yellow and green production areas. Second, it leads to a redistribution of income opportunities from aquaculture producers to land owners along salmon rivers selling fishing rights. If such distributional issues are considerable, and surpasses the monetized effects, it is problematic to apply an ECBA because the most important effects must be described outside of the main analysis.

4.2. Ethical, Political and Technical Factors

Analysis of the interviews demonstrates that ethical, political and technical constraints are more pronounced in the TS case compared to the GS case.

Expansion of the aquaculture industry is approved despite the threats it poses for the environment. Moreover, the damages are imposed on a charismatic species with a high cultural value. Therefore, using an ECBA that involves an explicit weighing of costs and benefits on a monetary scale is resisted on ethical grounds. Commodification of the environment by attaching prices to it and comparing its values with the value of market goods and services, causes concern among stakeholders, especially environmentally-oriented ones. One of the interviewees representing an environmental NGO summarizes widespread views on the issue of pricing the environment (hereinafter, our translation from Norwegian):

"As long as it is only about making values visible, we are positive. However, when one puts a price on something, this implicitly means that it can actually be bought. There are many examples where it [non-market valuation] did not work as intended. One can, for example, see that it becomes easier to get permission for building, if only you compensate [for the environmental damage], buy yourself out".

As seen from the statement above, the ethical concerns occur when environmental damage is involved. In this respect, the ethical barrier may be more influential in the TS case. Many respondents are positive to the economic approach and methods and find them useful in decision-making. However, even admitting that economic efficiency should be considered, most of the interviewed stakeholders were not willing to use economic arguments in the discussion of the environmental damages, because they "do not want this damage whatever the price [of compensation] is". In the GS project, which is about saving wild salmon rather than damaging it, the ethical concerns regarding commodification seem to be less obvious.

Political constraints in the TS case exist due to the presence of distributional effects, mainly the "wild-farmed salmon" controversy. The decisions under the TS will inevitably be in favor of one or another side. If the growth is allowed, the decision is in favor of the aquaculture industry. If the production capacity is reduced, this decision would benefit wild salmon interests. Thus, a certain political message is embedded in these decisions, and it becomes even clearer with the use of ECBA. Although ECBA has not been implemented for the TS, there is a common belief among the stakeholders that NPV of a growth scenario would indeed be positive, and therefore it would justify the environmental damage. Applying ECBA will then lead to the support of aquaculture industry. Many of the interview answers included statements like this one, made by a representative of a fishermen organization:

"Of course, there can be a choice [between aquaculture expansion and wild salmon], if you will put a number on that… A number on wild salmon against the value of aquaculture industry. This number will always be low, I think, unless the non-use values are extremely high. But the others [aquaculture producers] can point to the pure kroner anyway."

This opinion was also shared by the representative of the aquaculture industry, who points out that it is not easy to express the economic argument publicly when this argument is in favor of aquaculture growth. In his view, "one should be very brave as politician to support the aquaculture industry" as one will be met with a strong critique from the environmental organizations and fishermen." It is important to mention in this context, that the critique also applies to researchers, including economists whose conclusions are in favor of one of the stakeholder groups.

Such political risks are not present in the GS case. Here, the authorities and politicians were willing to use ECBA to justify the expenditures. The use of ECBA did not involve a controversial political message due to the absence of major distributional effects.

In addition to ethical and political constraints in ECBA application, there might also be technical barriers, such as time and budget limitations and weaknesses in the organization of decision-making process. The latter was found to be a greater constraint in the TS case.

As the instructions on CBA application are often vague, there seems to be much flexibility in the extent and quality of (E)CBA application by regulatory bodies. According to a representative in the NEA, there is no routines in place in their department for when and how ECBA and other analytical tools such as ES valuation should be applied. Talking about ES valuation, for example, he noticed: "We have tried different approaches… willingness-to-pay... and have done several studies, but not in a systematic way." Hence, there is a technical possibility to omit ECBA or make it only partially. This is well-demonstrated in the TS case. These new regulations definitely imposed huge costs and benefits for large groups. In addition, there is obviously much controversy and resistance to the current plans for aquaculture expansion in Norway. These two criteria make the TS a significant measure which requires a comprehensive analysis than includes ECBA. What we see in practice, is an analysis of the consequences for only one group of stakeholders. The failure to consider all effects, including environmental change, was noticed by stakeholders. Some of them expressed the concern about incomplete appraisal in the hearing process. However, this had no consequences for the regulator, as there is no formal requirement for quality control of the decision-making procedure.

5. Discussion

Although there is a general requirement to implement CBA for significant public projects and regulations (Ministry of Finance 2016), such analyses are often omitted (DFØ 2017). There may be various reasons for such omissions, among them lack of scientific knowledge, ethical, political and technical barriers. In this paper, our main focus was on the lack of scientific knowledge, but our empirical cases indicated that other reasons strongly contributed to the omission of ECBA. Hence, taking into account this duality when it comes to failure of ECBA implementation, this paper seeks to put the knowledge gaps issue in the context of societal constraints.

Knowledge requirements for conducting ECBA are broader than those related to ES analysis. In particular, the comparative study of two environmental measures in this paper highlights the cost dimension in the widely discussed problem of ecological-economic links quantification. The results suggest that challenges with regard to ECBA arise not only in defining the benefits of conservation, but in relating them to the costs. Without this knowledge, an ECBA cannot be set up. This aspect can partly explain the lack of ECBA in the TS case. The task of quantification of benefits in relation to costs is especially complex in cases where environmental externalities are among the main effects and where opportunity costs are among the main costs, as in TS example. Such quantification will also require incorporation of the polluters' behavior that represents a feedback effect in the ecological-economic system.

The problem of quantification of ecological-economic links has been mainly discussed in terms of determining the benefits of ecosystems. Examples include production function approach with application to fisheries-habitat interaction (Barbier 2007, Foley et al. 2010, Barbier 2019). The literature exploring the cost aspect of the relationship is not abundant. One of the few examples is the analyses of fishery-habitat links by Armstrong and Falk-Petersen (2008). They point to the possibility of fisheries itself to damage the habitat, which poses an externality directly back to the harvest and associated economic value. In an earlier study, Barbier (2003) compared the benefits of aquaculture to the environmental cost due to habitat destruction, but the functional relationship was not quantified.

The cost aspect is also lacking in the literature with regard to the behavioral feedback, where models such as one developed by Fenichel and Abbott (2014) apply to benefit quantification. Highlighting the cost dimension is the main contribution of the present paper and a potentially prominent research area that can contribute to the ECBA applicability.

Our analysis of the two cases in Atlantic salmon management also revealed some specific aspects of the known knowledge gaps in the process of ES valuation, NPV calculation, treatment of uncertainty and distributional issues of CBA. For example, we suggest that biases related to affective valuation is a research area which can be further developed. Another research question emerged from the study is how to conduct a CBA of complex measures designed as decision rules. Here, the methods for performing an NPV-test and incorporating uncertainty are called for. More research is needed on the methods for combining costs and benefits information with distributional considerations. Although these knowledge gaps might not be crucial for application of ECBA, addressing these issues would improve the overall applicability of this tool.

In addressing these research needs, it is important to account for ethical, political and technical constraints of ECBA use so that the methods and tools developed by economic scholars do not further strengthen these limitations. Adjusting the research agenda to the societal factors will increase ECBA applicability. While it is difficult to provide specific recommendations of how one can harmonize the research with the current ethical, political and technical environment, this study yields some general considerations.

Ethical factors seem to be particularly important for the valuation of ES and monetizing trade-offs. Here, researchers may prioritize methods that are less “commodifying” and therefore, ethically easier accepted, such as revealed preference methods. Development of the concepts for including non-priced effects into project appraisal together with monetary information would also be beneficial in this respect.

Political factors should be taken into account when addressing distributional issues and defining positive and negative effects of the measure. It is desirable that research can contribute to a more informative economic analysis by better incorporating distributional issues and emphasizing the bearers of the costs and benefits.

Technical constraints apply to all knowledge regarding ECBA. User-oriented research should prioritize the tools that are not too complex, easy to understand and apply. A related issue is the need for readily available values relevant for ongoing management problems. Research can also contribute to improved guidelines for CBA application in order to reduce the technical possibility of avoiding the full analysis where such analysis is required.

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Appendix 1

Survey questionnaire for Paper 2

Welcome!

How important it is to protect marine environment? Share your opinion and take part in a research project about Norwegian marine species!

[Following text is shown to participants in group 0]

The goal of this survey is to find out what wildlife in the ocean is worth for us and what we think about conservation of marine species in the Norwegian waters. Your answers will contribute to better prioritising of conservation measures and more effective environmental management. We have chosen four marine species to ask you about:

Photo: Colourbox.com

1. European lobster (*Homarus gammarus*)

Lobster is a marine crustacean which is usually found at 5-40 m depth on rocky substrates. In the Norwegian waters, lobster is abundant from the Sweadish boarder to Trøndelag, but can sometimes be found in Norland. Lobster population along the Norwegian coast is significantly reduced compared with the period 1928-1960, as a result of overfishing. In the Red List 2010 lobster was assessed as near threatened.

Photo: Colourbox.com

2. Lomvi (*Uria aalge*)

Lomvi is one of the most abundant seabirds in the temperate and subarctic areas in the northern hemisphere. In Norway, lomvi nests in few small colonies along the coast from Rogaland to Varangerfjord. The total estimated reduction of the Norwegian population of lomvi is over 80% for its last three generations (1967-2014). One of the reason is the lack of food because of natural changes in fish abundance combined with overfishing. The species is assessed as critically endangered in the Red List.

Photo: Øystein Aas/www.nina.no

3. Atlantic salmon (*Salmo salar*)

Salmon is an anadromous fish, which means that it hatches and grows in fresh water until it is one to five years old, before it migrates to the sea. Salmon is found in rivers all along the Norwegian coast. Abundance of salmon returning to rivers in Norway has reduced by over 50%. This is partly because of its reduced survival at sea. In addition, local and regional factors influence wild salmon significantly.

Photo: Jon Aars/Norsk Polarinstitutt

4. Bowhead whale (*Balaena mysticetus*)

Bowhead whale is a big baleen whale species which spends all its life in the arctic waters. In the period from 17^{th} to 19^{th} century it was almost extinct in all areas of habitat. Spitsbergen population, which is partly within Norwegian waters is one of the most threatened big whale populations in the world and therefore it is redlisted in Norway.

The questionnaire consists of two parts and takes only 5 minutes to fill inn. The first part is a quiz of 8 questions that will test your knowledge about these species. The right answers come in the end of the questionnaire. In the second part we will ask you about your attitudes to conservation of the species.

Part 1 – Quiz

1. The diet of lobster consists mainly of sea plants and algae.

- o True
- o False

2. Lobster can be 32 years old.

- o True
- o False

3. Lomvi is the biggest of all currently living auks.

- o True
- o False

4.. Lomvi usually lays two eggs because there is a big chance of one egg being lost.

- o True
- o False

5. Salmon juvenile which is ready to migrate to sea is called «parr».

- o True
- o False

6. Salmon produces a type of pigment called carotenoid in its body.

- o True
- o False

7. Females of bowhead whale is a bit larger than males.

- True
- False

8. Bowhead whale has no natural predators.

- True
- False

The right answers come in the end of part 2.

[Following text is shown to participants in the groups 1-4]

The goal of this survey is to find out what wildlife in the ocean is worth for us and what we think about conservation of marine species in the Norwegian waters. Your answers will contribute to better prioritising of conservation measures and more effective environmental management. We have chosen four marine species to ask you about:

[same information about the species is shown]

In the next step, you will be offered to fill in a questionnaire about one of these species. This is done in order to save time of each participant. The questionnaires are assigned in the random order. Each questionnaire consists of two parts and takes only 5 minutes to fill inn. The first part is a quiz of 8 questions that will test your knowledge about the species. Right answers come in the end of the questionnaire. In the second part we will ask you about your attitudes to conservation of the species.

[Group 1]

You have been assigned questionnaire about lobster.

Photo: Colourbox.com

European lobster (*Homarus gammarus*)

Lobster is a marine crustacean which is usually found at 5-40 m depth on rocky substrates. In the Norwegian waters, lobster is abundant from the Sweadish boarder to Trøndelag, but can sometimes be found in Norland. Lobster population along the Norwegian coast is significantly reduced compared with the period 1928-1960, as a result of overfishing. In the Red List 2010 lobster was assessed as near threatened.

Part 1 - Quiz

1. The diet of lobster consists mainly of sea plants and algae.

- True
- False

2. Lobster can be 32 years old.

- True
- False

3. Which statement is right?

- a) Catches of lobster in Norway are on the same level as they were in 1960s.
- b) Catches of lobster in Norway are 10 times higher compared to 1960.
- c) Catches of lobster in Norway were 10 times higher in 1960 than today.
- d) Fishing for lobster is prohibited in Norway today.

4. Which statement is right?

- a) Recreational fishers have to report catch of lobster regardless its size.
- b) Commercial fishers have to report catch of lobster regardless its size.
- c) Catch of lobster should be reported if it is over 25 cm.
- d) Catch of lobster should be reported if it is over 15 cm.

5. Which statement is wrong?

- a) Conservation areas for lobster appeared to have no effect on the abundance.
- b) Within the conservation areas, lobster it is protected through the restrictions on fishing equipment.
- c) Conservation areas for lobster are established in Skagerrak.
- d) In conservation areas, the average size of lobsters is larger than in other areas.

6. Which statement is wrong?

- a) Minimum size for lobster fishing is 25 cm.
- b) There is no minimum depth requirements for lobster traps.
- c) Lobster caught with other equipment than traps should not be released back to sea.
- d) Recreational fishers cannot sell lobster.

7. American lobster, which is a non-native species for Norway, can interbreed with the European lobster.

- True
- False

8. Lobster should not be killed by cooking.

- True
- False

The right answers come in the end of part 2.

[Group 2]

You have been assigned questionnaire about lomvi.

Photo: Colourbox.com

2. Lomvi (*Uria aalge*)

Lomvi is one of the most abundant seabirds in the temperate and subarctic areas in the northern hemisphere. In Norway, lomvi nests in few small colonies along the coast from Rogaland to Varangerfjord. The total estimated reduction of the Norwegian population of lomvi is over 80% for its last three generations (1967-2014). One of the reason is the lack of food because of natural changes in fish abundance combined with overfishing. The species is assessed as critically endangered in the Red List.

Part 1 - Quiz

1. Lomvi is the biggest of all currently living auks.

- True
- False

2. Lomvi usually lays two eggs because there is a big chance of one egg being lost.

- True
- False

3. Which statement is right?

- a) Lomvi is often caught as bycatch in the line fishery for Greenland halibut.
- b) Lomvi is rarely caught in fishing nets.
- c) Lomvi often drowns in cod trawls.
- d) Over 100 thousand lomvi die yearly in fishing nets in Norway.

4. Which statement is right?

- a) Healthy capelin stock is important for lomvi survival.
- b) Healthy stock of big cod is important for lomvi survival.
- c) Crab fishing is a direct cause of the reduction of lomvi populations.
- d) Shrimp fishing is a direct cause of the reduction of lomvi populations.

5. Which statement is wrong?

- a) Lomvi population on Bjørnøya in the Barents Sea has increased as a result of conservation measures.
- b) Lomvi population on Hornøya in Finnmark has increased as a result of good access to food.
- c) Lomvi population on Hornøya in Finnmark is considered as one of the most resilient of the Norwegian coastal lomvi populations today.
- d) Lomvi population on Bjørnøya in the Barents Sea is just a half of its size compared to 1986.

6. Which statement is wrong?

- a) Reduction of pollution can contribute to conservation of lomvi.
- b) Reduction of overfishing can contribute to conservation of lomvi.
- c) Hunting for predatory birds is established in order to prevent reduction of lomvi populations.
- d) It is recommended to deploy fishing nets deeper than 50 m in order to prevent bycatch of lomvi.

7. Oil spill can kill large amount of lomvi, but his has never happened in Norway.

- True
- False

8. Lomvi is a traditional food in Greenland.

- True
- False

The right answers come in the end of part 2.

[Group 3]

You have been assigned questionnaire about wild salmon.

Photo: Øystein Aas/www.nina.no

Atlantic salmon (*Salmo salar*)

Salmon is an anadromous fish, which means that it hatches and grows in fresh water until it is one to five years old, before it migrates to the sea. Salmon is found in rivers all along the Norwegian coast. Abundance of salmon returning to rivers in Norway has reduced by over 50%. This is partly because of its reduced survival at sea. In addition, local and regional factors influence wild salmon significantly.

Part 1 - Quiz

1. Salmon juvenile which is ready to migrate to sea is called «parr».

- True
- False

2. Salmon produces a type of pigment called carotenoid in its body.

- True
- False

3. Which statement is right?

- a) Hydropower development is considered to be the most serious threat to salmon populations.
- b) The introduced parasite Gyrodactylus salaris causes deadly infestations of Norwegian wild salmon.
- c) Acid rains is the biggest threat to salmon populations.
- d) Fishing in rivers is the main cause of the reduction in salmon populations.

4. Which statement is right?

- a) Sterile farmed salmon cannot interbreed with wild salmon.
- b) Escaped farmed salmon cannot survive at sea.
- c) Escaped farmed salmon cannot survive in fresh water.
- d) All farmed salmon in Norway is sterile.

5. Which statement is wrong?

- a) In Norway, salmon ranching without permission from authorities is not allowed.
- b) Ranching is allowed as a measure which compensates for negative impact of escaped farmed salmon.
- c) Ranching is allowed where the salmon population is lost.
- d) Hydropower plants owners are obliged to maintain local salmon populations.

6. Which statement is wrong?

- a) Over 20 fjords in Norway have the status of national salmon fjords.

- b) Over 50 rivers in Norway have the status of national salmon rivers.
- c) Salmon farming is not allowed in national salmon fjords.
- d) Hydropower installations are allowed on national salmon rivers.

7. Farmed salmon has genes which are not found in the wild salmon.

- True
- False

8. Two-three salmon lice attached are fatal for the migrating salmon smolt.

- True
- False

The right answers come in the end of part 2.

[Group 4]

You have been assigned questionnaire about bowhead whale.

Photo: Jon Aars/Norsk Polarinstitutt

4. Bowhead whale (*Balaena mysticetus*)

Bowhead whale is a big baleen whale species which spends all its life in the arctic waters. In the period from 17^{th} to 19^{th} century it was almost extinct in all areas of habitat. Spitsbergen population, which is partly within Norwegian waters is one of the most threatened big whale populations in the world and therefore it is redlisted in Norway.

Part 1 - Quiz

1. Females of bowhead whale is a bit larger than males.

- True
- False

2. Bowhead whale has no natural predators.

- True
- False

3. Which statement is right?

- a) Hunting for bowhead whale is allowed in Alaska, Canada, Greenland, and Svalbard.
- b) One-two bowhead whales are caught yearly near Svalbard.
- c) Hunting for bowhead whale is forbidden everywhere except Russia.
- d) Fangst av grønlandshval reguleres med kvotesystem i Alaska, Canada og Grønland.

4. Which statement is right?

- ○ a) Spitsbergen population of the bowhead whale is less than 100 animals.
- ○ b) Spitsbergen population of the bowhead whale is about 500 animals.
- ○ c) Initial size of the Spitsbergen population of the bowhead whale was about 300 000 animals.
- ○ d) Initial size of the Spitsbergen population of the bowhead whale was about 5 000 animals.

5. Which statement is wrong?

- ○ a) It is possible to use satellite transmitters to track bowhead whale.
- ○ b) Bowhead whale is tracked using passive acoustic buoys.
- ○ c) It is not possible to track bowhead whales using common counting techniques.
- ○ d) Bowhead whales are monitored using DNA analysis.

6. Which statementt is wrong?

- ○ a) Norway is a member of the International Whaling Commission.
- ○ b) International Whaling Comission introdused a total ban on all commercial catch of whales.
- ○ c) Total ban on the commercial catch of whales is not enforced in Norway.
- ○ d) Bowhead whale is one of the exceptions in the total ban on commercial whaling.

7. Products from bowhead whale is often used in cosmetics and perfumes.

- ○ True
- ○ False

8. It was easy to hunt bowhead whales in the old days because they swim very slowly.

- ○ True
- ○ False

The right answers come in the end of part 2.

Part 2 – conservation

[Following text is shown to participants in group 0]

[same information about the species is shown]

9. How would you describe your feelings thinking about current status and survival of these species?

	Very optimistic	Quite optimistic	Neutral/don't know	Quite worried	Very worried
Lobster	○	○	○	○	○
Lomvi	○	○	○	○	○

Atlantic salmon	◯	◯	◯	◯	◯
Bowhead whale	◯	◯	◯	◯	◯

10. Consider a hypothetic situation. A state-owned environmental fund has NOK 40 million available for four projects. Each project is designed for conservation of one of the species, where the measures are taken to reduce existing threats to species survival. How do you think the budget should be distributed? Write a sum (in millions) in the boxes for each project, so that the total sum becomes 40 million. It is possible to assign the whole budget or zero to a project. *

[ ] Lobster

[ ] Lomvi

[ ] Atlantic salmon

[ ] Bowhead whale

0 out of 40 Total

11. To what extent have you considered following factors in the distribution of the budget? *

	Did not consider	Considered to some extent	Considered to a large extent
Importance of the species for the ecosystem	◯	◯	◯
Importance of the species for the Norwegian economy	◯	◯	◯
Importance of the species for society and future generations	◯	◯	◯
Importance of the species for you personally	◯	◯	◯

The amount of money available and assumptions about the projects' costs	◯	◯	◯
Fairness	◯	◯	◯
Other (specify)	◯	◯	◯

12. If we were to donate NOK 100 on behalf of each participant of this survey to one of the projects, which project would you choose? *

- ◯ Lobster
- ◯ Lomvi
- ◯ Atlantic salmon
- ◯ Bowhead whale

13. Why? *

Don't know/have chosen randomly

14. How likely do you think the results from this survey will contribute to better prioritising and conservation of the species? *

1 (not likely)	2	3	4	5 (very likely)	Don't know
◯	◯	◯	◯	◯	◯

[Following text is shown to participants in groups 1-4]

9. How would you describe your feelings thinking about current status and survival of lobster/lomvi/Atlantic salmon/bowhead whale? *

- ◯ Very optimistic
- ◯ Quite optimistic
- ◯ Neutral/don't know
- ◯ Quite worried
- ◯ Very worried

10. Consider a hypothetic situation. A state-owned environmental fund has NOK 40 million available for four projects. Each project is designed for conservation of one of the species, where the measures are taken to reduce existing threats to species survival. How much of these 40 million should be assigned to the lobster/lomvi/wild salmon/bowhead whale-project in your opinion? *

0 40

NOK 40 million is available for conservation of the four species:

[same information about the species is shown]

11 To what extent have you considered following factors in assigning a budget to the project? *

	Did not consider	Considered to some extent	Considered to a large extent
Importance of lobster/lomvi/wild salmon/bowhead whale for the ecosystem	◯	◯	◯
Importance of lobster/lomvi/wild salmon/bowhead whale for the Norwegian economy	◯	◯	◯
Importance of lobster/lomvi/wild salmon/bowhead whale for society and future generations	◯	◯	◯
Importance of lobster/lomvi/wild salmon/bowhead whale for you personally	◯	◯	◯
The amount of money available and assumptions about the projects' costs	◯	◯	◯
Fairness	◯	◯	◯
Other (specify)	◯	◯	◯

12. If we were to donate NOK 100 on behalf of each participant of this survey to one of the projects, which project would you choose? *

- ◯ Lobster
- ◯ Lomvi
- ◯ Atlantic salmon
- ◯ Bowhead whale

13. Why? *

Don't know/have chosen randomly

14. How likely do you think the results from this survey will contribute to better prioritising and conservation of lobster/lomvi/wild salmon/bowhead whale? *

1 (not likely)	2	3	4	5 (very likely)	Don't know
◯	◯	◯	◯	◯	◯

[Following text is shown to all participants]

Almost finished! A couple of questions about you.

15. You are *

- ◯ Man
- ◯ Woman

16. How old are you? *

17. Where do you live (fylke)?

--Please select--

18. What is your highest completed education?

- ◯ Primary school
- ◯ Secondary school
- ◯ High school
- ◯ College or university, bachelor
- ◯ University, master
- ◯ Other education (spesify)

Thank you for participation! Click here to finish and register your answers.

Your result: [number of right answers is shown, the right answers for the quiz are shown to the participant]

Appendix 2

Interview guide for Paper 3

The value of wild salmon: application of environmental valuation and CBA in environmental management

This interview is part of the PhD-project at UiT The Arctic University of Norway. The project concerns the use of economic analytical tools in the wild salmon management. To ensure the quality of data collection, we are going to record the interview. All personal data will be processed confidentially. In the publication of the results, there will not be possible to identify individual participants.

Thank you for the participation!

The first few questions are about your role in the management of wild salmon.

1. How would you describe your role in the management of wild salmon?

2. In what cases regarding salmon management have you been involved in the recent years (e.g., aquaculture regulation, regulation of other industries that have impacts on wild salmon, treatment of rivers with rotenone, liming of rivers, protection measures, fishery management)?

In the invitation to this interview, we have suggested specific cases that vi can discuss more thoroughly. We now focus on them.

3. What interest groups/stakeholders were represented in these cases and what were their objectives? What was the standpoint of your institution in the case?

4. Were environmental-economic trade-offs discussed? Was a CBA presented in the assessment of the case?

Discuss the Instructions on official studies and reports (Utredningsinstruksen, 2016). Are they followed?

5. Are you familiar with the concept of ecosystem services and their valuation?

Discuss the values in the report by Atlantic Salmon Federation (2011) as an example.

6. Are you aware of any cases where these concepts were applied? If yes, to what extent it influenced the decision-making process? If no, why in your view, they were not applied? Same question about CBA.

In the following questions we would like to know your personal opinion, where possible.

7. How would you describe your attitude to the concepts of ecosystem services, valuation and CBA?

8. Do you agree that wild salmon has a value?

9. What does this value comprise?

10. In what follows, we will present arguments that stakeholders can use to influence decisions in various cases regarding salmon management. Do you agree or disagree with these arguments? Are they relevant for decision making in the particular cases?

1. Measures to eradicate G.Salaris and liming of rivers.

The benefits of wild salmon conservation are larger than the costs of the measures.	Agree ☐ Disagree ☐	Relevant ☐ Not relevant ☐

The measures are costly and the ways for minimizing costs should be considered.	Agree ☐ Disagree ☐	Relevant ☐ Not relevant ☐

2. Protection measures (for example, special protection regime under the status of National Salmon Rivers and Fjords)

We cannot put all salmon rivers under special protection. Some rivers and salmon populations are more important than others.	Agree ☐ Disagree ☐	Relevant ☐ Not relevant ☐

3. Regulation of angling and sea fishing for salmon

The economic benefits of sea fishing are smaller than the benefits of angling.	Agree ☐ Disagree ☐	Relevant ☐ Not relevant ☐

Sea fishing is part of the harvest culture and therefore contributes to the welfare.	Agree ☐ Disagree ☐	Relevant ☐ Not relevant ☐

Angling is an important activity for maintaining the quality of life and therefore contributes to the welfare.	Agree ☐ Disagree ☐	Relevant ☐ Not relevant ☐

4. **Aquaculture regulation**

Aquaculture has a certain negative impact on the environment, but the damage is compensated by its economic benefits.	Agree ☐ Disagree ☐	Relevant ☐ Not relevant ☐

Discuss the article «Let the wild salmon die out…» by Martinsen, 2010, about the trade-off between farmed and wild salmon

There can be found more sustainable ways to utilize natural resources, that give the same economic benefit as aquaculture (e.g., tourism industry)	Agree ☐ Disagree ☐	Relevant ☐ Not relevant ☐

A 6% reduction of the MAB under the TS will impose large costs. At the same time, we do not know whether the reduction will have a positive effect on the wild salmon.	Agree ☐ Disagree ☐	Relevant ☐ Not relevant ☐

5. **Regulation of other industries that have an impact on wild salmon (for example, mining project in Kvalsund)**

The expected economic benefits of the project outweigh the environmental damage such as abundance and quality of wild salmon.	Agree ☐ Disagree ☐	Relevant ☐ Not relevant ☐

Potential environmental effects of the mining project should be assessed on monetary scale to ensure complete costs and benefits information.	Agree ☐ Disagree ☐	Relevant ☐ Not relevant ☐

There can be found other ways of utilizing natural resources, that will have the same level of environmental damage, but will be more profitable.	Agree ☐ Disagree ☐	Relevant ☐ Not relevant ☐

Thank you for your time!

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