# ENVIRONMENTAL CHALLENGES IN THE BALTIC REGION

*A Perspective from Economics*

Edited by
Ranjula Bali Swain

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## A Perspective from Economics

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# Acknowledgements


*No one saves us but ourselves. No one can and no one may. We ourselves must walk the path.*

*Buddha*

The Baltic Sea is regarded as the most damaged sea in the world. Some of the major challenges in the Baltic Sea region are its eutrophication (caused by nutrient pollution), hypoxia (low oxygen), hazardous substances, oil spills, invasive species, marine litter and resulting changes in flora and fauna. To discuss some of these challenges, we organized an International Workshop in May 2016, during which the idea for this edited book precipitated. My gratitude and thanks are due to all the contributing authors of this edited volume who also provided stimulating intellectual discussions and inputs during the workshop—Stig Blomskog, Patrik Dinnetz, Katarina Elofsson, Ing-Marie Gren, Kari Lehtilä, Mari Jüssi, Markus Larsson, Helen Poltimäe, Sirje Pädam, Tea Nõmmann and Tomasz Zylicz. For excellent research assistance and support, a special thanks to Dr. Ashim Kar. Thank you Aimee Dibbens and Thomas Coughlan for your support, expertize and encouragement through the publication process.

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# Environmental Challenges in the Baltic Region: An Introduction

Ranjula Bali Swain

The Baltic Sea is one of the largest semi-enclosed bodies of brackish water in the world. Nine countries (Denmark, Estonia, Finland, Germany, Latvia, Lithuania, Poland, Russia, Sweden) with a population of over 90 million share the sea (Ahtiainen et al. 2014). Its geography, climatology and oceanography have great political, social, economic and cultural significance for the people in Baltic Europe and its importance has grown as the Baltic states have become a part of the European Union (HELCOM 2010). The sea is shallow and, being an almost entirely landlocked body of water, receives a considerable load of pollutants from surrounding countries. The severe environmental impact of human activities is altering the marine ecosystem, depleting renewable

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resources beyond safe biological limits, and jeopardizing the future use of the Baltic ecosystem goods and services (HELCOM 2007, 2010).

In most parts of the Baltic Sea, major concerns are related to its eutrophication (caused by nutrient pollution), hypoxia (low oxygen), hazardous substances, oil spills, invasive species, marine litter and subsequent changes in flora and fauna (Tynkkynen et al. 2014; Elofsson 2003; Conley et al. 2009; Ahlvik and Pavlova 2013). An increase in the inflow of nutrients into the sea from agriculture, wastewater, industry and traffic has led to growth in organic production considerable eutrophication (Swedish Agency for Marine and Water Management 2013; Österblom et al. 2007). The difficulty of managing this is exacerbated by the complexecological characteristics of the eutrophication problem, social differences across the Baltic Sea region, and the multiplicity of stakeholders involved in governing these efforts. This has resulted in a variation in the level of awareness of the problem, national and subnational goals, the ability to address it through national policies and the strengthening of policy implementation across the region. The absence of a legal arrangement to protect the Baltic Sea, covering all the coastal countries, makes the situation even more complex (Swedish Agency for Marine and Water Management 2013; Tynkkynen et al. 2014; Ahlvik and Pavlova 2013; HELCOM 2011).

Blue-green algal blooms at the bottom of the sea, along with hypoxia, have both extended by tenfold (Swedish Agency for Marine and Water Management 2013; Savchuk et al. 2008). Living organisms and bottom sediments are affected by hazardous substances in all parts of the Baltic Sea. Environmentally alarming shifts and imbalances appear in many habitats and across the food chain, particularly at the level of large fish (HELCOM 2010). These, in combination with overfishing, have resulted in several regime shifts in the food web. Climate change has caused the sea surface temperature to rise by 0.7 °C during the twentieth century (Swedish Agency for Marine and Water Management 2013). All these factors influence the ecosystem services of the Baltic Sea and hence diminish the benefits generated to the people and the society of this region (HELCOM 2010).

The Baltic Sea underwent a regime shift over the twentieth century (Österblom et al. 2007). Its ecological degradation has been a major challenge for the people and the governments. The surrounding countries have struggled to protect the sea by attempting to reduce the discharges from industries, municipalities and shipping (Mosin 2011). Signed in 1974, the Helsinki Convention was one of the first agreements in the world with the objective to protect a whole sea area from different pollutants. Other initiatives, like the Local Agenda 21, have also been adopted by all the coastal states to improve democratic environmental policymaking and protection.

Given the different agendas regarding issues of exploitation and environmental protection, there is an immense potential for international conflict over the Baltic Sea, which has been studied by a few researchers. Information on the environmental history of the Baltic Sea region, however, is limited as the literature pertaining to its various aspects is in several different languages. There is often pressure on policymaking within and among states to bring about change. Such change can be empirically observed in the form of the activation of different network structures in the Baltic Sea region, especially since the collapse of the Iron Curtain, the initiation of the Rio Process and the expansion of the European Union. Contemporary theoretical debates about governance highlight the changing conditions that underline the making and implementation of policy at all societal levels. Especially evident when it comes to environmental policies, these include the emergence of new types of networks across state borders, both at the supranational and the subnational levels. Joas et al. (2007) elucidate this process of change with empirical data from the project "Governing a Common Sea" within the Baltic Sea Research Program.

Reviewing the administrative and political structures, Joas et al. (2008) note that the littoral states in the Baltic Sea region have established several new forums and modes of cooperation to manage the sea.

Kapaciauskaite (2012) emphasizes the emergent role of non-governmental actors in regional environmental governance and highlights the coming to the fore of transnationalization, Europeanization tendencies and the largely fragmented nature of existing governance structures in the region. Gilek et al. (2015) present an interdisciplinary analysis of challenges and possibilities for the sustainable governance of the Baltic Sea ecosystem. Focusing on the Ecosystem Approach to Management (EAM) and associated multi-level, multi-sector and multi-actor challenges, they analyse the environmental governance structures and processes at the macro-regional Baltic Sea level. They conclude that the governance of the Baltic Sea may be improved by promoting environmental governance through coordination, integration, interdisciplinarity, precaution, deliberation, communication and adaptability. A comparative overview of the environmental and resource problems experienced in the Nordic and Baltic regions can be found in Aage (1998).

The main challenges at different governance levels include: differences between coastal countries in terms of environmental conditions, environmental awareness, policy overlap, inadequate spatial and temporal specification of policies, and the lack of policy integration. To meet these challenges, some researchers suggest the closer involvement of stakeholders and the public, improvement in the interplay of institutions and the introduction of a "primus motor" to govern themitigation of eutrophication in the Baltic Sea (Tynkkynen et al. 2014).

The initial sections of the book discuss the various aspects of eutrophication in the Baltic Sea. The food system and the specialization of agriculture have been the main source of this eutrophication (HELCOM 2005; Granstedt 2000). In Chap. 2, "Towards a Sustainable Food System in the Baltic Sea Region", Larsson compares conventional agriculture and Ecological Recycling Agriculture (ERA) in terms of their environmental and socio-economic effects, with a focus on nutrient losses. Larsson argues that socio-economic effects include production, costs and benefits at the macro, firm and household level. At the regional level, the main challenge is to make agriculture more environmentally friendly and reduce nutrient losses while maintaining food production. At the national level, it is to shift the product mix towards more vegetables and less meat and to address the geographical division between animal and crop production. Finally, at the local level, the challenge is to achieve sustainable environmental, economic and social rural development.

Larsson scales up the empirical findings at the regional level to create three scenarios. In the first,agriculture in Poland and the Baltic states is transformed to resemble the Swedish average structure and resource use, which results in a 58% increase in nitrogen and an 18% increase in phosphorus surplus in agriculture, with a substantial rise in food production. In the other two scenarios agriculture in the entire Baltic Sea area is converted to ERA. This results in a 47–61% reduction in nitrogen surplus in agriculture and eliminates the phosphorus surplus, while food production either decreases or remains stable, conditional on the strategy chosen.

On comparing the environmental effects of different production methods, modes of transport and food baskets at the national level, Larsson finds that the food basket content is as important as the production method in reducing the environmental effects. Local production and processing are less significant. He sees the expansion of the EU as an opportunity for better governance of the Baltic Sea and the agriculture sector. According to him, a new agricultural regime with large-scale ERA would produce several environmental gains. The sustainable governance of the Baltic Sea, as agreed in the Baltic Marine Environment Protection Commission (HELCOM) or the Helsinki Commission, cannot be achieved while simultaneously maximizing agricultural production in surrounding countries. Agricultural production has large external costs. There is substantial willingness to pay for an improved Baltic Sea environment among the public, justifying environmentally sound farming practices. Larsson argues that the contracting parties to HELCOM, including the Swedish government, have environmental and economic incentives to use the opportunities offered by the EU membership of Poland and the Baltic states.

Chapter 3, "Cost-effective Management of a Eutrophicated Sea in the Presence of Uncertain Technological Development and Climate Change", investigates the effects of climate change and technological development on the cost-effective abatement of nitrogen and phosphorus on a eutrophied Baltic sea. In this chapter, Gren develops a dynamic model, which accounts for differences in the sea's adjustment to changes in the nitrogen and phosphorus loads under two types of uncertainty. One is the uncertainty of climate change effects, which is approached with probabilistic constraints on nutrient pool targets. The other is uncertainty of technological development, which is treated within a mean-variance framework in the objective function. The analytical results show that the effects of introducing uncertainty on marginal abatement cost differ for the two types of uncertainty. Marginal abatement cost is increased by technological uncertainty but decreased by the reduction in the risk discount of climate change uncertainties. Gren also shows that abatement along the optimal time path is delayed by the introduction of technological uncertainty, but occurs earlier when considering climate change uncertainty. Applying this to the eutrophied Baltic Sea reveals that climate change and technological development can reduce the total abatement cost by one-third, but also increase it considerably when uncertainty is included.

Eutrophication of the Baltic Sea has been recognized as a major problem since the 1960s. Nutrient emissions originate from point and non-point sources in the agricultural, transport, energy and wastewater sectors. Elofsson examines the "Optimal Strategies for Inland and Coastal Water Monitoring" in Chap. 4. Over the last few years, there has been some success in nutrient load reduction in the Baltic Sea, but the environmental conditions of the sea have not improved significantly. Many large aquifers across the world suffer from increased eutrophication with negative consequences for biodiversity, fishery, recreation and ecosystem health. Challenges include identification of the relationship between activities at upstream sources and the state of the recipient, evaluation of the environmental status of the recipient and identification of the benefits of abatement.

Eutrophication of inland recipients, often but not always, occurs together with the eutrophication of downstream coastal waters. Sometimes, however, one of these recipients is eutrophicated but not the other. For example, high nutrient retention could imply that emissions from a source reach nearby lakes and rivers but do not reach downstream coastal waters. Also, downstream coastal waters could be in good condition even when nutrient loads from upstream sources are high, for example, if there is a high degree of dilution.

Elofsson investigates the optimal monitoring and abatement strategies in a situation where both upstream and downstream water quality is a potential problem. In particular, she examines how monitoring and abatement costs, and the regulators' degree of risk aversion, affect the choice of monitoring strategy. A stylized model with two upstream sources and one upstream and one downstream recipient is used for the analysis, and generic data are used for the simulations. Elofsson suggests that the optimal choice is either to not monitor, or to first monitor the sources and based on the outcome, decide whether to proceed with downstream monitoring. The latter strategy is preferred if the cost of upstream monitoring is relatively low, or abatement costs or risk aversion are relatively high.

The EU Marine Strategy Framework Directive (MSFD) requires countries to suggest new measures to achieve Good Environmental Status (GES) of the marine environment by 2020. MSFD explicitly asks member states to ensure that planned measures are cost-effective, technically viable and that impact assessments, including cost-effectiveness and cost-benefit analyses, have been carried out prior to the introduction of new measures.

In Chap. 5, "Public Policies towards Marine Protection: Benchmarking Estonia to Finland andSweden", Nõmmann and Pädam compare the approaches for cost-effectiveness analysis (CEA) and cost-benefit analysis (CBA) of the new measures proposed by Estonia, Finland and Sweden. Due to uncertainties, the lack of background studies and multidisciplinary models of sea ecosystem management, these countries have employed qualitative expert assessments. While Sweden and Estonia have applied standard methods to appraise impacts, Finland has adopted an innovative probabilistic approach.

Proposed measures are expressed in terms of intended objectives rather than in terms of their implementation. Administrative measures, awareness raising, research and development, and other means of information provision are part of the country's first National Programme of Measures. However, as means of implementation, the impact of information is often minor. Uncertainty regarding the choice of policy instruments for implementation complicates both the appraisal of the impact on the environmental target and the estimation of costs and benefits. For the next cycle, it is important to build up knowledge about policy instruments and implementation. There is a need for reviews of existing ex-post studies and further studies, which evaluate existing policy instruments to protect marine environments. Nõmmann and Pädam argue that in order to achieve GES in the entire Baltic Sea, it is important to consider cross-country coordination of measures, as one country alone cannot achieve GES in its national marine area. Limited public resources at the national level to conduct the requisite valuation studies for CEA and CBA is a problem, but coordination opens up opportunities for collaborations at the regional level and for valuation studies to arrive at the CEA and CBA across neighbouring countries.

The process of economic growth leads to several other modes of environmental degradation. In Chap. 6, Poltimäe and Jüssi study the "Factors Affecting Travel Mode Choice in Tallinn". Cars are increasingly being used for daily commuting as compared to modes ofpublic transport, cycling and walking. The city of Tallinn in Estonia has made several efforts to advance a sustainable transport policy: public transport is free of charge for its citizens, parking fees have been increased and the area of paid parking expanded. Still, car use is on the rise and the use of public transport is decreasing.

Poltimäe and Jüssi aim to investigate the key factors related to choice of mode of transport among Tallinn's citizens, specifically with respect to the use of cars and public transport. In this chapter they analyse the household travel survey data collected by TNS Emor in Tallinn during 2015. Although the number of trips made and daily time spent on travelling in Estonia is still lower than in most highly developed countries, these figures are rising rapidly. They find that increasing car use is not only related to income but also to car compensation, which is offered by employers and enabled by the Estonian tax system. Some of the daily car drivers prefer it for the independence and comfort. However, most of the respondents claim to use cars because of distance and accessibility. These people could potentially be weaned off cars in the presence of a public transport system or cycling network that could meet their needs.

A large share of public transport users claim to opt for it because it is comfortable. Poltimäe and Jüssi suggest building on this, both in terms of the quality of and accessibility to public transport. Urban planning is also significant since parts of Tallinn city have expanded without integrating public transport and mobility planning, which limits the choice of mode of transport available to its inhabitants.

Chapter 7 discusses the "Environmental Impacts of Rural Landscape Change During the Post-communist Period in the Baltic Sea Region".

In this chapter, Lehtilä and Dinnetz discuss the environmental effects of rural land use change in Eastern Europe during the post-communist period. They compare rural land use change and its effects in Eastern and Northern Europe, two areas with different histories of landscape change. They focus on the impact of land use change on biodiversity. They argue that landscape change is one of the most important anthropogenic processes affecting ecosystems. Throughout history, there have been several far-reaching transformations of Eastern and Northern European ecosystems due to agricultural transitions. The most recent one, which took place due to the collapse of the Soviet Union, resulted in large-scale changes in the rural landscapes of Eastern Europe. In many countries, more than 20% of agricultural land was abandoned, and the trend is especially strongin Estonia, where 54% of arable land was abandoned between 1992 and 2005. Land abandonment can affect a variety of ecosystem traits such as biodiversity, water supply, nutrient cycling and carbon sequestration. Lehtilä and Dinnetz argue that the effects of land abandonment on these environmental variables are diverse, and there are several possible outcomes depending both on the type of land that is abandoned and the management following the abandonment. The implications for environmental governance are similarly diverse and depend on perspectives on environmental and socioeconomic development.

Blomskog, in Chap. 8, presents "An Analysis of Permission Processes for Wind Power in Sweden". He investigates the formal reconstruction of the legal permission processes concerning permits establishing wind power stations. Reconstruction is based on the concepts applied in multiple-criteria decision making (MCDM). The motivation for reconstruction is drawn from the fact that the extensive academic analysis of these permission processes is performed in an informal everyday language. Many of the intricate conceptual problems that arise during the permission processes are, therefore, treated in an inappropriate manner. Blomskog reconstructs a typical permission process completed by the Swedish authority according to the guidelines of the Swedish Environmental Code. The reconstruction is performed in four stages. *First*, the basic decision problem and the basic norm applied in these legal permission processes are specified. In the *second stage*, according to a planned wind power installation, the expected value conflicts between value gains as production of "green" electricity and value losses as negative impacts on various environmental aspects are defined. In the *third stage,* Blomskog analyses the meaning of the application of *critical threshold values*, which is the first way of solving the value conflicts. He concludes that critical threshold values ultimately depend on the authorities' subjective, discretionary and situation-dependent judgements. In the *fourth stage*, he analyses *weighing*, which is the second way of solving value conflicts. Based on the reconstruction, Blomskog concludes that the weighing of decisions in these permission processes seems to be based on conceptual mistakes due to the use of the notion of importance. He concludes that one way to remedy misconceptions would be to implement a conceptual framework developed and applied in MCDM.

Pädam and Bali Swain investigate "Attitudes towards Paying for Environmental Protection in the Baltic Sea Region", in Chap. 9. They compare public attitudes to environmental protection in Estonia across neighbouring countries around the Baltic Sea. Responses to three questions covered by the Estonian Environmental Survey from 2010 and by the ISSP Environment III are compared and analysed using ordered logit regressions. Support for environmental protection is measured in the form of the willingness of individuals to make financial sacrifices through higher prices and higher taxes or accepting a cut in their standard of living, in order to protect the environment.

The cross-Baltic country comparison puts Estonia in the middle position. Estonia seems to have a lower-than-average acceptance to cuts in standard of living for environmental protection among countries in the Baltic Sea region. Country-level data suggest that Estonia is similar to Latvia, Lithuania, and Russia in this regard. On the other hand, its willingness to pay higher taxes and prices for environmental protection is higher than the average among countries in the region, placing it at a similar level to that of the Nordic countries and Germany.

Pädam and Bali Swain find that the demand for the protection of the environment tends to increase with income. This is true for both personal income and country-level income. Some difference can be detected between public attitudes in terms of willingness to accept cuts in standard of living, and the willingness to pay higher taxes and prices. A study of attitudes concerning monetary sacrifices shows a larger number of significant income categories than attitudes towards cuts in living standards. It is also interesting to note that the results reflect earlier findings of a stronger positive influence of personal income than of country-level wealth. Supported by previous research, this indicates that adjustments in GDP per capita do not perform well for the purposes of benefit transfer. It suggests that further attention should be paid to other variables when value estimates are transferred from one context to another.

Higher education is the second main determinant of support for environmental protection. Pädam and Bali Swain find that completion of university studies has a significant influence on the willingness to pay for environmental protection in the Baltic region. In Estonia, higher education significantly influences attitudes towards paying higher taxes. These results suggest that there is support among the general public to pay higher taxes for the purpose of environmental protection.

The final chapter in the book addresses the important question, "Is International Cooperation in the Baltic Sea Drainage Basin Possible?" Zylicz outlines the notion of Baltic Sea protection in terms of an economic public good. He argues that such a good is doomed to insufficient provision unless a financial mechanism is created to undertake abatement to a level which is justified by global considerations rather than local ones. By applying the Chander–Tulkens model of international cooperation, hypothetical transfers are estimated in order to conclude that the Baltic region is not yet ready to develop effective region-wide clean-up programmes.

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# 2

# Towards a Sustainable Food System in the Baltic Sea Region

Markus Larsson

## Introduction: Baltic Sea Agriculture

A research report from the Swedish Environmental Advisory Council, titled *A Strategy for Ending Eutrophication of Seas and Coasts*, argues that the Baltic Sea is facing an ecological flip, associated with changes characterized by excessive algal bloom and a fishing industry in crisis (MVB 2005). Eutrophication may be the most severe of the consequences faced by the Baltic Sea. According to HELCOM (2015, p. 12), "Eutrophication is a major problem in the Baltic Sea. Since the beginning of the twentieth century, the Baltic Sea has changed from an oligotrophic clear-water sea into a highly eutrophic marine environment". The increase in algae is the most obvious effect of eutrophication and its most severe impact is the establishment of dead zones, caused by a decrease in dissolved oxygen in bottom waters (Diaz and Rosenberg

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2008). Higher eutrophication diminishes the resilience of the Baltic Sea ecosystem, making it more vulnerable and heightening the likelihood of future disturbances causing a flip, or a regime shift (Folke 2006). "Substantially greater reductions in emissions" are required to avoid further degradation of the state of the Baltic Sea (MVB 2005: p. 31). Political cooperation is regarded as crucial for progress. Since 2004, eight of the nine countries around the Baltic Sea basin have become members of the EU, which may facilitate cooperation Larsson (2005).

Two parts of the food system combine to account for the lion's share of eutrophication in the Baltic Sea. The main source of nutrient emissions is food production in agriculture, followed by emissions from municipal wastewater treatment plants and private households (HELCOM 2005). Addressing different aspects of the food system—ranging from what is produced and consumed to where and how this is done—is therefore important for the environment in general, and with regard to the eutrophication of the Baltic Sea in particular. The main factor responsible for the increase in the load of nitrogen and phosphorus from agriculture to the Baltic Sea in recent decades is the specialization of agriculture, and the separation between crop and animal production (Granstedt 2000). One consequence of this is higher use of chemical fertilizers, and imported feed concentrates with high nitrogen content. Another is clusters of farms with high animal densities and large surpluses of plant nutrients in specific regions. More nutrients are concentrated on farms than can be used in on-farm crop production. As manures are too costly to be transported over large distances, there is a risk of surplus nutrients leaking into the surrounding environment. According to the Swedish Environmental Advisory Council, "drastic emission reductions and changes in our lifestyles" are required to avoid further degradation of the state of the Baltic Sea (MVB 2005: p. 26). One such lifestyle change that would reduce nitrogen emissions is if people consume more vegetables instead of meat. A similar message is conveyed in a government commission report on sustainable consumption (SOU 2005) while the Stockholm County Council takes the argument a step further in its S.M.A.R.T. recommendations (CTN 2001, 2008, 2015) by saying that local organic food is good for the consumer and for the environment.

Sustainable development is an important goal for Sweden, as it is for the UN, e.g. the Sustainable Development Goals (SEPA 2016;

UN 2015). The Swedish parliament has adopted a number of environmental objectives, several of which are directly or indirectly related to agriculture and rural development or the Baltic Sea.[1] Some of its goals include: 20% organic acreage; ecologically, economically and socially sustainable food production; and ecologically, economically and socially sustainable rural development. Several of these goals coincide with services that a growing organic agriculture sector is expected to deliver, e.g. environmentally friendly food production, thriving rural areas with small-scale farms and increased biodiversity (Milestad 2003). Thus, different aspects of sustainability need to be addressed for agriculture to be sustainable. Rockström et al. (2009) have attempted to quantify the safe biophysical boundaries outside which the ecosphere cannot function in a stable state. They have identified nine biophysical "planetary boundaries": climate change, ocean acidification, stratospheric ozone depletion, nitrogen and phosphorus cycles, global freshwater use, change in land use, biodiversity loss, atmospheric aerosol loading and chemical pollution. In the long run, humanity must stay within these boundaries to avoid unacceptable environmental change. In their original publication, Rockström et al. (2009) identify three boundaries as having been already transgressed—climate change, biodiversity loss and the nitrogen cycle. In a later, updated version, Steffen et al. (2015) added the phosphorus cycle and change in land use (referred to as "land-system change"). Thus, in all, four planetary boundaries (nitrogen and phosphorus cycles counted as one) are no longer within a "safe operating space".

Six of these boundaries—climate change, the nitrogen and phosphorus cycles, global freshwater use, land-system change, biosphere integrity (genetic diversity) and chemical pollution/novel entities—are clearly related to agriculture. Atmospheric aerosol loading and ocean acidification are related to agriculture production to a limited extent, while stratospheric ozone depletion is not related to agriculture.

In this chapter, the main focus is on eutrophication, i.e., the nitrogen and phosphorus cycles. Together with other countries in the region, Sweden has agreed, through HELCOM,[2] to participate in an effort to reduce the emission of nutrients in the marine ecosystems to sustainable levels (HELCOM 2007b). This goal has not been reached, but there has been a considerable reduction, particularly in emissions from sewage treatment plants and other point sources. Reduction in emissions from non-point sources has been achieved in eastern Germany, Poland and the Baltic states since their independence from the Soviet Union, a period in which these countries reduced animal production and manure and chemical fertilizer use (HELCOM 2003). With the admission of Poland and the three Baltic states into the EU, however, there is a risk of these gains being reversed. Agricultural production is expected to rise as a consequence of the expansion of EU, and unless steps are taken to reduce nutrient emissions, there could be an increase in the load from new EU members (HELCOM 2004a, 2007a, 2011). In the words of HELCOM (2011: 86): "A worst-case scenario is that the amounts of nitrogen and phosphorus leaching into the Baltic Sea will increase". The latest available HELCOM report reveals reduced inputs of both nitrogen and phosphorus to the Baltic Sea. However, the inputs of nutrients are still higher than the maximum allowable inputs (MAI) (HELCOM 2015: 110). Today, nearly the entire Baltic Sea is considered to be affected by eutrophication. "This indicates that despite measures taken to reduce external inputs of nitrogen and phosphorus to the sea, good water quality status has not yet been reached" (HELCOM 2015: 12).

A production technique called Ecological Recycling Agriculture (ERA), which could be described as a stricter form of organic agriculture, has an important role in this chapter (see Fig. 2.1). The main difference between ERA and conventional agriculture is that there is more recycling of nutrients in ERA. Diaz and Rosenberg (2008) highlight the need for new agricultural methods that close the nutrient cycle. ERA is an example of a method for closing the nutrient loop. Other agricultural techniques may perform better or worse than ERA in terms of production levels or emissions to the environment, but there are not examined here.

## The Environmental Pressure from Agriculture

Agricultural production is affecting the environment in various ways, ranging from changes in the rural landscape to leaching of eutrophying nutrients and pesticides. Many of these effects are neither unavoidable

Ecological Recycling Agriculture (ERA) is a form of organic agriculture. Organic agriculture is usually defined according to principles of health, ecology, fairness and care (IFOAM, 2008; KRAV, 2016). The principle of ecology, which is the most relevant here, includes banning chemical pesticides, artificial fertilisers and genetically modified organisms (GMOs). KRAV[ii] develops organic standards in Sweden and is an active member of the International Federation of Organic Farming Movements (IFOAM). KRAV emphasises the importance of nutrient recycling and aims to maximise feed production within animal farms. For agricultural production to be sustainable, the nutrient cycle needs to be closed (e.g., Diaz & Rosenberg, 2008) and ERA is a form of organic agriculture that includes stricter rules on animal feed production on the farm.

An ERA farm is defined as an ecological (organic) farm (or farms working in close cooperation as one farm unit) that does not use artificial fertilisers and pesticides, with a high rate of recycling of nutrients based on organic, integrated crop and animal production and an external feed rate of <0.15, i.e., less than 15% of the feed may be imported from outside the farm. The absence of these or similar restrictions can result in organically certified production that nevertheless causes substantial nutrient losses.

**Fig. 2.1** Principles of ecological recycling agriculture. *Source* Adapted from Larsson and Granstedt (2010). www.krav.se

nor irreversible, but are determined by a number of controllable factors, including what is produced and how this is done. This chapter focuses on the effects of food production on the Baltic Sea. Poland is the largest contributor of nitrogen and phosphorus flows to the Baltic Sea. However, when expressed as emissions per capita, the Swedish contribution is considerably higher for nitrogen and marginally higher for phosphorus than Poland. Finnish per capita emissions are even higher (see Sect. 2.1; Larsson 2005; Table 1 in Larsson and Granstedt 2010).

Agricultural production is responsible for around 50% of the nutrients deposited in the Baltic Sea by surrounding countries (HELCOM 2007b). The input of nitrogen, in the form of artificial fertilizers, to agriculture increased drastically during the second half of the twentieth century. However, only a third of this nitrogen input is usefully exported from the system in the form of food products such as milk, meat and bread grain. If meat production is considered in isolation, the losses are even more substantial. A low surplus of nitrogen implies a lower risk of nitrogen loss to watercourses per hectare (Hoffman and Wivstrand 2015). On average, organic farms have a lower surplus of nitrogen per hectare than conventional farms. However, in relation to food production, nitrogen losses to water "are generally similar to or higher in organic production with large variation depending on system and management" (Hoffman and Wivstrand 2015, p. 17). Thus, a general conclusion cannot be drawn regarding emissions of nutrients from organic agriculture (Hoffman et al. 2014).

The extent of nutrient leakage is also affected by the geographical division of food production (Granstedt 2000). The concentration of animal production is high in southern Sweden and low in the rest of the country. Extensive imports of concentrated feed (Deutsch 2004), feed bought from specialist crop farms and additional use of artificial fertilizer, all contribute to a surplus of plant nutrients in the form of manure in southern Sweden. This region also has the most favourable conditions for the leaching of nutrients with respect to soil texture and climate (Fig. 2 in Larsson and Granstedt 2010). One solution would be for Swedish agriculture to have fewer animals, particularly in the southern parts Larsson (2006). This issue is discussed further in Sect. 4. Furthermore, forage could be used to a larger extent instead of grain-based concentrate. Today 40% of the grain produced in Sweden is consumed by humans, while the remaining 60% is used in animal feed (Jordbruksverket 2014). Thus, the existing system requires higher grain production than an extensive production system with grazing animals Larsson (2006).

## Socio-economic Aspects of a Sustainable Food System

The environmental effects of food production and their mitigation through agricultural reforms also have a socio-economic impact. Two examples include the European Nitrogen Assessment and a report from the research network BalticSTERN. A study by BalticSTERN found that the willingness of the population around the Baltic Sea to pay for an improved marine environment amounts to €4 billion per year (Ahtiainen et al. 2012). This can be compared with the social costs of nitrogen fertilization in EU agriculture, which have been estimated by the European Nitrogen Assessment to amount to €20–150 billion per year. The annual benefits of nitrogen fertilization for EU27[3] farmers are in the range €10–80 billion (Brink et al. 2011). HELCOM aims for a Baltic Sea with diverse biological components that function in balance and support a wide range of sustainable human economic and social activities by 2021 (HELCOM 2007a).

## Sustainable Agriculture

Different concepts have been used interchangeably to describe sustainable agriculture. According to Pretty (2000, p. 26), "the basic challenge for sustainable agriculture is to make better use of available physical and human resources. This can be done by minimizing the use of external inputs, by regenerating internal resources more effectively, or by combinations of both". In addition to the IFOAM (2008) and KRAV (2016) standards for organic agriculture, ERA specifies a spatial dimension and can be described as a form of local organic agriculture. For the purposes of this chapter, sustainable agriculture includes the following attributes: low nutrient losses (i.e., recycling), minimal harm to biodiversity (no pesticides), production of a food basket that consumers demand and contribution to self-reliance/local development. ERA is only one interpretation of sustainable agriculture and is geared towards the first two attributes listed above.

Sustainable agriculture is only one of the aspects of a sustainable food system, which must also include sustainable production, processing, distribution and consumption. It has environmental, social and economic dimensions. The focus of this chapter is the environmental and economic sustainability of the food system.

## Defining a Sustainable Food System in the Baltic Sea Region

The Baltic Sea drainage area is densely populated and the Baltic Sea is a very sensitive and environmentally exposed marine ecosystem. A sustainable food system for this region has to acknowledge specific problems that might be of less relevance to other marine environments.

Sustainable agriculture is usually defined more broadly than organic agriculture. Sustainable agriculture "does not mean ruling out any technologies or practices on ideological grounds. If a technology works to improve productivity for farmers and does not cause undue harm to the environment, then it is likely to have some sustainability benefits" (Pretty 2008: 451). Organic farming focuses on the absence of inputs such as chemical fertilizers and pesticides and supports nutrient cycling through animal feed self-sufficiency ratios and limiting the number of animals per hectare (KRAV 2016). However, it is possible that certified organic farming along the coast to the Baltic Sea can result in substantial nutrient losses, causing eutrophication (Hoffman et al. 2014). "For agricultural systems in general, methods need to be developed that close the nutrient cycle from soil to crop and back to agricultural soil" (Diaz and Rosenberg 2008: 926). This is certainly true for the Baltic Sea and other regions where reducing eutrophication is an important social goal. One production method that addresses nutrient losses is ERA (Granstedt 2000; Granstedt et al. 2008; Larsson and Granstedt 2010), which covers all the environmental principles of organic farming and adds quantitative goals for nutrient losses (see Fig. 2.1).

In addition to sustainable agriculture, this chapter addresses the food system, from a wider perspective. Dahlberg (1993: 75) argues that a regenerative (i.e., sustainable) food system includes "not only production, but processing, distribution, use, recycling, and waste disposal". The scientific journal *Agroecology and Sustainable Food Systems* "focuses on the changes that need to occur in the design and management of our food systems in order to balance natural resource use and environmental protection with the needs of production, economic viability, and the social well-being of all people" (Taylor and Francis Online 2015). A sustainable food system therefore encompasses social, economic and environmental aspects of food and agriculture, and sustainable production is one of several aspects that are considered. Kloppenburg et al. (2000) have identified a set of attributes of a sustainable food system. Several of these are related to lifestyle, including health and consumption, e.g. "In a sustainable food system the production and consumption of food would preserve and enhance the health and well-being of both workers and eaters" (Kloppenburg et al. 2000: 183).

## Aim and Research Questions: Aspects of Sustainable Food Systems

The aim of this chapter is to examine different aspects of a sustainable food system, with the objective of minimizing eutrophying emissions of nutrients to the Baltic Sea. For this purpose, economic sustainability has been studied across the whole food chain, while ecological sustainability has mainly been considered at the level of the Baltic Sea. This chapter tackles three research questions in order to study the problems relating to a sustainable food system in the Baltic Sea region:

1. What environmental effects (primarily eutrophication) are expected from a large-scale change towards ERA?
2. What governance strategies are effective in supporting ecosystem management and sustainable food systems?
3. What socio-economic effects (food production, household expenditure and firm-level income) are expected from a transition towards organic production/ERA?

Other important questions include: What are the environmental effects of today's typical agriculture and those of ERA? What would be the effects of different large-scale transformations of agricultural production in the Baltic Sea region on the environment and on output? What are the environmental effects of different food baskets and of similar food baskets produced with different techniques? The environmental impact of an average food basket that is mostly produced and processed far away is compared with those of a locally produced and processed food basket. The effects of food transport and of locally produced food have previously been studied by Carlsson-Kanyama (1999) and Pretty et al. (2005), among others.

There is a long tradition of studying the socio-economic consequences of agriculture on the Baltic Sea environment. The external costs of food production and other economic aspects of the eutrophication of the Baltic Sea have been studied by Gren (2001), Gren and Folmer (2003) and Ahtiainen et al. (2012), among others. Collaboration in combating eutrophication in the Baltic Sea has been studied by Elofsson (2007) and others. This chapter builds on this tradition in different ways: it discusses the various effects of measures on production and employment, the economic impacts of different measures from the perspective of households and producers, and the importance of collaboration in combating the eutrophication of the Baltic Sea.

The relevant theories are presented below, followed by an overview of the methods used. The results are presented and discussed from different perspectives, before the concluding remarks.

## Methodology

This chapter covers the environmental and economic aspects of sustainable food systems. The journal *Agroecology and Sustainable Food Systems* states, as its aims and scope: "Rather than focus on separate disciplinary components of agriculture and food systems, this journal uses an interdisciplinary approach to food production as one process in a complex landscape of agricultural production, conservation, and human interaction" (Taylor and Francis Online 2015).

A range of methods have been used to study the different aspects of local organic food production and consumption. These are described in greater detail in the BERAS, GEMCONBIO and HealthyGrowth background reports.[4] The main focus of the BERAS project, for example, was to study the environmental effects of ERA in comparison with those of conventional food production. Surplus and emissions of nitrogen and of phosphorus compounds in the agriculture-society system were quantified. Most of this work was done in Sweden and Finland and, to a lesser extent, in other EU countries around the Baltic Sea (Larsson and Gransted 2010; Larsson et al. 2012). Social and economic consequences were also evaluated. The collection of economic primary data (Larsson et al. 2016) and households (Larsson et al. 2012) was limited to firms and households in Sweden.

# Results

## What Environmental Effects Are Expected from a Large-Scale Change Towards Ecological Recycling Agriculture?

*Results from nutrient balance studies in the Baltic Sea region:* The ERA system studied in this chapter showed lower levels of nutrient surplus than conventional production. Among the 12 Swedish ERA farms studied in Larsson and Gransted (2010), the average nitrogen surplus was 36 kg N per hectare per year in 2002–2004. The average for Swedish agriculture was 79 kg per hectare per year in 2000–2002 (Table 2.1). The average nitrogen and phosphorus surplus in average agriculture in all countries in the thesis was 56 and 11 kg per hectare, respectively, in 2000. The average nitrogen surplus observed on the selected ERA farms was 32% lower, i.e., 38 kg per hectare. The phosphorus surplus was completely eliminated in ERA agriculture, and there was a net deficit of 1 kg/ha per year. However, since production per hectare was higher using conventional methods, the difference in nutrient surplus was smaller when surplus nitrogen and phosphorus was expressed per unit output of food (animal and crop production) rather than per hectare (see Sect. 2.3.2). Nitrogen and phosphorus surpluses and calculated ammonia ($NH_4$) losses for all countries covered by the study and the ERA farms are presented in Table 2.1.

Comparing the nutrient load to the Baltic Sea today [Table 3, latest available figures are from 2010 (HELCOM 2015)] and that in 2000 [Table 1 in Larsson and Granstedt (2010), figures are from HELCOM (2005)], the increase in total load of nitrogen from the countries under study, Russia included, was 863 kt per year. Excluding Russia, the total load from the listed countries marginally decreased. Sweden, Finland, Estonia and Demark now produce lower loads of nitrogen, while Latvia, Lithuania, Poland, Germany and Russia have increased their loads. The total load of phosphorus from the countries is lower today (36.1 vs 41.2 kt/yr, including Russia). Sweden, Finland, Estonia, Poland and Germany now have lower loads and Latvia, Lithuania, Denmark and Russia have increased their loads over time.

**Table 2.1** Arable land (million hectare, Mha), total load (according to calculations from HELCOM 2005) and calculated total farm-gate surplus of nitrogen (N) and phosphorus (P) and ammonia ($NH_4$) losses (40%), expressed per unit area (kg/ha) and as kilotonnes per year (kt/yr) in average agriculture and for ERA farms

| HELCOM | | | | Average agriculture | | | | | | ERA agriculture | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Arable land[a] | | Loads year 2010[b] | | N surplus | | P surplus | | $NH_4$ loss | | N surplus | | P surplus | | $NH_4$ loss | |
| | Mha | N kt/yr | P kt/yr | kg/ha | kt/yr | kg/ha | kt/yr | kg/ha | kt/yr | kg/ha | kt/yr | kg/ha | kt/yr | kg/ha | kt/yr |
| Sweden | 2.7 | 119 | 3.6 | 79 | 184 | 3 | 8.1 | 22 | 58 | 36 | 97 | −2 | −5.4 | 21 | 57 |
| Finland | 2.4 | 72 | 3.0 | 75 | 179 | 7 | 16.7 | 14 | 33 | 38 | 91 | 3 | 7.2 | 18 | 43 |
| Est/Lat/Lit | 7.5 | 174 | 6.1 | 19 | 141 | 3 | 21.4 | 16 | 117 | 41 | 308 | −1 | −3.8 | 12 | 99 |
| Poland | 14.2 | 302 | 14.8 | 57 | 812 | 19 | 270.7 | 15 | 217 | 32 | 456 | −2 | −28.5 | 16 | 233 |
| Germany | 2.1 | 38 | 0.6 | 74 | 152 | −2 | −4.1 | 9 | 19 | 16 | 33 | −3 | −6.2 | 6 | 13 |
| Denmark | 2.1 | 57 | 1.8 | 129 | 268 | 8 | 16.6 | 54 | 112 | 87 | 181 | 5 | 9.3 | 49 | 103 |
| Total[c] | 31.0 | 762 | 29.9 | 56 | 1736 | 11 | 329.4 | 18 | 556 | 38 | 1165 | −1 | −27.3 | 18 | 548 |

[a]Land in the Baltic Sea drainage area only

[b]HELCOM (2015, Table 4.1a)

[c]Loads from Russia are 101 kt N/yr and 6.2 kt P/yr. Total loads including Russia are 863 kt N/yr and 36.1 kt P/yr (HELCOM 2015)

*Source* Adapted from Tables 1 and 2 in Larsson and Granstedt (2010) and HELCOM (2015). EST/Lat/Lit = Estonia–Latvia–Lithuania

ERA differs from organic agriculture in one important aspect, namely that animal and plant production are integrated. In this type of system, it is possible to make efficient use of plant nutrients in manure to reduce the nutrient surplus. The need for external nitrogen in such a system is much lower. Note that it is not necessary for each farm to have both animal and vegetable production, as farms within regions can cooperate. However, it is important to use manure in an efficient manner to avoid associated environmental problems. Inappropriate storage and spreading of manure results in a loss of nutrients to water and air. The benefits of using organic fertilizers instead of chemical fertilizers as a means to reduce eutrophication have been questioned (Kirchmann and Bergström 2001; Larsson 2005; Kirchmann et al. 2001). There may be difficulties in applying the right quantity of organic fertilizers and field trials have shown that chemical fertilizers can cause less leaching of nutrients and thus less eutrophication. Then again, it can be argued that in an agricultural system with regional specialization and a need for chemical fertilizers in crop production, there will be a surplus of nutrients in regions specializing in animal production and this surplus is rarely used efficiently in production.

*Scaling up the results—scenarios on emissions:* Larsson and Granstedt (2010) developed three scenarios based on results from 42 farms in eight EU countries, which are presented in Table 2.1. In the "conventional scenario" (Fig. 2.2), agriculture in Poland and the Baltic countries convert to the production methods currently used in Sweden. This scenario is supported by the present system of EU subsidies. ERA Scenarios 1 and 2 assume the conversion of agriculture across the Baltic Sea drainage area to less intensive local organic agriculture. To echo the conclusions from the Millennium Ecosystem Assessment (2005: 1), these scenarios would "involve significant changes in policies, institutions, and practices that are not currently under way".

In the conventional scenario, nitrogen and phosphorus surplus increase by 58% and 18% per hectare, respectively (Fig. 2.2). If Poland and the Baltic states were to intensify their agriculture according to Danish standards, nutrient loads would increase further (HELCOM 2007b). In this scenario, referred to as "Business as usual in Agriculture", phosphorus loads to the Baltic Proper double and nitrogen loads increase by 70% (HELCOM 2007b). This expected increase may be more than

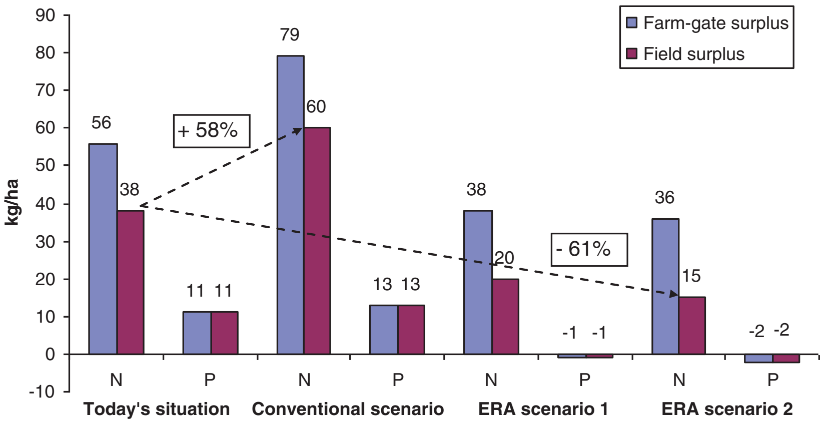


**Fig. 2.2** Surplus of nitrogen and phosphorus, kg/ha & yr, in farm-gate and field balances calculated for four alternative governance regimes: Today's agriculture situation (Today's situation); a scenario where agriculture in Poland and the Baltic countries converts to conventional agriculture similar to that in Sweden (Conventional scenario); all agriculture in the Baltic Sea drainage area converts to ERA as practised in the respective countries (ERA Scenario 1); all agriculture converts to ERA as practised in Sweden (ERA Scenario 2). Field surplus equals farm-gate surplus minus ammonia ($NH_4$) losses, see Table 2.1. *Source* Adapted from Larsson and Granstedt (2010)

what the already stressed ecosystem of the Baltic Sea can cope with (MVB 2005). The consequences of ERA Scenario 1, where all agriculture in the Baltic drainage area converts to ERA, following the techniques used in each country, are very different. Calculations indicate a decrease in the nitrogen surplus of 47% in that case (Fig. 2.2). If all agriculture in the Baltic Sea region were to be converted to ERA as practised in Sweden (ERA Scenario 2), there would be a 61% reduction in the nitrogen surplus. In the two ERA scenarios, there is a negative phosphorus surplus, which would significantly reduce the phosphorus load to the Baltic Sea.

## What Governance Strategies Are Effective in Supporting Ecosystem Management and Sustainable Food Systems?

*The importance of consumption—Swedish case studies:* In addition to production methods, consumption patterns also determine environmental impact. A Swedish government commission report (SOU 2005) proposes increasing the shares of vegetables and local organic food to reduce global warming from the food chain. Larsson et al. (2012) provide results that partly support this proposal. They compared four scenarios:

The environmental impact of conventional Swedish food production applied to an average food basket (Scenario 1) was compared with food produced with ERA methods (Scenario 2); food produced with ERA methods and processed locally (Scenario 3); and an alternative food basket with less meat and more vegetables produced with ERA methods and processed locally (Scenario 4).

Scenario 4 resulted in slightly more than a third of the nitrogen emissions attributed to average Swedish consumption. The nitrogen emissions per hectare in Scenario 4 were higher than in Scenario 2 (average food basket produced with ERA methods), but the total or per capita nitrogen emissions were lower (Larsson et al. 2012). This is due to the differences in the food baskets and the resulting difference in acreage needed for food production. A reference group studied in a household survey in Scenario 4 consumed more vegetables (100% more), less meat (75% less) and substantially more local and organic food than the average Swedish consumer. The share of local and organic food was 33% for the families in this survey. Among "real food" purchases, i.e., excluding sugar, sweets, beverages, etc., 73% was organic, compared with 2% for the average Swede. Today, consumption of organic food has increased, but it is not close to 73% (Larsson et al. 2016). These results agree well with findings from the Environmental Advisory Council that a diet consisting of two-thirds animal products results in fourfold more nitrogen emissions from agriculture into water and air than a fully vegetarian diet (MVB 2005).

If the average Swedish food profile was similar to that of the group in the household survey (Larsson et al. 2012), i.e., more vegetables and less meat, the area currently used for agriculture in Sweden would be more than sufficient. Simply replacing conventional production with ERA without changing consumption patterns would require an additional (and unrealistic) 2.3 million hectare of arable land, an increase of 90%. This larger areal requirement is due to the lower yield per hectare on organic farms and the larger share of ruminant meat (70%, compared with 30% in conventional production), which requires more arable land for feed production compared with pork or poultry (Larsson et al. 2012). On the other hand, conventional production relies to a larger extent on imports, e.g. soy products from agricultural production in other countries. The difference in required area for food production is therefore smaller than might be expected. While this would also lead to a reduction in phosphorus emissions, additional studies on the link between lower surplus and real losses on farm level are needed to provide a quantitative estimate.

*Other environmental effects of different governance strategies*: Moving towards ERA production and a change in diets could also result in gains with respect to global warming. However, local processing and distribution result in less significant gains.

Scenario 2 (food produced with ERA methods) resulted in a 10% reduction in the Global Warming Potential (GWP) and Scenario 4 (an alternative food basket with less meat and more vegetables produced with ERA methods and processed locally) led to a 40% reduction in GWP compared with Scenario 1 (conventional Swedish food production of an average food basket) (Larsson et al. 2012). The results indicate potential environmental gains from local food production and consumption due to reduced transportation, as reported in previous studies by Carlsson-Kanyama (1999) and Pretty et al. (2005). Local processing and distribution (Scenario 3) resulted in additional GWP reductions, compared with Scenario 2. One explanation for the better environmental performance of Scenario 4 is the smaller share of meat in the food profile. Meat production is generally more energy-intensive than vegetable production.

## What Socio-Economic Effects Are Expected from a Transition Towards Ecological Recycling Agriculture?

*What are the expected economic consequences for households?* The environmental gains from local organic production are promising, but the food produced is more expensive. According to the household survey carried out in Larsson et al. (2012), a food basket high in organic and locally produced food was 24% more expensive than the Swedish average basket. This may obstruct large-scale expansion of environmentally friendly production and consumption. It may be difficult to convince consumers to increase food expenditure for the sake of the environment, and the consumption pattern with high levels of eco-local food found in the case study (Larsson et al. 2012) is unlikely to be found in many places. However, 24% higher food prices that the consumers in the reference group faced (Larsson et al. 2012) may be misleading from a societal perspective. The increased costs are associated with reduced environmental effects compared with conventional food production and consumption, where environmental effects are largely externalized. In other words, it can be argued that local and organic agriculture such as ERA also contributes to economic sustainability at the regional level.

Although local and organic food is more expensive according to the results in Larsson et al. (2012), the demand has increased over time (Larsson et al. 2016).

*What are the expected effects on production levels?* A large-scale transition from conventional to ERA production in Sweden would result in a 22% reduction in animal production and a 28% reduction in crop production. Annual crop and animal production have also been calculated for conventional and ERA scenarios in Poland and the Baltic states (Fig. 2.3). Neither ERA nor conventional production is currently optimized in Poland or the Baltic states. For conventional production, this was expressed by increasing production in the conventional scenario, where production in Poland and the Baltic states was changed in accordance with conventional, mainly industrial, agriculture in Sweden. The conventional scenario led to an increase in crop production at around 30%, and an increase in animal production at around 40%.

For the two ERA scenarios, production estimates were very different. ERA 1, where all production was altered according to ERA as practised in each country, resulted in 15% lower crop production and 40% lower animal production. This lower output is explained by extensive production with low productivity on ERA farms in Poland and the Baltic countries. The differences between scenarios ERA 1 and ERA 2 shown in Fig. 2.3 illustrate the potential production gains if local organic agriculture in Poland and the Baltic states were to introduce the production methods practised in Sweden. Should ERA as practised in Sweden be introduced on a large-scale in the Baltic Sea drainage area, production

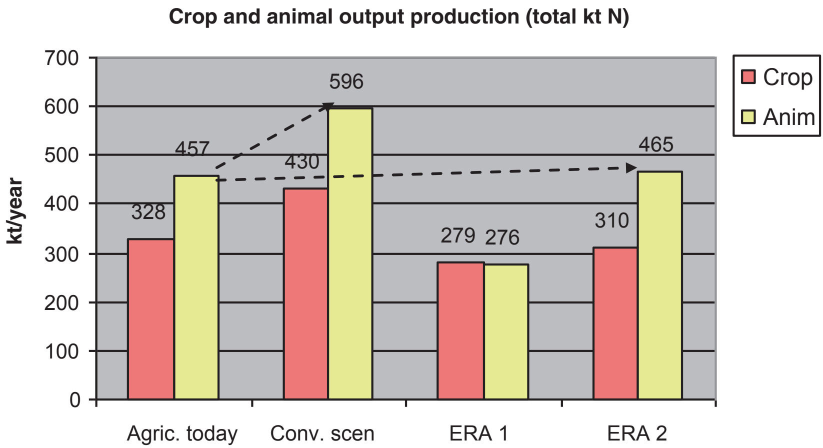


**Fig. 2.3** Crop and animal production in the Conventional (Conv.) scenario and two alternative ERA scenarios (1, 2), which resulted in increased, reduced and (close to) unchanged agricultural production, respectively, compared with today's average agriculture. *Source* Adapted from Larsson and Granstedt (2010)

figures would essentially remain at current levels (ERA 2 scenario). In fact, if conversion to efficient ERA methods were restricted to Poland and the Baltic states, output in the region would increase, while nutrient emissions would decrease. This would obviously require a concerted governance effort.

Compared with scenarios carried out by SEPA (2008), the large-scale introduction of ERA in Sweden in Larsson and Granstedt (2010) showed relatively low production losses and relatively high reductions in emissions. Assuming that the recycling principles of ERA are followed, animal production needs to be decreased in southern Sweden, while a corresponding increase is required in central Sweden.

The results from Larsson and Granstedt (2010) indicate that sustainable governance of the Baltic Sea cannot be achieved while simultaneously maximizing agriculture production in the surrounding countries. A clear choice has to be made between these desired outcomes. Production and nutrient emissions differed substantially depending on the selected agriculture regime. The differences in outcomes were most evident in Poland and the Baltic states. Historically, newer EU members have shown relatively modest food output, accompanied by a fairly low nutrient surplus.

*Other socio-economic effects:* Larsson and Granstedt (2010) discuss economic aspects of introducing ERA on a large-scale, including the effect on income, production levels, employment and reduction costs. The results indicate that benefits outweigh costs at firm level, but that consumers face increased prices. A group of local organic producers in Sweden referred to in Larsson and Granstedt (2010) reported their economic situation to be satisfactory and organic production was often stated to be at least as profitable as conventional production. Production costs were higher and yields lower, but this was compensated for by the price premium received and EU subsidies under the CAP programme paid to organic producers.

Reduction in costs depends on several factors, including demand for food and bioenergy, the extent of measures undertaken and implementation time (Larsson and Granstedt 2010). With a lot of capital bound to present production, as in Sweden, the cost will increase with shorter implementation time. If major measures in the agricultural sector are undertaken, it could prove difficult for other sectors to absorb labour and other factors of production. Finally, if a measure can be targeted at areas with low productivity but high leaching, or if affected areas can be used for alternative production, costs will be reduced.

According to Larsson and Granstedt (2010), the main costs of implementing ERA in Sweden and other old EU member states are probably in the form of farm level infrastructure investments, lower yields and, to some extent, provision of training and advisory services. Social costs for conversion towards local organic production have not yet been estimated. Large-scale conversion might cost more than is gained, but this question is beyond the scope of this chapter. However, over the years the costs of implementing other reduction measures have been estimated and could serve as benchmarks. Gren (2001) has identified three main components of a cost-efficient mix of measures to reduce nitrogen emissions: measures aimed at agriculture; extending the capacity of municipal wastewater treatment plants, and the (re-) creation of wetlands as nitrogen traps. Turner et al. (1999) quantified the marginal costs of these measures for different countries. Several reduction measures have been implemented ever since. This has increased the estimated reduction costs of additional measures. The marginal cost of 1 kg nitrogen reduction in agriculture was SEK 22–42 according to Turner et al.

(1999), whereas more recent estimates by the Swedish Environmental Protection Agency and the Swedish Board of Agriculture are higher, in the range of SEK 10–5800.

According to SEPA (2008), there are cost-efficient measures to carry out water treatment. However, the potential reduction in nitrogen emissions is limited. Many investments in reducing Swedish point-source emissions have already been made and investments to further reduce phosphorus emissions in wastewater treatment plants are not justified (SEPA 2008; Larsson and Granstedt 2010). Other measures thus become more interesting and most reductions have to be made in agriculture.

Elofsson (2003) compared the cost efficiency of different measures to reduce the agricultural load of nitrogen and phosphorus to the Baltic Sea. Reduction in the use of chemical fertilizers was the most important measure identified, followed by changes in land use, primarily increasing the area of catch crops. Using no chemical fertilizers and a change in land use in terms of less intensive production, for example, by introducing ERA production, may or may not be as cost-efficient as the measures examined by Elofsson (2003), but this has not been examined here. Economic aspects of agriculture and the Baltic Sea are discussed below.

## Discussion

The aim of this chapter is to examine different aspects of a sustainable food system, with the overall objective of minimizing eutrophying emissions of nutrients to the Baltic Sea. The following sections discuss some of these aspects: socio-economic consequences (Sect. 3.1); and cooperation at different levels (Sect. 3.2).

### Costs and Benefits of Measures Targeting Production

In order to reach the targets set out by HELCOM, costly, ambitious and perhaps unrealistic measures are needed (SEPA 2008; Larsson and

Granstedt 2010). SEPA (2008) presents three hypothetical scenarios of large-scale reduction of agricultural production in southern Sweden (see Table 5 in Larsson and Granstedt 2010). Compared with other measures, e.g. those presented in Sect. 2.1 (see also Table 7 in Larsson and Granstedt 2010), these are extremely ambitious and are the only way to match the reduction in nutrient emissions from large-scale conversion to ERA. In the three SEPA scenarios, between 230,000 and 940,000 ha of productive land in coastal zones in southern Sweden are turned fallow. This results in a reduction in nitrogen emissions to the Baltic Sea of 3000–8500 tonnes per year. Large-scale conversion to ERA methods in Sweden are estimated in this chapter to result in a reduction of 15,000–16,000 tonnes worth of nitrogen emissions (Larsson and Granstedt 2010). Turning large areas of productive agricultural land fallow and the large-scale conversion to ERA will both result in substantial losses in food production for Sweden (see Fig. 5 and Table 5 in Larsson and Granstedt 2010).

However, on a regional scale, food production based on ERA techniques could reduce nutrient emissions in comparison with conventional production methods without substantially reducing food production in the Baltic Sea region (Larsson and Granstedt 2010). While this sounds promising, the conversion to ERA requires large investments in new production facilities, mainly at the farm level. The cost of these investments has not been estimated here. However, there are estimates of other reduction measures that can serve as benchmarks. Even though some of the costs referred to are a few years old, they are indicative of the magnitude of expected reduction costs.

*Measures to reduce nutrient emissions:* A cost-efficient mix of measures, i.e., methods resulting in the largest reduction per krona invested, aimed at agriculture, extending the capacity of municipal wastewater treatment plants and the (re)creation of wetlands as nitrogen traps are needed to reduce nitrogen emissions to the Baltic Sea (Gren 2001). Efforts that target agriculture include cultivation of nitrogen-fixing crops and reduced fertilization. These are also cost-efficient ways to reduce phosphorus emissions (Elofsson 2003). ERA involves a transformation of the farming system. However, the measures suggested by Gren (2001) and Elofsson (2003) can be seen as supporting ERA. ERA does not allow chemical fertilizers, and the less intensive crop production in ERA can be viewed as an alternative to catch crops. Together with efforts aimed at wastewater treatment plants and wetlands, etc., ERA or local organic agriculture could help improve the environmental condition of the Baltic Sea (SEPA 2008; Table 1).

The studies referred to above do not measure costs and benefits of a change towards ERA-like farming. The ERA scenarios (Larsson and Granstedt 2010) would reduce the nitrogen surplus in agriculture by 47–61% and the phosphorus surplus by 100%. If the cost of achieving this is less than the estimated costs of other measures, then ERA is a more cost-effective way to reach reduction targets. The estimated cost of reducing nitrogen emissions ranges from 10 to 5800 SEK/kg (Table 7 in Larsson and Granstedt 2010). For phosphorus, the estimated cost ranges from 350 to 30 000 SEK/kg. A detailed analysis of the cost question is beyond the scope of this chapter.

The effect of the three scenarios from SEPA (2008) mentioned above on Swedish food production is substantial. Much of the affected acreage is in highly productive areas in southern Sweden. In SEPA Scenario 2, the production of several crops is more or less reduced by half, and in Scenario 3, large parts of production in southern Sweden cease (see Table 6 in Larsson and Granstedt 2010). Animal production is also expected to change according to the reduced crop production. In comparison, a large-scale transition from conventional to ERA could prove just as costly. The main costs for implementing ERA in Sweden and other old EU member states are probably in the form of lower yields,[5] training and extension services and farm level infrastructure investments. A large-scale transition from conventional to ERA production in Sweden would reduce animal production by 22% and vegetable production by 28%. To fulfil the recycling principles of ERA, animal production needs to be decreased in southern Sweden, while a corresponding increase is required in central Sweden (see Fig. 2a in Larsson and Granstedt 2010). Many farms that specialize in animal production in southern Sweden would thus become obsolete as a result of the change to a system of local food production. Meanwhile, moving animal production from southern Sweden would require large investments in new production units in central Sweden. This would induce additional costs, over and above the loss in production. It is reasonable that these costs, at least to some degree, are borne by society at large and not by individual farmers. These costs must be considered when deciding on future policies (Larsson 2005, 2006).

*Benefits of reduced nutrient emissions:* The cost of reducing nitrogen emissions to the Baltic Sea may be high, but so is the willingness to pay (WTP) for reducing eutrophication to sustainable levels. According to Gren et al. (1997a, b), the combined WTP for a healthier Baltic Sea (SEK31 bn) is twice as high as the cost (SEK15 bn) of reducing nitrogen and phosphorus emissions by 50%. The WTP has been notably stable over time. A more recent study shows that people in the nine countries around the Baltic Sea are willing to pay a total of €4 billion (SEK36.8 bn) per year to improve the marine environment (Ahtiainen et al. 2012). However, these figures should be interpreted with a degree of caution. The WTP may be an overestimate and the cost of reducing the emissions may be an underestimate. Even if these figures prove to be incorrect and the reduction costs prove to be higher than the WTP, from a socio-economic perspective it is worth making an effort to improve the situation in the Baltic Sea. Ahtiainen et al. (2012: 25) argue that although their figures are prone to uncertainties, "they suggest that the benefits of reducing eutrophication in the Baltic Sea may be substantial". The relatively high total WTP indicates that taxpayers expect an increase in welfare if investments are made in measures to reduce eutrophication. This is further emphasized by the figures pertaining to the external costs of agricultural production.

The European Nitrogen Assessment estimates the annual environmental costs in EU agriculture of nitrogen fertilization alone to be €20–150 billion (Brink et al. 2011). The annual benefit of nitrogen fertilization for EU farmers is roughly half this cost. This is in contrast to the findings in Larsson et al. (2012), where a household survey revealed that families with a high share of local and organic food in their food basket faced 24% higher food expenses than the average Swedish family. Contrary to findings by Pretty et al. (2005), less was gained from local processing and distribution than by turning to organic production in the study of ERA farms in Larsson et al. (2012).

Today, taxpayers subsidize production which affects them negatively in some ways and leads to immense external costs. Using scarce resources in this manner is not rational. Taxpayers are willing to give up substantial amounts to address these problems—the average Swedish taxpayer is willing to pay €110 per year to reduce the eutrophication of the Baltic Sea (Ahtiainen et al. 2012). There is thus a savings potential in society for a move towards more environmentally friendly agriculture. For this potential to be realized low emission techniques need to be applied (Brink et al. 2011). If not, the low fertilizing efficiency of nitrogen in manure, in comparison with that of chemical fertilizers, and the high emission factors for ammonia could cause the use of manure nitrogen to result in more harm than good for society.

Taking a broader perspective, efforts aimed at environmentally friendly agriculture look even more attractive than other measures. They contribute to reaching the goals of sustainable food production and sustainable rural development set out by the government. A less eutrophied Baltic Sea could also mean improved opportunities for the fishing industry and tourism, etc. It could improve the well-being of a large number of people, irrespective of whether they make use of the Baltic Sea (Larsson 2005, 2006). The WTP for an improved Baltic Sea environment is one expression of this.

There are also costs associated with changing farming practices, as discussed in Sect. 3.1.1, but not all measures imply additional costs, and some may in fact result in overall savings for society. For example, the artificial lowering of grain prices due to subsidized production stimulate its excessive use for feed (60% of total grain production) and there are few incentives to reduce the loss of nutrients. If the agricultural sector had to deal with all the negative effects it caused, this would be reflected in increased food prices but not necessarily in increased costs for society. Consumers currently pay the full price of food production, in the grocery bill, through their tax bill and through a degraded environment (Larsson 2006; Brink et al. 2011).

## On Regional Cooperation

The substantial WTP among taxpayers for a healthier Baltic Sea and the pay-off in terms of reduced external costs from agro-environmental investments argue in favour of agricultural production reform (Larsson and Gransted 2010; Larsson et al. 2012; Sect. 3.1). The gains are larger if this action is coordinated internationally: "in order to combat eutrophication (especially in the open sea), nutrient reduction measures should be considered jointly for the whole Baltic Sea region" (HELCOM 2005: 15). For solutions to be cost-efficient, investments need to be made where the highest nutrient reduction can be achieved for the money spent, and this requires international cooperation. All countries would benefit from participation in an effort to combat eutrophication but "some countries [including Sweden] will incur substantially larger benefits than others, which may necessitate the implementation of a redistribution scheme of the increase of the net benefits due to cooperation" (Gren and Folmer 2003: 40).

For example, it may be more efficient for the Baltic Sea region if Sweden were to pay Poland to reduce its emissions to a greater extent in order to offset or compensate for a lower reduction in Swedish emissions. This holds true if the same sum buys a larger reduction in Poland than in Sweden (Larsson 2005). This is already happening in practice. For example, private foundations Baltic Sea 2020 in Sweden and John Nurminen Foundation in Finland are financing water treatment in Warsaw, Poland, because this is predicted to result in larger environmental benefits for the Baltic Sea than similar investments in Sweden and Finland.[6]

Far-reaching measures from individual countries, on the other hand, may not have significant effects (Larsson 2012). Elofsson (2007) argues that there is greater uncertainty around unilateral efforts in terms of costs and reductions achieved compared with bilateral measures. For example, a unilateral Swedish conversion towards ERA is not possible without lowering food production due to lack of arable land (Larsson and Granstedt 2010). If conversion is coordinated between countries in the Baltic Sea region, food production can remain stable while reducing nutrient emissions, according to the ERA 2 scenario in Larsson and Granstedt (2010). Measures towards a more ERA-like agriculture at the regional level, especially in Poland and the Baltic states, may be a cost-effective way to combat eutrophication in the Baltic Sea. A win–win solution thus appears possible (Larsson and Granstedt 2010).

# Implications for Policy

## A Policy Window for Sustainable Agriculture

After the recovery of the Russian economy and the entry of Poland and the Baltic states in the EU, agricultural production is likely to increase, and with it, nutrient loads (Larsson and Granstedt 2010). "These trends will be highly dependent on the future agricultural policies of the EU" (HELCOM 2004b, p. 18). The awareness among policymakers that the present policy implemented among the new EU members is unsustainable and needs to be changed is growing, and the expansion of EU could be viewed as a policy window (Kingdon 1995) or a window of opportunity (Olsson et al. 2004). Poland and the three Baltic states are currently regulated by EU environmental legislation, but they also have access to funding through the EU Common Agricultural Policy (CAP). This provides an opportunity for decision-makers to stimulate agricultural production in an efficient and environmentally friendly direction. If this present opportunity is missed, there is a risk that the agricultural sector will be modernized in a less desirable direction from the perspective of the Baltic Sea environment (Larsson and Granstedt 2010). Once a new regime is established, it will be difficult to change things around again.

The new, renegotiated CAP that was agreed on by the EU members in 2013 introduced some changes in terms of general support and support for environmentally friendly production. According to the European Commission, the new CAP is "more equitable and greener" and is "adapted to meet the challenges ahead by being more efficient and contributing to a more competitive and sustainable EU agriculture" (European Commission 2013: 1). Others, including the European Environmental Bureau, the largest environmental NGO in Europe, question the green ambitions of the reformed CAP: "the greening of the (CAP) is on course to end in failure by allowing farmers to secure European funding while not taking measures to protect the environment".[7] In a similar vein, Friends of the Earth argues that the Commission's initial plans "were a positive step towards sustainability in farming. However, the CAP reform process was mostly business as usual, with little real reform. Greening—the idea that direct payments to farmers would have to include strong elements of environmental protection and sustainable agri- and eco-system services—was weakened.”.[8]

Whether by influencing agricultural practice among old or new EU members, advocates of ERA or other forms of more sustainable agriculture have to make their alternative attractive to decision-makers. In the words of Smith (2007: 446), “Performance criteria in niche and regime need to come into some kind of correspondence—translating what works in the niche into something that also works in the regime”. Having demonstrated that alternative forms of agriculture (a niche) work, a common ground is needed for alternative agriculture to link with and influence conventional practices (the regime). There is of course a risk that practices that are flexible enough to work under such different contexts are not particularly sustainable. Moreover, the regime, i.e., conventional agriculture, enjoys an influential position whereas the green niche, ERA/organic agriculture, is far from mainstream and is disputed. Thus, there is a “power relation influencing how socio-technical practices that ‘work’ in the context of the niche are subsequently interpreted, adapted and accommodated within the incumbent regime” (Smith 2007: 447).

## Implications for New EU Members

According to the results presented in Larsson and Granstedt (2010) and Larsson et al. (2012), sustainable governance of the Baltic Sea cannot be achieved with a policy that strives to maximize agriculture production in the surrounding countries. The outcome in terms of production and nutrient emissions will differ substantially depending on the agriculture policy adopted, especially in Poland and the Baltic states (Larsson and Granstedt 2010). Historically, agricultural production in these countries has resulted in a limited surplus/emission of nutrients and relatively low levels of food production. The rural economies in these countries will most likely change following the access to EU subsidies and the internal market (Larsson et al. 2013). From a policy perspective, it is an opportunity that can be exploited by policy entrepreneurs (Kingdon 1995). As a result, the system could move right in the direction of industrialized agriculture, high yields and, consequently, increased nutrient emissions; or it could move towards an agricultural system of environmentally friendly production with higher yields than today, but lower yields than those offered by a move in a conventional direction.

The Swedish Environmental Advisory Council argues that reducing emissions may not be sufficient to restore the Baltic Sea to its state prior to the industrialization of agriculture. The degradation may have gone on for too long, and there may be an excess of nutrients stored in the sediment. If this is true, the Baltic Sea is heading towards a new stability state, or equilibrium. In order to return to its previous state, a necessary, but perhaps insufficient condition is substantial cuts in emissions. These reductions are required in any case in order to avoid further degradation (MVB 2005; Larsson 2005, 2006). The rural economies of Poland and the Baltic states are going through major changes which are influenced not least by EU's CAP, which offers support for both scenarios above.

## Possible Policy Measures Towards a Sustainable Food System

A number of different policy instruments are available to combat the adverse environmental effects of agriculture. Tradable emissions rights for nitrogen and phosphorus are attractive in theory, but less so in practice. Diffuse emissions, such as nitrogen and phosphorus from agriculture, are considered too difficult to control (Collentine 2002). A related tool could be used for animal production or spreading manure (Alkan-Olsson 2004; Larsson and Granstedt 2010). A system of quotas for livestock with reduced quotas in southern Sweden is one possible measure. In central Sweden, increased quotas may be necessary, combined with subsidies, to increase animal production in order to match crop production (Larsson 2006).

However, Larsson et al. (2012) show that in addition to production methods, what is produced and consumed is of interest. Increasing the share of, say, vegetables could be equally important as increasing the share of organically produced food (Larsson et al. 2012). A transition to sustainable agriculture implies a changed production mix. If sustainable agriculture is to become the dominant regime and not just a niche (Smith 2007), the chosen mode of production must be equipped to meet the consumer demand. One of the key issues for Stockholm County Council's S.M.A.R.T. project is to change consumption patterns and to give recommendations for diets that both improve health and reduce environmental impacts (CTN 2001, 2008, 2015). Several of the S.M.A.R.T. recommendations support organic production and ERA, including those aimed at increasing the share of vegetables consumed; increasing the share of organically certified food; choosing meat from among grazing animals, such as lamb; choosing food according to season; and giving preference to local food more often (Larsson 2005, 2006). The importance of food choices for the environment has also been emphasized in Swedish government reports and Södertälje Municipality, south of Stockholm, has turned theory into practice in its policy for public procurement (Ekomatcentrum 2014, 2015). One government commission report (SOU 2004) suggests increasing public procurement of organic food and strengthening domestic science as a subject taught in schools. One measure discussed by the Swedish Environmental Advisory Council (MVB 2005) is to stimulate radical lifestyle changes. This includes consuming more vegetables instead of meat as a way to reduce nitrogen emissions (Larsson 2005, 2006).

Whether the desirable share of public demand for organic food is 25% (SOU 2004), 50% or even 100%, as suggested by Södertälje municipality (Ekomatcentrum 2014; Larsson et al. 2012), is entirely a political discussion. Efforts aimed at local production and processing could be made more attractive if they are framed in terms of public policies or subsidies for local development rather than for environmental benefits. Food basket content and organic production methods are equally important in terms of impact on the environment, and both are more important than local food (Larsson et al. 2012). This should be considered while taking policy decisions.

Smith (2007: 447) asks for a "policy to help nurture green niches and put incumbent regimes under sustainability pressure". The municipal policy on public procurement of organic food in Södertälje, as described above, is an example of such a policy.

While deciding on policy instruments, it is important to evaluate their potential effects. If farmers are hit so hard that, say, all of Swedish or northern European agriculture is threatened, then the proposed solution is not sustainable. The same applies if taxpayers believe that the new agricultural system is too expensive, if production experiences a sharp fall, or if produce becomes so expensive that consumers switch to imported goods. The expansion of EU is a policy window (Olsson et al. 2004)—it creates a choice. If Poland and the Baltic states follow in the footsteps of old EU members in the Baltic Sea region, there is a risk of nutrient emissions increasing by 50–60% (Larsson and Granstedt 2010). If agricultural subsidies are used to steer production towards an environmentally friendly route, this could be avoided. Instead, there is potential for reduced emissions, as well as profitable rationalization of the farming sector (Larsson 2005, 2006).

## Concluding Remarks: A Sustainable Food System in the Baltic Sea Region

According to HELCOM, eutrophication is the main threat to the Baltic Sea environment and agriculture is the main source of nutrients entering the Baltic Sea. A transition towards the low-input recycling system of ERA is one way of reducing emissions from agriculture. Large investments have already been made in the agricultural sectors of and the Baltic states. There is potential for outlining a new policy where sustainable governance of the agricultural sector is coherent with sustainable governance of the Baltic Sea. If the relatively efficient ERA production or other sustainable production methods that are in use in Sweden were to be introduced on a large-scale in Poland and the Baltic states, there is a possibility to reduce the emission of nutrients from agriculture without lowering food production. If, however, the new EU members develop in the direction of conventional Swedish agriculture, there is considerable risk of an increase in nutrient emissions in parallel with increased levels of food production. The calculations in this chapter are conservative in comparison with HELCOM figures. If all Baltic Sea agriculture were to change in line with ERA, nitrogen emissions from agriculture would be reduced by half and phosphorus emissions would be completely eliminated.

The Swedish government and the other contracting parties in HELCOM have environmental and economic incentives to use this opportunity in Poland and the Baltic states. The costs of transformation can be relatively modest, albeit high in absolute terms, with a progressive EU agricultural policy. A similar transformation towards sustainable agriculture in Sweden and other older EU members is likely to be more expensive. In order to be successful and efficient, the measures taken should be coordinated internationally.

The large-scale transformation of agriculture in the Swedish or Baltic Sea region is likely to depend on government intervention, since the alternative food basket examined here is more expensive than the Swedish average. However, the increased cost is somewhat misleading from a socio-economic perspective, as this move will greatly reduce environmental costs. Compared with conventional food production, the environmental costs of ERA-produced food are internalized to a great extent. People's WTP for an improved environment is substantial and several of the more cost-efficient solutions for reducing eutrophication of the Baltic Sea are also steps towards adopting ERA, which will reduce the emission of nutrients compared with conventional agriculture. The aggregate crop production in the Baltic Sea region would marginally decrease and animal production would marginally increase if all production were to change to effective ERA. A broader cost efficiency analysis should take these effects into account. The environmental performance can be improved further with changed food profiles, i.e., the content of food baskets. Local production and processing of food are less important in terms of environmental effects but do have an impact on local rural development.

At the national level, using Sweden as an example, a regional nutrient balance is necessary. Assuming that the recycling principles of ERA are followed, animal production should be reduced in southern Sweden, while a corresponding increase is required in central Sweden. Furthermore, an altered food profile with less meat and more vegetables would facilitate the transition to a sustainable food system.

These should also be applicable to the other countries. If ERA is not coupled with an altered food profile, the demand for agricultural land will increase substantially.

## Notes

1. The parliament has decided on 16 environmental quality objectives. These are: Reduced Climate Impact; Clean Air; Natural Acidification Only; A Non-Toxic Environment; A Protective Ozone Layer; A Safe Radiation Environment; Zero Eutrophication; Flourishing Lakes and Streams; Good Quality Groundwater; A Balanced Marine Environment, Flourishing Coastal Areas and Archipelagos; Thriving Wetlands; Sustainable Forests; A Varied Agricultural Landscape; A Magnificent Mountain Landscape; A Good Built Environment; Biological Diversity. Details can be found at http://www.miljomal.se/sv/Environmental-Objectives-Portal/. 5 May 2016.
2. The Baltic Marine Environment Protection Commission-Helsinki Commission, www.helcom.fi.
3. Croatia became the 28th member of EU in 2013.
4. See www.beras.eu, http://ecologic.eu/1795 and www.healthygrowth.eu.
5. Over time, world market food prices have been volatile. Higher prices increase the alternative cost of measures that lower the yield and make measures that increase the yield more tempting.
6. Baltic Sea 2020, www.balticsea2020.se/.
7. "New study shows CAP reform risks being greenwashed", http://www.eeb.org/index.cfm/news-events/news/new-study-shows-cap-reform-risks-being-greenwashed/. 7 September 2015.
8. "The Common Agricultural Policy", https://www.foeeurope.org/CAP. 7 September 2015.

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# 3

# Costs of Nutrient Management with Technological Development and Climate Change

Ing-Marie Gren

## Introduction

The excessive loads of nutrients, nitrogen and phosphorus in coastal and marine waters are the main sources of eutrophication, which has globally been acknowledged to be a serious environmental problem because of the creation of damages such as increased frequency of harmful algal blooms, sea bottom areas without biological life, cyanobacteria, and decreases in water transparency and populations of commercial fish species (e.g. Gilbert 2007; Heisler et al. 2008). These environmental damages were recognized in the mid-1970s, followed by the implementation of different types of abatement measures directed mainly towards discharges from households and industry into the seas. However, in spite of these measures and the development of new abatement technologies such as nutrient traps in drainage basins, these damages have aggravated

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because of the substantial nutrient loads from land-use activities, such as agriculture. Further degradation is expected due to climate change, through effects on nutrient pools, and on biological activities in waters, which may require more stringent and expensive eutrophication policies. On the other hand, further technological development could make nutrient abatement less expensive. However, the uncertainties around both climate change and technological development, in addition to abatement costs, are a matter of concern for a risk-averse society. The purpose of this study is to arrive at the method most suited for the cost-effective nutrient management of a eutrophied sea under conditions of uncertain climate change effects and technological development. It engages with the condition of the Baltic Sea, which is regarded as the most damaged sea in the world (e.g. Conley et al. 2009; Elmgren and Larsson 2001).

Climate change is likely to alter precipitation and temperature in the sea and its catchment where changes in $CO_2$ in the atmosphere affect the pH of seawater (e.g. Kabel et al. 2012). Changes in precipitation and temperature can affect nutrient loads from the catchment to the sea, and the processes in the sea. Nutrient loads to the catchment are determined by the runoff from emission sources and retention of nutrients from the emission sources to the sea. Different processes drive the nutrient pools in the sea: nutrient loads from the catchment, nitrogen fixation, denitrification, and nutrient sedimentation and burials. This is also the case with damages caused by eutrophication, such as populations of commercial fishery, invasive species, algal blooms, cyanobacteria and water transparency. It is therefore likely that climate change affects nutrient pools and the determination of nutrient targets for achieving improvements.

In this study, we consider two types of uncertain climate change effects—nutrient loads to the sea and nutrient target for a sea—and apply a safety-first decision framework. Nutrient targets are then formulated as maximum nutrient pools at the latest in a future period to be achieved with a minimum probability and minimum total cost. This so-called chance-constraint programming has an old tradition in economics and has been applied, among others, to food supply and water quality management (e.g. Byström et al. 2000; Kataria et al. 2010;

Shortle 1990). There is also a large body of literature on the economics of technological development (for a review, see e.g. Carraro et al. 2010). In this study, we use the learning-by-doing approach, where costs decline over time as firms gain experience in using a certain technology. Learning by doing is most often described as a function where the repetition of the production process leads to efficiency gains, but can also occur through abatement activities, since cutting back on emissions usually implies the adoption of new, cleaner technologies (Rosendahl 2004). The uncertainty in technological development is parameterized as the uncertain elasticity of learning and is treated in a mean–variance framework where the objective function includes mean and variance in total abatement costs.

Our study rests mainly on the empirical literature on economics of eutrophication. Starting from the mid-1990s, there is by now a relatively large body of literature on cost-effective or efficient nutrient load reductions to a eutrophied sea. To the best of our knowledge, only two studies evaluate the effects of climate change with respect to the sea, i.e. Gren (2010) and Lindkvist et al. (2013), and one considers the implications of technological development (Lindkvist and Gren 2013). Most studies calculate cost-effective or the efficient allocation of abatement among the riparian countries in a deterministic setting (Gren et al. 1997; Elofsson 2006, 2007; Ollikainen and Honkatukla 2001; Laukkanen and Huhtala 2008; Ahlvik and Pavlova 2013). The focus is often on optimal nutrient management in one drainage basin including only agriculture (Hart and Brady 2002; Hart 2003) or this sector together with sewage treatment (Elofsson 2006; Laukkanen and Huhtala 2008; Laukkanen et al. 2009; Helin et al. 2008). However, none of the studies on eutrophication in a sea consider the uncertain effects of climate change and technological development. On the other hand, this combination of uncertainties has been applied to energy policy in the works of Held et al. (2009) and Schmidt et al. (2011), who use a similar approach that includes assigning probabilistic constraints on emission targets. This chapter, therefore, adds to earlier literature on the dynamic management of eutrophied seas and lakes by addressing uncertain climate change effects and technological development.

The study is organized as follows: First, the model is presented, which is followed by an analysis of the properties of cost-effective solutions. In Section "Application on the Baltic Sea", the model is analysed with respect to the Baltic Sea, followed by concluding remarks in the next section.

## A Simple Dynamic Model with Uncertainty

The numerical dynamic model builds on the work of Gren et al. (2013), while adding climate change and endogenous technical change as conditions of uncertainty. The total load of a nutrient in each time period, $L_t^E$ for $E = N$, $P$ where $N$ is nitrogen and $P$ is phosphorus, is the sum of business as usual (BAU) discharges, $I_t^{i,E}$, from all countries $i = 1, \ldots, n$ into the sea, minus abatement, $A_t^{i,E}$, according to

$$L_t^E = \sum_i I_t^{i,E} - A_t^{i,E}. \tag{3.1}$$

The response mechanisms and time required for the sea's adjustments to the loads described by Eq. (3.1) differ across nutrients. There is phosphorus cycling in the sea due to biotic activity, but it is also sequestered in the sediment pool in normal oxygen conditions. Under conditions of oxygen deficit, part of this sequestered phosphorus can be released into the water body and re-enter the cycle. In addition to similar biotic cycling, nitrogen is denitrified into harmless nitrogen gas and, thus, removed from the cycle, but can also be supplied to the Baltic Sea by the nitrogen-fixing cyanobacteria under appropriate conditions. These adjustment mechanisms may result in a non-linear system with associated difficulties of identifying optimal abatement paths (e.g. Mäler et al. 2003). Furthermore, the responses of nitrogen and phosphorus cycles are connected. For example, reductions in phosphorus pools may decrease the nitrogen fixation by cyanobacteria (e.g. Savchuk and Wulff 2009). However, these connections are poorly understood in quantitative terms, and we therefore assign a simple linear relation between stock of nutrient $E$ in period $t+1$, $S_{t+1}^E$, and prior period $t$ and nutrient load, which is written as

$$S_{t+1}^E = (1 - \alpha^E)S_t^E + L_t^E, \tag{3.2}$$

$$S_0^E = \overline{S}^E.$$

Following Gren et al. (2013), targets are set on maximum nutrient pools in a certain period, $K_T^E$, which are expected to bring about desired improvements in water transparency, algal blooms and populations of commercial fish. Climate change is then assumed to affect both nutrient stocks in each period of time, $S_t^E$, and the targets, $K_T^E$. We assign simple representations of these effects by assuming a multiplicative impact of climate change on nutrient pools, $\phi^E > 0$, and on nutrient pool target in time $T$, $\gamma^E > 0$. When $\phi^E = \gamma^E = 1$, there is no climate change impact. For parameter values below (above) unity, climate change implies a decline (increase) in nutrient pools and a reduction (increase) in the acceptable nutrient pools in the target year. The net effect is then either a reduction or an increase in total abatement cost for achieving the target.

Both climate change parameters are assumed to be normally distributed with an average of $\mu^{\phi,E}$ and variance $\sigma^{\phi,E}$ for nutrient pool impacts, and $\mu^{\gamma,E}$ and $\sigma^{\gamma,E}$ for impacts. When decision-makers have a relatively strong aversion against deviations from a target, safety-first decision rules can be particularly useful. The safety-first criterion can, in turn, be formulated in different ways, with different outcomes (e.g. Pyle and Turnovsky 1970). This chapter applies the safety-first criterion originally suggested by Tesler (1955), which allows for the adoption of relatively easy and accepted decision rules, while minimizing costs under nutrient pool constraints, where the pool constraints are formulated in probabilistic terms. Chance-constrained programming is then applied, where it is assumed that the objective of the policy maker is to minimize total abatement costs for achieving a probabilistic target constraint for maximum allowable nutrient pools (see e.g. Charnes and Cooper 1964; Birge and Louveaux 1997). The nutrient pool targets, $K_T^E$, must then be achieved with a minimum level of a chosen probability $\beta^E \in (0, 1)$, which is written as

$$\text{prob}(\phi^E S_T^E \leq \gamma^E K_T^E) \geq \beta^E. \tag{3.3}$$

Similar probabilistic targets have been formulated in several studies in environmental economics, water quality management (Shortle 1990; Byström et al. 2000; Elofsson 2003; Kataria et al. 2010), biodiversity protection (Gren et al. 2014) and climate change (Held et al. 2009; Gren et al. 2012). We follow this literature and apply chance-constrained programming to translate the restriction in Eq. (3.3) into a deterministic framework, which allows for relatively easy numerical solutions (e.g. Taha 2007). The probability restriction in Eq. (3.3) can then be written as

$$\text{prob}\left[\frac{S_T^E - K_T^E - (\mu^{\phi,E} S_T^E - \mu^{\lambda,E} K_T^E)}{(\sigma^E)^{1/2}} \leq \frac{0 - (\mu^{\phi,E} S_T^E - \mu^{\lambda,E} K_T^E)}{(\sigma^E)^{1/2}}\right] \geq \beta^E, \tag{3.4'}$$

where $\sigma^E = \text{Var}(\phi^E S_T^E - \gamma^E K_T^E) = S_T^{E^2} \sigma^{\phi,E} + K_T^{E^2} \sigma^{\gamma,E} - 2S_T^E K_T^E \text{Cov}(\phi^E, \gamma^E)$. The term $\frac{S_T^E - K_T^E - (\mu^{\phi,E} S_T^E - \mu^{\lambda,E} K_T^E)}{(\sigma^{\phi,E} + \sigma^{\gamma,E})^{1/2}}$ shows the number of standard errors, $\psi^E$, that $S_T^E - K_T^E$ deviates from the mean values. By the choice of $\beta^E$, there is a level of acceptable deviation, $\psi^{\beta,E}$, and the expression within brackets in Eq. (3.4') then holds only if

$$\mu^{\phi,E} S_T^E + \psi^{\beta,E} \sigma^{E^{1/2}} \leq \mu^{\gamma,E} K_T^E, \tag{3.4}$$

The above expression shows how total abatement costs are affected by uncertain climate change impacts through the implications for the nutrient pool constraints. The minimum cost is reduced when $0 < \mu^{\phi,E} < 1$ and $\mu^{\gamma,E} > 1$. The nutrient pools decline and the targets are revised upwards. On the other hand, the existence of uncertainty in one or both of the climate change impacts results in increased cost when $\psi^{\beta,E} > 0$.

Following Bramoullé and Ohlson (2005), endogenous technical change is described as learning by doing, from accumulation of knowledge through abatement and an initial knowledge stock in each country, $H_0^i$. It is assumed that knowledge is created by the sum of abatement of both nutrients, which is often the case for several technologies involving land-use changes, such as cultivation of catch crops or construction of wetlands. The accumulated knowledge in period $t$, $H_t^i$, is then written as

$$H_t^i = H_0^i + \sum_E \sum_{\tau=0}^{t} A_t^{i,E}. \tag{3.5}$$

The abatement cost in each period of time is assumed to exhibit economies of scope where the cost of simultaneous abatement of both nutrients, $A_t^{i,N}$ and $A_t^{i,P}$, for achieving specific nutrient targets is lower than separate abatement, i.e. $C^i(A_t^{i,N}, A_t^{i,P}) < C^i(A_t^{i,N}) + C^i(A_t^{i,P})$ (e.g. Panzar and Willig 1981; Baumol and Oates 1988). Further, the cost depends on the accumulated knowledge described by Eq. (3.5). The cost function is then written as

$$C_t^i = C^i(A_t^{i,N}, A_t^{i,P})(H_t^i)^{-\theta,i}, \tag{3.6}$$

where $\theta^i = N(\mu^{\theta,i}, \sigma^{\theta,i})$ is the uncertain learning elasticity in absolute terms, which is assumed to follow a normal distribution with mean $\mu^{\theta,i}$ and variance $\sigma^{\theta,i}$. The mean elasticity shows the percentage decrease in costs from 1% increase in abatement accumulation.

The stochastic elasticity of learning gives rise to uncertain decrease in abatement costs, and it is assumed that the decision-makers are risk averse. The objective function is therefore expressed in terms of mean and expected costs in each time period. The decision problem is then formulated as the choice of abatement in different countries and periods that minimizes expected abatement costs plus the cost of uncertainty under the restriction imposed by Eqs. (3.1)–(3.6), which is written as

$$\begin{aligned} &\text{Min} \sum_t \sum_i \sum_E \rho_t \left( \overline{C}_t^i + \eta^i \sigma_t^{C^i} \right) \\ &A_t^{i,N}, A_t^{i,P} \\ &\text{s.t.}(3.1)\text{–}(3.6) \end{aligned} \tag{3.7}$$

where $\rho_t$ is the discount factor, $\overline{C}_t^{i,}$ is expected cost, $\eta^i$ is a measurement of risk aversion and $\sigma_t^{C^i}$ is the variance in costs. The variance can be found from a second-order Taylor expansion around $\theta^i$, which gives $\sigma_t^{C^i} = \mathrm{Var}(C_t^i) = C^i(A_t^{i,N}, A_t^{i,P})^2(-H_t^{-\theta} \ln H_t)^2 \sigma^\theta$.

## Properties of Cost-Effective Solutions

The first-order conditions for a cost-effective solution are obtained by formulating the Lagrangian

$$L = \sum_t \sum_i \sum_E \rho_t \left( \overline{C}_t^i + \eta^i \sigma_t^{C^i} \right) + \lambda_T^E (\mu^{\gamma,E} K_T^E - \mu^{\phi,E} S_T^E - \psi^{E,\beta} \sqrt{\sigma E}), \tag{3.8}$$

where $\lambda_T^E$ are Lagrange multipliers for the restrictions on nutrient pools. Differentiation of Eq. (3.8) with respect to $A_t^{i,E}$ gives, for $E = N$, $P$, $i = 1, \ldots, g$, and $t = 0, \ldots, T$

$$\rho_t \left( \frac{\partial \overline{C}_t^i}{\partial A_t^{i,E}} + \eta^i \frac{\partial \sigma_t^{C^i}}{\partial A_t^{i,E}} \right) = \lambda_T^E \left( \frac{\partial \overline{S}_T^E}{\partial A_t^{i,E}} + \psi^{\beta,E} \frac{1}{2} \left( \frac{\partial \sigma^E)}{\partial A_t^{i,E}} \right)^{-1/2} \right), \tag{3.9}$$

where

$$\frac{\partial \overline{C}_t^i}{\partial A_t^{i,E}} = \frac{\partial C^i(A_t^{i,N}, A_t^{i,P})}{\partial A_t^{i,E}} H_t^{-\theta,i} - \theta \left( \sum_t^T C^i(A_\tau^{i,N}, A_\tau^{i,P}) H_\tau^{-(\theta+1),i} \right) \geq 0, \tag{3.10}$$

$$\begin{aligned} \frac{\partial \sigma_t^{C^i}}{\partial A_t^{i,E}} \frac{1}{2\sigma^\theta} &= \frac{\partial C^i(A_t^{i,N}, A_t^{i,P})}{\partial A_t^{i,E}} (-H_{t,i}^{-\theta} \ln H_t^i)^2 \\ &\quad + H_t^{-(\theta+1),i} \left( \theta \sum_t^T C^i(A_\tau^{i,N} A_\tau^{i,P})^2 (\ln H_t^i - 1) \right), \end{aligned} \tag{3.11}$$

$$\frac{\partial \overline{S}_T^E}{\partial A_t^{i,E}} = -\mu^{\phi,E} \sum_t^T (1 - \alpha^E)^{T-\tau+1} \leq 0, \tag{3.12}$$

$$\frac{\partial \sigma^E}{\partial A_t^{i,E}} = -2(\sigma^{\phi,E} S_T^E - K_T^E \mathrm{Cov}(\phi^E, \gamma^E)) \sum_t^{T-1} (1-\alpha^E)^{T-\tau+1} \leq 0. \quad (3.13)$$

The first-order condition in Eq. (3.9) simply states that a cost-effective solution becomes possible in cases where the marginal cost of achieving the nutrient pool target is equal to $\lambda_T^E$ for all countries. The terms on each side of Eq. (3.9) include marginal impacts of uncertainty—on future development of costs from accumulated abatement on the left-hand side (LHS) and on the pool restriction on the right-hand side (RHS). The condition thus shows that marginal abatement increases the mean and variability of abatement costs and contributes with a reduction in mean and variance in nutrient pools and targets.

The first term on the RHS of Eq. (3.10) is the marginal abatement cost without consideration of effects on future technological development, which is positive. The second term on the RHS is negative and shows the decline in future costs during the period *T–t* due to technological development from a marginal abatement in time *t*. The marginal effect on variance in costs is positive when $\ln H_t^i - 1 > 0$, which can be seen from Eq. (3.11). The signs of marginal effects on the mean and variance in nutrient restriction in Eqs. (3.12) and (3.13) are unambiguously negative, but the magnitude depends on the climate impact parameter $\mu^{\phi,E}$—the lower the parameter, the smaller is the impact in absolute terms. This, in turn, reveals two counteracting effects of $\mu^{\phi,E}$. The first effect can be seen from the constraint (4) where a higher (lower) $\mu^{\phi,E}$ implies a larger (lower) reduction in nutrient loads in order to reach the targets, which raises (reduces) the cost of reaching the targets. The second effect counteracts this cost increase (decrease) by raising (reducing) the marginal effect of abatement on nutrient pools, which can be seen from Eq. (3.12). A larger impact from the given marginal costs of abatement implies lower costs of achieving the targets. Similarly, the uncertainty in climate change effects increases the stringency of the target, which raises the cost, but, as shown in Eq. (3.13), also increases the effect of a marginal abatement on the constraint.

With respect to the derivation of optimal development of abatement over time, the full-fledged condition in Eq. (3.9) does not lend itself to an easy interpretation, and we therefore investigate the optimal paths under different simplifications. In the simplest case, without technological development and uncertainty, the optimal development of abatement over time is guided by

$$\frac{\partial C_{t+1}^{i}}{\partial A_{t+1}^{i,E}} \Big/ \frac{\partial C_{t}^{i}}{\partial A_{t}^{i,E}} = \frac{1}{\rho(1-\alpha^{E})}. \tag{3.14}$$

According to Eq. (3.14), abatement is delayed because of the discount factor and the self-cleaning capacity $\alpha^E$. The discount rate reduces future costs of abatement, and the self-cleaning capacity allows for the use of "free" nutrient pool depreciation. Adding technological development, but not uncertainty, changes the condition to

$$\left(\frac{\partial C^{i}}{\partial A_{t+1}^{i,E}} H_{t+1}^{-\theta,i} - \theta\left(\sum_{t+1}^{T} C^{i} H_{\tau}^{-(\theta+1),i}\right)\right) \Big/ \left(\frac{\partial C^{i}}{\partial A_{t}^{i,E}} H_{t}^{-\theta,i} - \theta\left(\sum_{t}^{T} C^{i} H_{\tau}^{-(\theta+1),i}\right)\right) = \frac{1}{\rho(1-\alpha^{E})}. \tag{3.15}$$

The first term in the numerator and denominator on the LHS of Eq. (3.15) is positive, and for a given marginal abatement cost, a larger cost decrease is obtained from technological development in period $t+1$ than in period $t$ since $H_{t+1}^{-\theta,i} < H_{t}^{-\theta,i}$ for $\sum_{E} A_{t+1}^{i,E} > 0$. This, in turn, reinforces the delay in abatement caused by the discount rate and self-cleaning capacity shown on the RHS of Eq. (3.15). On the other hand, abatement is made earlier while including the second term in the numerator and denominator since $\partial H_{t}^{-\theta,i}/\partial A_{t}^{i,E} > \partial H_{t+1}^{-\theta,i}/\partial A_{t+1}^{i,E}$ because of the longer time period that the marginal abatement acts on cost decreases from technological development. Depending on the relative magnitude of these two forces, optimal abatement is either delayed or advanced, compared with the optimal abatement path without technological development.

The introduction of uncertainty in the learning elasticity will affect the optimal rate of abatement over time. The expression for optimal development paths then becomes quite difficult to solve analytically. Intuitively, the effect of uncertainty on the development in learning rate over time is likely to delay abatement compared with the deterministic case. Relatively early abatement leads to accumulation of abatement over a longer period of time, on which costly uncertainty can act.

Adding uncertainty in nutrient pools in time $T$ does not affect the optimal path of abatement since the effect on $\mathrm{Var}(S_T^E)$ makes no difference, as can be seen on the RHS of Eq. (3.13). Both $\partial \overline{S}_T^E / \partial A_t^{i,E}$ and $\partial \mathrm{Var}(S_T^E)/\partial A_t^{i,E}$ are driven by the same time developments—see Eqs. (3.12) and (3.13). The self-cleaning capacity reduces the mean and variability of the nutrient pools at the same rate. However, since the target stringency increases with the risk discount in nutrient pool variability Eq. (3.4), there is a need for more abatement than under-deterministic conditions, or when $\psi^{\beta,E} > 0$. This, in turn, implies larger abatement in the starting period under uncertainty. The uncertainty in target setting has the same impact, i.e. it will not affect the optimal rate of abatement over time—only at the start.

The main conclusions from this theoretical analysis are as follows:

- The introduction of technological development reduces the overall costs, but can either delay abatement or advance it, depending on the relation between cost reductions from implemented and future abatement.
- The effect of uncertainty in technological development is unambiguous; total costs are increased because of risk aversion. An uncertain learning elasticity increases the variation in future costs and thereby the total cost for risk-averse agents, but the impact of abatement on optimal timing can be determined only by empirical analysis.
- The uncertainty in nutrient pools and targets also increases the cost, which is a result of the need for more abatement in order to reach a minimum probability of achieving average targets. This implies earlier abatement in order to achieve the targets.

## Application on the Baltic Sea

The Baltic Sea is not only the largest brackish water sea in the world but also the sea with the largest area of sea bottom without biological life caused by eutrophication (Conley et al. 2009). This is not a new finding—signs of damage from eutrophication had already been detected in the 1960s, and an international administrative body, the Baltic Marine Environment Protection Commission (HELCOM), also known as the Helsinki Commission, was established in 1974 in order to monitor the status of the sea and coordinate mitigation actions. Since then, three international governmental agreements on nutrient load reductions have been signed (HELCOM 1988, 2007, 2013). All these agreements are supposed to be based on desired improvements in the functioning of the sea, such as reduced frequency of toxic algal blooms, larger populations of commercial fish and higher water transparency, but only one of them presents required reductions in the nutrient pools (HELCOM 2007), which are reported in Gren et al. (2013). We therefore apply the dynamic model to this agreement. The latest agreement from 2013 contains only modest changes in nutrient load reductions, and the calculations will therefore also be valid for this agreement.

## Data Retrieval

The study makes use of data on nutrient loads and abatement costs for each country from a static cost minimization model sea (Gren and Lindkvist 2014). The static model is used for provision of data for the estimation of cost functions for each country as functions of N and P abatement. A quadratic regression equation is assigned to each country, and the data on costs and nutrient reductions are obtained by Monte Carlo simulations with cost-effective solutions of 500 random combinations of nitrogen and phosphorus reductions. The ordinary least square estimator is then applied for the estimation of coefficients in a quadratic cost function for nitrogen and phosphorus for each country; the results of which are shown in Table 3.1. This approach for deriving cost functions in each time period assumes that cost-effective reductions of nitrogen and phosphorus are implemented in each country.

There are no data on risk aversion in abatement costs for the riparian countries, which are required for calculating the cost of uncertainty in technological development. It is, however, generally agreed that the constant relative risk aversion (CRRA) for market risks ranges between

**Table 3.1** BAU loads of nitrogen and phosphorus loads, abatement cost functions and risk premium in riparian countries in 2008

| Country | Nitrogen, kton[a] | Phosphorus, kton[a] | Parameter values[b] in the cost function, in million SEK $C^i = a^i A_t^{i,N^2} + b^i A_t^{i,P^2} - c^i A_t^{iN} A_t^{i,P}$ $a^i$ | $b^i$ | $c^i$ | CARA[c] |
|---|---|---|---|---|---|---|
| Sweden | 74 | 1.6 | 3.57 | 1576.63 | 20.89 | $0.019 \times 10^{-3}$ |
| Poland | 318 | 22.0 | 0.35 | 94.01 | 3.18 | $0.091 \times 10^{-3}$ |
| Finland | 49 | 1.7 | 5.65 | 2089.23 | 16.68 | $0.020 \times 10^{-3}$ |
| Denmark | 44 | 1.1 | 5.29 | 1945.18 | 58.83 | $0.016 \times 10^{-3}$ |
| Germany | 46 | 0.5 | 4.95 | 11,836.62 | 149.80 | $0.023 \times 10^{-3}$ |
| Estonia | 56 | 1.6 | 1.27 | 1394.43 | 31.44 | $0.068 \times 10^{-3}$ |
| Latvia | 44 | 3.0 | 3.61 | 1021.82 | 45.30 | $0.093 \times 10^{-3}$ |
| Lithuania | 93 | 3.5 | 0.78 | 511.14 | 13.15 | $0.094 \times 10^{-3}$ |
| Russia | 83 | 4.0 | 2.90 | 340.17 | 17.36 | $0.116 \times 10^{-3}$ |
| Total | 824 | 38.9 | | | | |

[a]Gren and Lindkvist (2014), Table 3.1
[b]Gren and Lindkvist (2014), Table A2
[c]Constant absolute risk aversion calculated from an assumed relative risk aversion of 5 and evaluated at the mean GDP in 2008

1 and 10, although it can be lower and higher (e.g. Azar 2010). We assume an average CRRA of 5 and calculate a constant absolute risk aversion (CARA) for each country, which is evaluated at the mean GDP/capita in Table 3.1.

Poland is the largest emitter of both nitrogen and phosphorus, and accounts for 39% of total nitrogen load and 57% of total phosphorus.

With respect to estimates of learning elasticities, there is a relatively large body of literature relevant to manufacturing and energy technologies (e.g. MacDonald and Schrattenholzer 2001; Rasmussen 2001; Jamasb 2007). However, there is no study considering all the different abatement technologies included in Gren and Lindkvist (2014), which constitute a mix of mature, emerging and new technologies with different learning elasticities. Jamasb (2007) has calculated estimates of a combination of different technologies for electricity provision, with a range of 0.03–0.21. We use an average of 0.12 in this study, and assuming normal distribution, the standard deviation for a confidence interval of 0.95 is 0.045. The coefficient of variation is 0.38.

Data on nutrient pools and carry-over rates are obtained from simulations with an oceanographic model (Savchuk and Wulff 2007, 2009) for consistent estimates of nutrient pools and self-cleaning capacities, which are reported in Gren et al. (2013). The carry-over rates vary for different marine basins of the Baltic Sea. We calculated a weighted average for the entire Baltic Sea from the basin-specific carry-over rates and nutrient pools reported in Gren et al. (2013), where the pools constitute weights. HELCOM defines targets for different marine basins depending on their ecological conditions, which vary between 0% and 15% for nitrogen and 0% and 50% for phosphorus (Gren et al. 2013). In this study, we have chosen the most stringent target for each of the nutrients since the nutrient loads are mixed. In a similar vein, targets as measured in average nutrient pools reductions are calculated as the weighted average of reductions in Gren et al. (2013).

Figures quantifying the impact of climate change on nutrient pools are not readily available. Instead, there is a relatively large body of literature on the estimation of impacts on nutrient discharges from single drainage basins in the catchment (see compilation of studies in Lindkvist et al. 2013). The general approach is to use a regional Baltic Sea model, the so-called Rossby Centre Atmosphere Ocean Model (RCAO), for simulating impacts of different climate change scenarios obtained from two global circulation models, at the Hadley Centre, United Kingdom and Max Planck Institute for Metrology in Germany, which are used for setting the boundary conditions that drive the regional RCAO model.

Each global model applies two different $CO_2$ emission scenarios, high and low emissions, obtained from the Intergovernmental Panel on Climate Change (IPCC). This results in four different climate change scenarios with a high or a low future $CO_2$ level and with boundary conditions from one of two different global general circulation models. The results show different impacts on nutrient loads in different parts of the Baltic Sea. The loads of nutrients are expected to decrease for the largest marine basin, Baltic Proper, between 15 and 61% for nitrogen and between 14 and 49% for phosphorus (Lindkvist et al. 2013). On the other hand, loads are expected to increase between 8% and 31% in all other marine basins. Ranges in impacts on total Baltic Sea are calculated

**Table 3.2** Nutrient pools, carry-over rates, targets and uncertainty quantification in pools and targets

| | Pools, kton[a] | Carry-over rate, $(1-\alpha^E)$[b] | Target, % of initial pools[c] | Average climate impact on pools, $\mu^{\phi,E}$[d] | CV in $\mu^{\phi,E}$[e] | Average target effect, $\mu^{\gamma,E}$ | CV in $\gamma^E$ |
|---|---|---|---|---|---|---|---|
| Nitrogen | 2567 | 0.76 | 85 | 0.89 | 0.13 | 0.9 | 0.1 |
| Phosphorus | 558 | 0.94 | 50 | 0.84 | 0.14 | | |

[a]Bioavailable nutrients in Gren et al. (2013), Table 3.1 with shares of total N of 0.844 and total P of 0.943
[b]Weighted average from nutrient pools and self-cleaning rates in Gren et al. (2013), Table 3.1
[c]Weighted average from nutrient pools and target reductions in Gren et al. (2013), Table 3.1
[d]Weighted average from nutrient pools in Gren et al. (2013), Table 3.1, and climate change effects in Lindkvist et al. (2013), Table 3.2
[d]CV, coefficient of variation calculated from the data obtained under[d]

by weighting the impacts calculated for each marine basin with its nutrient pools, which gives a range of 0.66–1.12 for nitrogen and 0.61–1.08 for phosphorus. Standard deviations are then calculated by assuming a 95% confidence interval of a normal distribution.

Currently, there is no study that quantifies the impact of climate change on the targets. Changes in temperature are likely to affect the anoxic sea bottom area, and the $CO_2$ uptake by oceans causes acidification, which impacts biological activities (e.g. Kabel et al. 2012). It is therefore quite likely that, for the given nutrient pools, the damage could be higher, thus counteracting the calculated climate change effects of reduction in nutrient pools. However, in the absence of any figures pertaining to expected changes on settled targets, we simply assume that $\mu^{\gamma^E}$ is the same for both nutrient targets and amounts to 0.9, that is, the coefficient of variation is 0.1, and that the covariation with climate change effects on the nutrient pools is zero.

Finally, there is a need to define the target date when the improvements are to be achieved, the discount rate and minimum probability of achieving targets under uncertainty. The target date is determined by the implementation of abatement measures and the response time of the sea basins. HELCOM-BSAP suggests making 2021 the deadline for the implementation of nutrient load reductions. However, this is not based on any discussion on when the targets are supposed to be achieved. We therefore follow Gren et al. (2013) and apply a time period of 60 years. We choose a relatively low discount rate of 0.015. With respect to choice of probability of achieving the targets, it is assumed to be the same for both nitrogen and phosphorous, amounting to $\beta^N = \beta^P = 0.8$.

## Results

Minimum costs are calculated for the impacts of learning and the two climate change impacts, separately and in combination, and with and without uncertainty. The GAMS/CONOPT2 solver is used for the numerical solutions (Rosenthal 2008).

The total cost for achieving the targets in the reference case, without climate change impacts and technological development, amounts to 768 billion SEK. This corresponds to an average annual cost of approximately 12.8 billion SEK, which is lower than the cost (15 billion SEK) for achieving the same targets as calculated by Gren et al. (2013). The reason for the difference is the focus on the entire Baltic Sea in this study, which makes the average nutrient carry-over rates lower. Gren et al. (2013) assign targets for each of the seven marine basins, where the self-cleaning capacity of the largest basin is two-third of the rate employed in this chapter. However, the minimum cost could be even lower, in particular when the favourable condition of climate change impacts on pools or learning act. Without any uncertainty, the total discounted cost is reduced to 358 billion SEK. The cost-reducing effect of learning and the lower nutrient loads to the sea then dominate the cost-increasing effect of more tight nutrient targets from climate change effects. On the other hand, when all three types of uncertainties are included, the total cost rises to 1169 billion SEK. These differences in total costs are transferred to the costs for different countries under the three types of scenarios. However, as can be seen in Fig. 3.1, the cost for various countries differs.

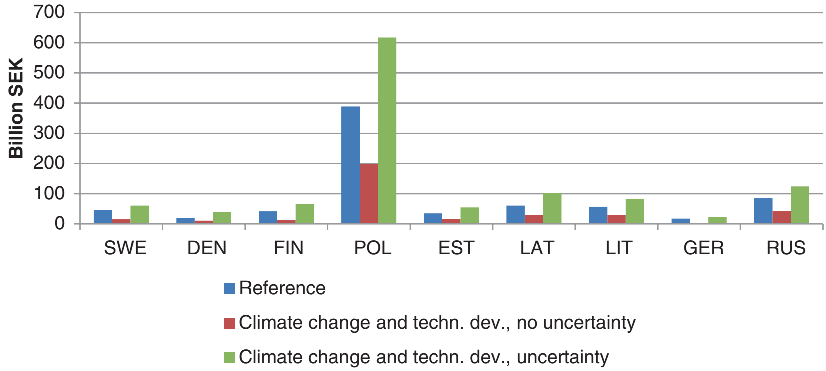


**Fig. 3.1** Allocation of costs on countries for achieving 50% reduction in phosphorus and 15% reduction in nitrogen pools in 60 years in the reference case and with effects of climate change and technological development with and without certainty (*SWE* Sweden, *DEN* Denmark, *FIN* Finland, *POL* Poland, *EST* Estonia, *LAT* Latvia, *LIT* Lithuania, *GER* Germany, *RUS* Russia)

It has been mentioned earlier that Poland accounts for the largest loads of both nutrients. This, in combination with the large reduction requirement of phosphorus, explains why the abatement costs are highest for Poland in all scenarios.

With respect to the timing of abatement costs, all scenarios show three phases: (i) an initial period of about 20–25 years with low annual costs, (ii) a slow increase in cost for a period of 15–25 years and (iii) a rapid increase during the last 5–10 years (see Fig. 3.2). The first part is explained by the cost savings made from delaying abatement due to the discount rate and the self-cleaning capacities, as shown by Eq. (3.13) in Section "Properties of Cost-Effective Solutions". The second phase arises from the cleaning of phosphorus in order to reach the target in period 60. The abatement of nitrogen, which has a higher turnover rate, is carried out in the third phase.

However, in all scenarios, the costs are sensitive to the assumption of the chosen reliability level in achieving the targets. Under the assumption of normal probability distributions, a reliability level of 0.5 corresponds to the deterministic case. The uncertainty in technological development then acts, and the total abatement cost amounts to 550

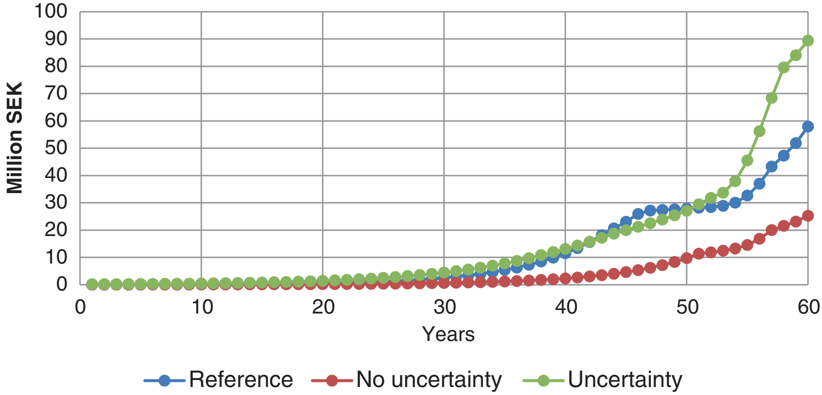


**Fig. 3.2** Optimal time paths of annual discounted abatement costs in the reference case and with climate change and technological development with and without uncertainty

billion SEK. If the reliability concern increases to $\beta^N = \beta^P = 0.95$, the total cost increases to 1728 billion SEK.

## Conclusions

Climate change is likely to result in several uncertain impacts on a eutrophied sea, and this study investigates the implications for cost-effective solutions for two of them—effects on nutrient pools and target setting. Depending on the direction of the impacts, the total abatement cost can either increase or decrease. Unlike this ambiguous result, the introduction of technological development from learning by doing always results in a decrease in cost. On the other hand, the introduction of uncertainty always increases the cost, irrespective of its origins—nutrient pools, target setting or technological development. However, the combined impact can either increase or decrease the total abatement cost of all impacts, depending on the magnitude of the effects and on risk aversion against non-attainment of targets and variability in costs.

The model was applied to the most recent intergovernmental agreement for combatting eutrophication in the Baltic Sea, which requires an average of 50% reduction in the phosphorus and 15% in the nitrogen pool. The minimum cost for achieving these targets in 60 years is 768 billion SEK in the reference without any technological development or climate change. However, the cost can either decrease by approximately 50% or increase by 125%, depending on the magnitude of impacts and levels of reliability concern of achieving the targets. The most favourable condition is the combined effect of technological development and climate impact on nutrient pools. The latter is expected to decrease by approximately 15% due to climate change. However, if it is climate change that requires a larger reduction in nutrient pools, the cost can increase considerably.

Undoubtedly, our empirical results for the Baltic Sea show that the existence of uncertainty, and the aversion against it, increases the abatement cost considerably. However, the results also show that climate change may facilitate the implementation of nutrient abatement strategies because of the expected reduction in nutrient pools. These results point out the importance of analysing and quantifying different climate change impacts and, in particular, their combined effect on nutrient pools, since, in isolation, they may underestimate or overestimate minimum cost solutions to pre-specified targets.

Our empirical results also show that one country, Poland, faces the largest cost burdens in all cases. Whether or not such cost-effective solutions can be implemented in practice is likely to depend on international policy and compensation system. Another policy challenge is the need for “ecosystem service stacking” for abatement measures that affect both nitrogen and phosphorus loads, such as the construction of wetlands (see e.g. Robertsson et al. 2014). This means that such abatement measures should be employed for the abatement of both these nutrients and not only one, which has been assumed in the calculations of cost-effective solutions in this chapter. The costs increase if this is not the case.

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# 4

# Optimal Strategies for Inland and Coastal Water Monitoring

Katarina Elofsson

## Introduction

Many large aquifers across the world suffer from increased eutrophication, which has negative consequences for biodiversity, fisheries, recreation, and ecosystem health (Chai et al. 2006; Gustafsson et al. 2012; Rabotyagov et al. 2014). The Baltic Sea is one of the most severely affected seas, and has the largest dead zone in the world (Diaz and Rosenberg 2008). The major cause of this is the increasing load of nutrients flowing into the sea. Nutrient emissions originate from point and in the agricultural, transport, energy, and wastewater sectors. The difficulties faced in identifying measures that need to be prioritized based on economic efficiency have been widely discussed in literature on the subject (Brouwer and DeBlois 2008; Elofsson 2003; Gren et al. 1997; Ribaudo et al. 1999). Some of these lie in identifying the

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relationship between activities around upstream sources and the state of the recipient, evaluating the environmental status of the recipient, and identifying the value that people attach to improving the environment.

Problems associated with the eutrophication of upstream lakes and rivers can often be observed together with the eutrophication of downstream coastal waters. For example, simultaneous eutrophication of upstream and downstream waters can be found in many parts of the southern half of the Baltic Sea drainage basin, where agricultural production is substantial. In such a situation, it is easier to establish linkages between emission sources and the environmental status of nearby lakes and rivers than between the emission sources and environmental status of downstream coastal waters (Smith 2003). Thus, despite the relatively high certainty with which an emission source can be linked to the environmental deterioration of an upstream lake or river, this need not imply that the abatement of emissions will positively affect downstream water quality. There are several reasons for this. For example, high nutrient retention, i.e. high uptake of nutrients in the lake and river vegetation and sediments could imply that the emissions from a given source may be reaching nearby lakes and rivers but not downstream coastal waters. In addition, downstream coastal waters could be in comparatively good condition even when nutrient loads from upstream sources are high—for example, if there is a high degree of dilution, or if the hydrological properties of coastal waters imply that the area is insensitive to nutrient pollution. Conversely, downstream water quality could be heavily diminished even if nutrient loads from upstream sources are low. This could happen either because of the historical load of nutrients owing to sources from which emissions have ceased, or external loads from distant sources (Smith 2003).

In order to identify major sources and to evaluate the status of water quality, monitoring is required. Lovett et al. (2007) argue that existing monitoring programmes are less expensive than the cost of abatement and the monetary benefits associated with improving the environment. They claim that the cost of actual water quality monitoring constitutes only 0.4–2.1% of the total cost of complying with the Clean Water Act in the USA. In contrast, Kampas and White (2004) estimate that the monitoring costs incurred for nitrate-sensitive areas constitute as much as 40% of the total administrative cost for that regulation, corresponding to 42 EUR/ha. Strobl and Robillard (2008) argue that many existing water quality monitoring schemes are limited in their scope and scale, and their design is far from optimal. For example, monitoring objectives are often unclear, and the representativeness of sampling locations and the suitability of sampling frequencies and monitored water quality variables could be questionable. On the other hand, full-scale monitoring of, for example, sediment and nutrient runoff from all farmers, would be expensive, given the large number of farmers (Claassen et al. 2008). The ideal monitoring strategy requires clear identification of programme objectives; evaluation of the importance of the programme's spatial distribution (i.e. the number of monitoring stations), of trends (i.e. the sampling frequency), and of different pollutants; and the full coverage of all three aspects is likely to be very costly (Chapman 1992). In practice, a mix of different monitoring activities is typically applied at a modest scale, including monitoring of activities at the source, emissions from each source, water quality in lakes and streams, and downstream water quality.

The purpose of this chapter is to investigate optimal monitoring and abatement strategies in a situation where both upstream and downstream water quality pose a potential problem. The study focusses on the scale of overall monitoring and how it is determined by the relative benefits and costs of upstream and downstream monitoring stations. It is assumed that there is a risk-averse regulator who aims to achieve the largest possible utility from the choice of combined upstream and downstream monitoring, given the possible outcomes of monitoring, and the consequential decision on abatement at the emission sources. This chapter investigates how monitoring and abatement costs, combined with the degree of the regulator's risk aversion affect the choice of monitoring strategy.

There is scant economic literature on water quality monitoring strategies. Farzin and Kaplan (2004) use a static model for the minimization of expected pollution damage under a given budget, which can be used for abatement at different emission sources, and an endogenously chosen data collection effort. Data collection changes the relative expectations about the contribution of different sources to aggregate pollution.

Kaplan et al. (2003) extend this approach in a dynamic sediment control model where the decision-maker simultaneously decides on abatement at multiple sources and intensity of a single data collection activity. The empirical application is restricted to a comparison of two discrete levels of data collection intensity, while also elaborating on the use of the sequential entropy filter as a means for updating beliefs about probabilities. Contrary to these, other studies treat the design and intensity of monitoring as exogenous. Bond and Loomis (2009) analyse adaptive management of a shallow lake where there is uncertainty regarding both phosphorus background loads and a threshold in the damage function. The decision-maker can actively vary emissions in order to learn more about lake responses, and is assumed to apply Bayesian updating of beliefs. A discrete-time dynamic programming model is used to empirically estimate the value of experimenting, and results indicate that this value can be relatively small. Similar to Bond and Loomis (2009), Peterson et al. (2003) analyse regime switches in a shallow lake for a given monitoring scheme. They assume that the decision-maker considers two competing models for the lake ecosystem response to phosphorus loads, an oligotrophic and a eutrophic model, and attaches different probabilities to these models. The subjective probabilities are updated over time as the decision-maker observes phosphorus concentrations.

Environmental monitoring is also considered in a Climate change context. For example, Kelly and Kolstad (1999) consider passive Bayesian learning about the relationship between greenhouse gas levels and global mean temperature changes, under a given monitoring scheme, and no experimentation. Cunha-e-Sá and Santos (2008) add the possibility of experimentation under an exogenously given monitoring scheme, which could improve learning about carbon stock decay and carbon emission coefficients, but they find no gains from experimentation.

White (2005) analyses monitoring strategies for the reestablishment of vegetation on a given piece of land, with both conservation and monitoring efforts as endogenous variables. Assuming that the regulator applies Bayesian updating, he shows that decisions on both monitoring and continuation of a contract can be solved if the vegetation state follow a partially observed Markov decision process. The empirical application is restricted to a discrete set of conservation and monitoring effort levels.

The present study has similarities with the above studies, which treat monitoring as endogenous. It differs from them in making a distinction between monitoring of emission sources and monitoring of downstream coastal water quality. It also differs from them by treating monitoring as a discrete, one-time choice, and hence by formulating the monitoring decision as one of monitoring stations rather than data collection effort. The choice of location of the monitoring station is more important if monitoring is capital rather than labour intensive. Data for Swedish river mouth monitoring of nutrients suggest that fixed and variable cost are of similar magnitude (Fölster 2014), confirming that the fixed costs are at least of similar importance as the variable costs. This chapter further departs from most of the above-mentioned studies by abstracting from the role of continuously added information through additional samples, while instead emphasizing discrete improvements in information following the investment in monitoring stations.

This chapter is organized as follows. First, a model of upstream and downstream monitoring choices is presented. This is followed by numerical simulations. This chapter ends with discussion and conclusions.

## A Model of Upstream and Downstream Monitoring Choices

A simple model of a stylized watershed is developed to analyse the optimal monitoring choices, when monitoring and abatement decisions are taken sequentially. It is assumed that there is a regulator who decides on monitoring and abatement. There are two potential emission sources in the watershed, which can be thought of as two farms. Without monitoring, the regulator is uncertain as to whether there are actually emissions from the two sources. Even if there are activities at the sources which can potentially generate emissions, management practices, technology, and locally specific soil conditions determine whether emissions actually occur. In proximity to the two potential sources, there is an upstream water recipient, which can be a lake or a river. The regulator knows with certainty that the upstream recipient is eutrophicated if, for example, the water is turbid and there are large algal blooms and a strong tendency for the overgrowth of shores. Therefore, no monitoring is necessary to verify the environmental status of the upstream recipient. However, the regulator cannot be sure whether this is due to emissions from the two potential sources. Alternative causes of eutrophication of the upstream recipient are, for example, historical emissions from other sources, where emission activities have already ceased.

Further, there is a downstream recipient, a coastal marine bay. The environmental status of the coastal water is not fully known, due to the scientific difficulties in evaluating the complex state of coastal marine waters. If monitoring is carried out, the state of the downstream recipient can be assessed. If it is in a good state, abatement at the sources will not have any beneficial effect on downstream water quality. A bad state could be a result of either emissions from the two upstream sources, or high background loads and/or emissions from distant external sources. It is assumed that if it is established that there are emissions from the two sources in the watershed, the regulator can be certain that abatement of these emissions will improve downstream water quality if the water quality is bad. The regulator can then draw this conclusion if there are considerable emissions from the sources, and the rate of retention in lakes and rivers is known. In such a case, emissions from the inspected sources are likely to explain the bad water quality in coastal waters to at least some extent.

This situation is described as a sequential decision-making problem where, first, nature determines the state of emissions at the source and the environmental status of the downstream recipient. Thereafter, the regulator decides on a monitoring strategy, implements it, and decides on whether abatement should be carried out. Based on all alternative outcomes, the regulator can then choose the monitoring strategy that gives highest utility *ex ante*. The different steps in the decision process are illustrated in Fig. 4.1, followed by a detailed description of each step.

Step 1: Nature determines the state of upstream emissions and downstream water quality.

Before any decisions are taken by the regulator, nature determines the state of emissions from the source and the state of downstream water

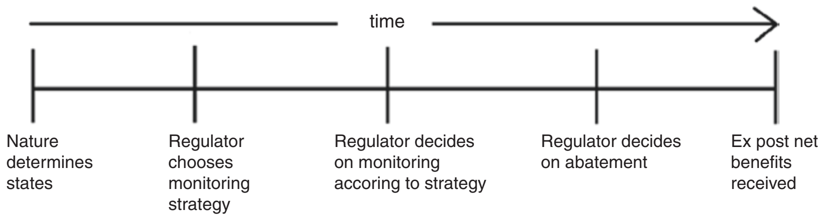


**Fig. 4.1** Timing of events. *Source* Author

quality. Assume that there are two non-point sources $i$, with $i = 1$, 2. The initial emissions from the sources , $e^0_{\text{in}}$, depends on the state of nature $n$, with $n = Y, N$, where $Y$ indicates the presence of emissions, and $N$ indicates that there are no emissions. It is assumed that emissions are a function of a vector of management practices $\boldsymbol{\omega}$, which consist of factors such as manure spreading practice and precision farming technology, and a vector of site-specific conditions $\boldsymbol{\theta}$, such as rainfall, soil type, slope, and proximity to water courses, i.e. $e^0_{\text{in}} = e^0_{\text{in}}(\boldsymbol{\omega}, \boldsymbol{\theta})$, which ultimately determine the two discrete states.

It is assumed that the state of the emission sources can be monitored. An alternative would be to assume the monitoring of management practices and site-specific conditions. If the regulator knows the relationship between management practices and site-specific conditions on the one hand, and the emissions on the other, he/she could instead monitor the former two if this would entail a lower cost, without affecting the formulation of the model.

It is assumed that nature assigns one of two possible emission levels, each associated with a state $n$, to a source. Emissions can then be either $e^0_{iY} = e^0_i > 0$ or $e^0_{iN} = 0$, i.e. emissions are either at a fixed, positive level, or zero.

Second, nature assigns the environmental state, $s$, with $s = G, B$, of the downstream recipient, where $G$ denotes good water quality, implying that abatement at the sources will not improve utility. If, on the other hand, water quality is bad, denoted by $B$, abatement at the sources will improve water quality and hence the regulator's utility, provided that emissions from the source are initially positive, i.e. $e^0_{iY} = e^0_i$.

Step 2: Regulator decides on overall monitoring strategy

We assume that the decision-maker has three major alternative monitoring strategies to choose from. First, he/she could simultaneously decide on up- and downstream monitoring. However, given that the knowledge of the outcome of upstream monitoring affects whether it is worthwhile to carry out downstream monitoring as well, and vice versa, the regulator could also consider first implementing upstream monitoring, and depending on the outcome, decide on downstream monitoring, or the other way around. We consider three alternative strategies: *I* simultaneous, one-shot decision on up- and downstream monitoring; *II* monitoring the sources first; and *III* monitoring the downstream recipient first. Finally, it is assumed that the regulator is risk averse, and evaluates the choice between strategies *I–III* based on a comparison of the *ex ante* utility of each strategy. We return to the choice of monitoring strategy after a closer look at the decisions under each strategy, and the abatement decision.

Step 3: Regulator decides on upstream and downstream monitoring

Once the overall monitoring strategy is decided, it still remains to be ascertained whether there will be a full or no monitoring under strategy *I*, and whether upstream monitoring will be followed by downstream monitoring or not under strategy *II*, and vice versa in the case of strategy *III*.

While it is evident that the regulator can reduce uncertainty by monitoring, monitoring is costly. Therefore, there needs to be a trade-off between reduced uncertainty and additional costs. By monitoring the sources, the regulator can establish the volume of emissions that originate from each source. He/she could also choose to monitor downstream water quality in order to establish whether abatement at the sources is meaningful. It is assumed that the regulator has a discrete choice between monitoring and not monitoring each source. The monitoring effort at source $i$ is denoted by $m_i$, with $m_i = [0, 1]$ . The regulator has beliefs about the size of emissions from source $i$, $p_i\left(e_i^0 \middle| m_i\right)$, which depend on whether monitoring is carried out or not. The prior beliefs, $\hat{p}_i\left(e_i^0\middle|0\right)$, are the subjective beliefs that the regulator holds before monitoring is carried out. It is assumed that $\hat{p}_i\left(\hat{e}_i^0\middle|0\right) = \hat{p}_j(0|0) = 0.5$, i.e. the regulator assigns equal probabilities to positive and zero emissions. If monitoring is carried out, it is assumed that the regulator will, thereafter, know the volume of emissions with certainty. The posterior beliefs, i.e. beliefs after monitoring, are defined by $p_i\left(e_i^0\middle|1\right) = \left[0, \bar{e}_i^0\right]$.

The regulator also decides on whether to monitor downstream recipient water quality. The downstream monitoring effort $d$, is a discrete choice between monitoring and not monitoring, such that $d = [0, 1]$. As mentioned above, it is assumed that if both upstream and downstream monitoring is carried out, then the regulator will know with certainty if, and how much, abatement can improve downstream water quality. The regulator consequently has beliefs $q(\cdot)$, about increased benefits, $B^D$, of downstream water quality, $q\left(B^D(a_i)\middle|m_i, d\right)$, which depend on both upstream and downstream monitoring. For abatement at a particular source to improve downstream water quality, it is necessary that emissions at that source are initially positive and that the state of water quality at the downstream recipient is bad. Beliefs prior to downstream monitoring are then given by $\hat{q}\left(B^D(a_i)\middle|m_i, 0\right)$ and posterior beliefs are given by $q\left(B^D(a_i)\middle|m_i, 1\right)$.

The monitoring of sources is assumed to be associated with a cost $C^{MU} = \sum c_i m_i$, where $c_i$ is a fixed cost of monitoring at source $i$. Similarly, downstream monitoring is associated with a cost, $C^{MD} = td$, where $t$ is the fixed cost of downstream monitoring. The assumption about a fixed monitoring cost is a simplification. Monitoring typically entails not only a fixed, one-time capital cost for equipment, automatic measuring stations and infrastructure, but also operation cost in terms of, for example, labour (World Meteorological Organization 2013). Data for the Swedish river mouth nutrient monitoring, fixed costs constitute approximately half of the total cost, bolstering the assumption that fixed costs are important in a monitoring context.

Step 4: Regulator decides on abatement

Once monitoring has been decided on and carried out, the regulator proceeds to decide on abatement, $a_i$, at the sources. This is, again, assumed to be a discrete choice between abating and not abating, i.e. $a_i = [0, 1]$. Abatement is assumed to be associated with a cost $C^A(a_i) = \sum_i g_i a_i$, where $g_i$ is the fixed cost of abatement at source $i$. One may note that fixed costs of abatement are relevant if the abatement involves an investment, for example, an investment in a manure storage facility.

Having carried out abatement, the regulator achieves the *ex post* benefits of upstream and downstream water quality improvement. The *ex post* benefits of upstream water quality $B^U$, with $B^U = B^U\left(a_i | e_i^0\right)$, is a function of abatement at the two sources and initial emissions. Zero benefits from abatement at given source $i$ are achieved when initial emissions or abatement is zero for the source in question, i.e. $B^U(a_i|0) = B^U\left(0|e_i^0\right) = 0$. The benefits of abatement at a single source, where emissions are initially positive, is assumed to be equal to $k_i$, i.e. $B^U\left(a_i > 0, a_j = 0 \middle| e_i^0 > 0, e_j^0 \geq 0\right) = k_i a_i$. The achieved benefits are assumed to be additive in abatement, i.e. $B^U\left(a_i > 0|e_i^0 > 0\right) = \sum_i k_i a_i$, i.e. when emissions at two sources are abated, and both have positive initial emissions, the value of the improvement of upstream water quality is doubled. This is a simplification compared to the standard assumption of a convex damage function, motivated by our focus on the trade-off between upstream and downstream monitoring.

*Ex post* downstream benefits of improved water quality, $B^D$, is a function of abatement at the two sources, emissions from the sources, and the state of downstream water quality, and is defined by $B^D\left(a_i|e_i^0, s\right)$. It is assumed that $B^D(a_i|0, s) = B^D\left(0|e_i^0, s\right) = B^D\left(a_i|e_i^0, G\right) = 0$. Thus, it is pointless to have zero abatement, zero initial emissions and abatement when the downstream water quality is good. If downstream water quality is initially bad, abatement at one source, which has positive emissions, yields a benefit equal to $l_i$: $B^D\left(a_i > 0, a_j = 0 \middle| e_i^0 > 0, e_j^0 \geq 0, B\right) = l_i a_i$, and abatement at two sources with positive emissions results in higher, additive, benefits: $B^D\left(a_i > 0|e_i^0 > 0, B\right) = \sum_i l_i a_i$.

Summing up, the potential impacts of upstream and downstream monitoring on *ex post* benefits are different in nature. The advantage of upstream monitoring is that the regulator will know with certainty whether there is any point to abatement at all—if emissions are zero, abatement will not affect upstream or downstream water quality. Once upstream monitoring is carried out, the decision-maker will at least know with certainty whether abatement affects upstream water quality, and uncertainty about impact of abatement on downstream water quality will be reduced. The disadvantage of upstream monitoring is that full-scale monitoring requires monitoring efforts at several emission sources—in this case, two—implying that this exercise typically becomes more expensive compared to monitoring a single downstream location. The benefits of downstream monitoring are more limited as it sheds light on the likelihood that abatement will positively affect downstream water quality, but the impact will still not be certain, and it does not add to knowledge about the impact on upstream water quality. On the other hand, only a single location needs to be monitored, implying that the monitoring cost is lower. Thus, the relative advantages of upstream and downstream monitoring depend on the net benefits of abatement with respect to upstream and downstream water quality, given the decision-maker's risk aversion and monitoring costs.

In step 4, the levels of monitoring have already been chosen. These levels are denoted by $\bar{m}_i$ and $\bar{d}$ respectively. The regulator then chooses abatement at the two sources given his/her posterior beliefs about emissions and downstream water quality, with an aim to maximize expected net benefits of abatement:

$$\begin{aligned} \text{Max}_{a_i}\ E[NB(a_i)] = \arg\max \Big\{ & B^U\left(a_i|p_i, e_{in}^0, \bar{m}_i\right) \\ & + B^D\left(a_i|p_i, e_{in}^0, \bar{m}_i, s, \bar{d}\right) - C^A(a_i) \Big\} \end{aligned} \tag{4.1}$$

The expression in Eq. (4.1) implies that the regulator will maximize the expected net benefit by choosing abatement levels that yield the largest possible expected benefits from improved upstream and downstream water quality, given the cost of abatement at the sources and the chosen monitoring strategy. This results in optimal abatement $\tilde{a}_i$ as a function of the chosen monitoring effort, i.e. $\tilde{a}_i = \tilde{a}_i(p_i(m_i), q(d))$.

The Pre-monitoring problem

In this section, we proceed to define the *ex ante* decision problem under each of the three alternative strategies. To do this, we solve the problem described above using backwards induction. This gives us the *ex ante* expected net benefits of abatement. However, given that the regulator is assumed to be risk averse, we also evaluate the *ex ante* utility, which is associated with the net benefits from a strategy, assuming that utility is concave in net benefits.

We begin with the decision problem in step 4 and successively work backwards to identify *ex ante* utility from among the alternative monitoring strategies, assuming that the regulator has a mean–variance utility function, and hence attaches importance to both expected net benefits and the variance between them. The regulator is assumed to prefer the monitoring strategy, which gives the highest *ex ante* utility. We first calculate the *ex ante* net benefits of each strategy. Subsequently, the associated *ex ante* utility is calculated.

## Strategy I: Simultaneous Decision on up- and Downstream Monitoring

Under strategy *I*, the regulator decides simultaneously on up-and downstream monitoring. We substitute the optimal abatement functions $\tilde{a}_i(p_i(m_i), q(d))$, derived from Eq. (4.1), in the pre-monitoring optimization problem, which gives:

$$\begin{aligned} \text{Max}E[NB(m_i, d)] = \arg\max_{m_i,d} \Big\{ & B^U(\tilde{a}_i|p_i(m_i)) \\ & + B^D(\tilde{a}_i|p_i(m_i), q(d)) - C^{MU}(m_i) - C^{MD}(d) \Big\} \end{aligned} \tag{4.2}$$

The formulation in Eq. (4.2) implies that the regulator knows that monitoring will reveal the state of emissions and downstream water quality as well as how this knowledge will affect his/her abatement decision. The optimal monitoring that results from the solution to Eq. (4.2) are denoted as $m_i^I$ and $d^I$, where the superscript $I$ denotes case $I$.

## Strategy II: Decide First on Monitoring of the Sources, then on Monitoring of Downstream Water Quality

In the second case, the decision on monitoring of the sources is taken first. Subsequently, and conditional on the outcome of the monitoring of sources, it is decided whether to monitor downstream water quality. We therefore start with the downstream monitoring decision. Given the optimal abatement functions $\tilde{a}_i(p_i(m_i), q(d))$, derived from (1), and posterior beliefs $p_i\left(e_i^0 \middle| 1\right)$ about upstream emissions, the regulator will choose downstream monitoring according to (3):

$$\text{Max}E[NB(d)] = \arg\max_d \left\{ B^D\left(\tilde{a}_i\left(p_i\left(e_i^0\middle|1\right), q(d)\right)\right) - C^{MD}(d) \right\} \tag{4.3}$$

The solution to (3) will give the optimal downstream monitoring $\tilde{d}^{II}(m_i)$ which is conditional on upstream monitoring that has already been carried out, and where the superscript *II* denotes the scenario in question. Knowing $\tilde{a}_i\left(p_i(m_i), q\left(\tilde{d}^{II}\right)\right)$ and $\tilde{d}^{II}(m_i)$, we can turn to the first stage problem, which is to decide on monitoring of emissions at the sources according to the following:

$$\begin{aligned} \text{Max } E[NB(m_i)] = \arg\max_{m_i} \Big\{ & B^U(\tilde{a}_i|p_i(m_i)) \\ & + B^D\left(\tilde{a}_i|p_i(m_i), q\left(\tilde{d}^{II}(m_i)\right)\right) - C^{MU}(m_i) \Big\} \end{aligned} \tag{4.4}$$

The solution to (4) yields the optimal upstream monitoring strategy, $m_i^{II}$, for case II.

## Strategy III: Decide First on Downstream Monitoring, then on the Monitoring of Emissions from Sources

In this case, we begin with what is usually the second last decision, i.e. the monitoring of sources. When approaching the decision to monitor sources, the regulator knows the optimal abatement functions $\tilde{a}_i(p_i(m_i), q(d))$, derived from (1) above, and has posterior beliefs about downstream water quality, $q\left(B^D(a_i)\middle|m_i, 1\right)$ when taking the decision on upstream monitoring $m_i$. Upstream monitoring choices should then maximize:

$$\begin{aligned} \text{Max } E[NB(m_i)] = \arg\max_{m_i} \Big\{ & B^U(\tilde{a}_i|p_i(m_i)) \\ & + B^D(\tilde{a}_i|p_i(m_i), q(m_i)) - C^{MU}(m_i) \Big\} \end{aligned} \tag{4.5}$$

Equation (4.5) yields the optimal monitoring of emissions at the sources, $\tilde{m}_i^{III}(d)$, which can be plugged into the first stage, i.e. making a decision on downstream water quality monitoring, which is defined by:

$$\text{Max } E[NB(d)] = \arg\max_d \left\{ B^D\left(\tilde{a}_i|p_i\left(\tilde{m}_i^{III}(d)\right), q(d)\right) - C^{MD}(d) \right\} \tag{4.6}$$

Equation (4.6) gives the optimal downstream monitoring in case *III*, for which the optimal monitoring scheme becomes $m_i^{III}, d_i^{III}$.

## Regulators Comparison of Alternative Monitoring Strategies

We assume that the regulator is risk averse and maximizes utility from alternative monitoring strategies while taking uncertainty into account. The regulator is assumed to have an exponential utility function $U$, which is assumed to be a function of the net benefits of water quality improvement, $NB$:

$$U = 1 - e^{-\alpha NB}$$

For an exponential utility function of this type, the expected utility can be expressed in terms of mean and variance of $NB$:

$$E[U] = E(NB) + \frac{\alpha}{2}\text{Var}(NB)$$

In the complete absence of water quality monitoring, the net benefit function is assumed to be:

$$NB = \sum_i k_i a_i + \sum_i l_i a_i - \sum_i c_i a_i$$

where $a_i$ is abatement at source $i$, $k_i$ and $l_i$ are the marginal benefits of improved upstream and downstream water quality, respectively, due to abatement at source $i$, and $c_i$. is the marginal cost of abatement. In the absence of monitoring, $k_i$ and $l_i$ are both random variables from the perspective of the regulator. Assuming that covariances between all states are zero, we arrive at expected net benefits in the following manner:

$$E(NB) = \sum_i E(k_i)a_i + \sum_i E(l_i)a_i - \sum_i c_i a_i$$

while the variance of net benefits is:

$\mathrm{Var}(NB) = \sum_i \mathrm{Var}(k_i)a_i^2 + \sum_i \mathrm{Var}(l_i)a_i^2$ The expected utility function in the absence of monitoring can then be expressed as:

$$E[U] = \sum_i [E(k_i) + E(l_i)]a_i - \frac{\alpha}{2}\sum_i [\mathrm{Var}(k_i) + \mathrm{Var}(l_i)]a_i^2 - \sum_i c_i a_i \quad (4.7)$$

In the Appendix, we develop a linear Taylor series approximation of the expected utility function in Eq. (4.7) as well as for the corresponding cases with monitoring strategies *I–III*.

## Numerical Simulations

To numerically simulate the model, simple generic data are used. The benefits of water quality improvements upstream and downstream, $k_i$ and $l_i$, are normalized to one. In the baseline case, upstream and downstream monitoring costs, $c_i$ and $t$, as well as abatement costs, $g_i$, are assumed to equal 0.1. The low abatement costs, compared to the benefits, implies that abatement is always worthwhile if there is a positive probability that it will improve water quality, an assumption which is convenient when the focus is on monitoring strategy choice. Finally, $\alpha$, which is the Arrow–Pratt coefficient of absolute risk aversion, is assumed to equal 0.1. Literature suggests that countries' relative risk aversion ranges around one (Gandelman and Hernandez-Murillo 2013); hence, a conservative assumption about risk aversion is used here. This is followed by an investigation into how the choice of strategy and the resulting net benefits differ with changes in assumptions about monitoring and abatement costs, and risk aversion.

Four different monitoring strategies are considered: ALL implies monitoring of both sources and recipient; NONE implies zero monitoring; EMISS implies monitoring of the sources, with a subsequent decision on whether to monitor the recipient; and RECIP implies monitoring of the recipient, with a subsequent decision on whether to monitor the sources. Thus, ALL and NONE assume a single, simultaneous decision on monitoring, both corresponding to strategy *I* in the model section, while EMISS and RECIP assume a sequential decision on monitoring, corresponding to strategies *II* and *III,* respectively, in the model section. In all cases, the monitoring decision is followed by a decision to abate or not to abate at the source. The choice of strategy is made with a focus on the trade-offs between upstream and downstream monitoring, where we abstract from other issues in the simulations that could have a bearing on the choice of monitoring strategies. For instance, the heterogeneity of sources has consequences for the choice of sources to monitor. Thus, the decision to have two potential sources merely reflects the fact that upstream monitoring typically requires a larger number of monitoring stations, compared to downstream monitoring.

## Results

This section investigates how expected utility [as defined by Eq. (4.7) above and equations (A4)–(A7) in the Appendix] and the optimal choice of monitoring strategy depend on assumptions about parameters for upstream and downstream monitoring costs, abatement costs, and risk aversion. In Fig. 4.2, the expected utility of different monitoring strategies is shown for alternative assumptions about these parameters. The purpose of the figure is to show the ranking of the monitoring strategies under different sets of parameters. Table 4.1 is complementary

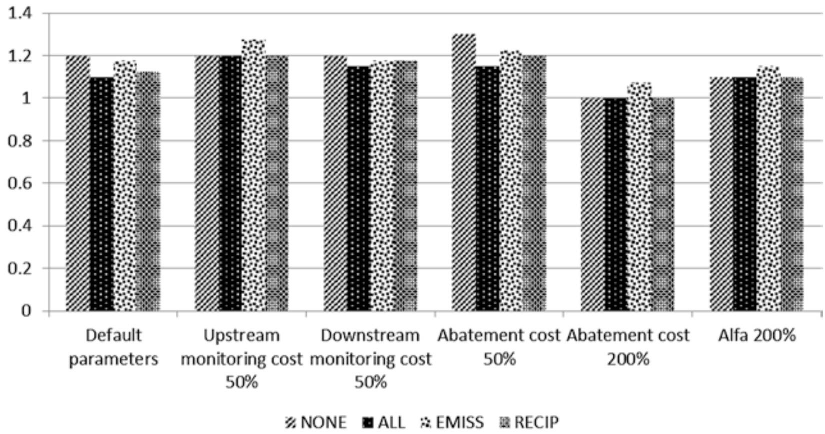


**Fig. 4.2** Expected utility for alternative strategies for monitoring, under different assumptions about monitoring and abatement cost, and risk aversion

to Fig. 4.2, and illustrates how expected utility under a particular strategy changes, compared to the baseline scenario, when parameter assumptions change. This table facilitates a discussion of the underlying reasons for the outcome shown in Fig. 4.2.

The leftmost group of columns in Fig. 4.2 shows the outcome for default parameters. In this case, the optimal choice is to neither monitor the sources nor the recipient. Complete monitoring would imply higher certainty about benefits obtained, and make it possible to abstain from abatement if emissions from the sources are zero or the recipient water quality is good. However, the benefits of monitoring do not outweigh the additional cost. A strategy involving sequential monitoring, where either the emission sources or recipient water quality are first monitored, would result in lower expected utility than no monitoring but higher than complete monitoring. Under EMISS, there is no subsequent downstream monitoring, while under RECIP, there is only subsequent upstream monitoring if the recipient is in a good state.

The second group of columns shows expected utility when upstream monitoring costs are reduced by half. The relative impact of a reduction in upstream monitoring cost on the expected utility of a strategy, which is shown in Table 4.1, is determined by the likelihood of

**Table 4.1** Change in expected utility compared to baseline, when parameters are changed

| | Default parameters | Upstream monitoring cost reduced by 50% | Downstream monitoring cost reduced by 50% | Abatement cost reduced by 50% | Abatement cost doubled | Alfa doubled |
|---|---|---|---|---|---|---|
| NONE | 1.00 | 1.00 | 1.00 | 1.08 | 0.83 | 0.92 |
| ALL | 1.00 | 1.09 | 1.05 | 1.05 | 0.91 | 1.00 |
| EMISS | 1.00 | 1.09 | 1.00 | 1.04 | 0.91 | 0.98 |
| RECIP | 1.00 | 1.07 | 1.04 | 1.07 | 0.89 | 0.98 |

upstream monitoring being carried out. The increase in net benefits is therefore the largest, 9%, in the case of strategies ALL and EMISS, where the sources are always monitored, and zero for NONE, where no upstream monitoring is carried out. Under RECIP, upstream monitoring is always carried out in the second stage, which increases expected utility substantially if the recipient is in a good state, and has a small but positive impact on expected utility if the recipient is in a bad state. Yet, expected utility increases only by 7%. The reason is that downstream monitoring is carried out under RECIP, the total cost of monitoring is therefore higher than under EMISS, where there is no monitoring of the recipient. Lower upstream monitoring cost therefore has a relatively small impact on the total monitoring cost and hence a smaller impact on expected utility. The lower impact compared to ALL is due to the difference in expected utility in the baseline case. Comparing monitoring strategies, EMISS gives the highest expected utility. Monitoring of sources in the first stage eliminates uncertainty about emissions, and reduces uncertainty about the impact of abatement of the recipient.

The third column group in Fig. 4.2 shows expected utility when the downstream monitoring cost is reduced by half. A low downstream monitoring cost has a smaller impact on expected utility as it is carried out at a single location, whereas upstream monitoring is carried out at several locations. The reduction of downstream monitoring cost by half helps save costs under ALL, thereby increasing net benefits from that strategy by 5%, while having no impact on NONE (see Table 4.1). Under EMISS, there will, of course, be no subsequent monitoring of the recipient if there are no emissions from the sources, but if emissions can be established, the decision-maker will be indifferent to monitoring the state of the recipient, as the additional benefits in terms of higher certainty barely outweigh the additional cost. Hence, the lower monitoring cost will not affect the expected utility of the strategy. Under RECIP, monitoring of the recipient will be followed by monitoring of the sources if the recipient is in a good state, but not if it is in a bad state. Expected utility is 4% higher than under default parameters. The smaller impact on the outcome compared to ALL is due to the reduced likelihood of upstream monitoring being carried out. For this choice of parameters, the expected utility under EMISS and RECIP is equal, but both are lower than under NONE.

The fourth column group in Fig. 4.2 shows expected utility when abatement costs are lowered by half. The impact on the outcome is larger for strategies where there is a higher likelihood of abatement being actually carried out. Abatement is always carried out under NONE, implying a cost saving equal to the cost reduction and increasing net benefits by 8% (see Table 4.1). This strategy is the best choice when abatement costs are halved. There is a smaller increase in expected utility under ALL and EMISS, since in both cases there is no abatement if monitoring shows zero emissions and in these cases there is no cost saving. There is an increase in net benefits by 7% under RECIP—the relatively large impact is due to both the low net benefits under default parameters and the change in strategy compared to the baseline. For these parameters, subsequent monitoring of sources is always carried out, implying more certain, and therefore higher benefits of abatement.

The fifth group of columns depicts the outcome when abatement costs are doubled. In this case, NONE, ALL, and RECIP, all give the same expected utility (see Fig. 4.2). The net benefits under EMISS are higher than under other strategies. The reason is that prior information about the existence of emissions from sources, and hence, about whether abatement is worthwhile, is of higher importance when abatement costs are larger. Compared with the situation under default parameters, the effect of higher abatement costs on the outcome is greater in scenarios where abatement is carried out with a high probability, and when expected utility is low due to a strategy being associated with high uncertainty about the achieved benefits. Thus, the impact of doubling the abatement cost is largest under NONE, where abatement is always carried out but is associated with large uncertainty. The second largest impact is under RECIP, where expected utility under default parameters are comparatively low. The smallest impact is under ALL and EMISS strategies, where expected utility under default parameters is high, and uncertainty is zero (under ALL) or comparatively low (under EMISS). While there is no subsequent monitoring of downstream water quality under EMISS, under RECIP this is done independently of the revealed state of the recipient. This implies that there is a reduction in expected utility under RECIP due to higher abatement cost, although the impact of the increase in cost is countered by the reduced uncertainty about benefits achieved.

The sixth group of columns depicts the outcome when risk aversion, as captured by the $\alpha$-term in Eq. (4.7), is doubled. Higher risk aversion affects the outcome of a strategy more if there is higher uncertainty. Therefore, there is maximum reduction in expected utility under NONE, only a small impact under EMISS and RECIP scenarios, and no impact under ALL. Under RECIP, subsequent monitoring of the sources is only carried out if the recipient is in a good state. If it is in a bad state, the decision-maker is indifferent to additional monitoring of the sources. Under EMISS, the decision-maker is indifferent to additional monitoring of the recipient if it is revealed that there are emissions from the sources. In both cases, the expected benefits of the second stage monitoring decision are thus lower than under default parameters. On the whole, the outcome is qualitatively similar as in the foregoing case: EMISS is the most advantageous, as monitoring of the sources in the first stage effectively reduces risks around both upstream and downstream benefits obtained, and abatement is made conditional on the outcome.

## Critical Thresholds for Choice of Strategy

As shown above, assumptions about parameters affect the optimal choice of monitoring strategy. This section investigates the critical threshold levels of parameters, which alter the choice of strategy. Three aspects are of interest: (i) at what parameter values do EMISS and

RECIP give higher expected utility than NONE?; (ii) at what parameter values does EMISS imply that subsequent monitoring of the recipient is carried out, if it turns out that there are positive emissions from the sources?; and (iii) at what parameter value does RECIP imply that subsequent monitoring of the sources is carried out, given that the recipient turns out to be in a good or a bad state respectively?

Table 4.2 shows that EMISS gives higher expected utility than NONE under a relatively wide range of parameter values, if the upstream monitoring cost is relatively low, or abatement costs or risk aversion are relatively high. RECIP gives higher expected utility than NONE under a much narrower range of parameter values: upstream monitoring costs must be lower, and abatement costs or risk aversion higher, for RECIP to yield a better outcome than NONE. The consequence is that RECIP will never yield a better outcome than EMISS.

Table 4.2 further shows that under EMISS, subsequent monitoring of the recipient water quality is only chosen if emissions are found at the sources, and either downstream monitoring is sufficiently cheap or risk aversion is sufficiently high. Low cost of monitoring the recipient and high risk aversion imply that monitoring is worthwhile, because both reduce the uncertainty about the impact of abatement on water quality, hence increasing the risk-adjusted benefits of abatement, even

**Table 4.2** Critical threshold values for change of strategy

| | Upstream monitoring cost (% of default) | Downstream monitoring cost (% of default) | Abatement cost (% of default) | Alfa (% of default) |
|---|---|---|---|---|
| NB(EMISS) > NB(NONE) | <0.87 | >0 | >1.25 | >1.35 |
| NB(RECIP) > NB(NONE) | <0.49 | <0.25 | >2.01 | >2.01 |
| NB(RECIP) > NB(EMISS) | Never | Never | Never | Never |
| EMISS includes monitoring of recipient | | | | |
| –state YES | Never | <0.49 | Never | >2.02 |
| –state NO | Never | Never | Never | Never |
| RECIP includes monitoring of sources | | | | |
| –state GOOD | Always | Always | Always | Always |
| –state BAD | <0.75 | Never | >1.51 | >2.01 |

though the decision to abate is not affected. Under RECIP, subsequent monitoring of the sources is always carried out if the state of the recipient is good, as within the range of parameters investigated, abatement can only be defended if it is known with certainty that the sources give rise to emissions. If the recipient is in a bad state, subsequent monitoring of the sources is only carried out if upstream monitoring costs are relatively low or abatement costs are relatively high. In the first case, the additional cost of monitoring is outweighed by the higher benefits of abatement under certainty. In the latter case, high abatement costs require that sources are monitored to evaluate whether abatement is worthwhile.

To sum up, a risk averse regulator takes into account that the degree of risk varies across monitoring strategies. The choice of monitoring strategy therefore depends on the magnitude of abatement and the monitoring cost. It includes not only picking one out of NONE, ALL, EMISS, and RECIP, but also a decision on whether there should be subsequent monitoring of the sources (under RECIP) or recipient (under EMISS). These choices affect both the likelihood of abatement and the risk-adjusted benefits of abatement, while simultaneously altering the monitoring cost, all of which have an impact on expected utility and hence on the preferred monitoring strategy.

## Conclusions and Discussion

The present study investigates the expected utility of alternative monitoring strategies, when upstream sources potentially affect both upstream and downstream water quality in a negative way. It is assumed that the monitoring of sources provides knowledge on whether emissions enter the environment, and the monitoring of the downstream recipient reveals whether water quality is actually being adversely affected. A simple sequential model is developed where the decision-maker first chooses the monitoring strategy, and subsequently on abatement, depending on the outcome of monitoring. The decision-maker is assumed to be risk averse, and prefers a certain impact of abatement compared to an uncertain impact. He/she adopts a monitoring strategy after a comparison based on expected utility. The model captures important features on water quality management problems, such as simultaneous impact of upstream sources on both upstream and downstream water quality, uncertainty about the size of emissions from individual sources, and difficulties in determining the environmental status of coastal waters.

The theoretical analysis shows that upstream monitoring reduces uncertainty more efficiently than downstream monitoring, while being more expensive. Thus, the relative advantage of upstream and downstream monitoring depends on the net benefits of abatement with respect to upstream and downstream water quality, given the decision-maker's risk averseness and monitoring costs.

The model is simulated numerically with an aim to understand the role of upstream and downstream monitoring costs, abatement costs, and risk aversion with respect to the choice of monitoring strategy. The results suggest that the optimal choice is to either carry out no monitoring, or to first monitor the sources, and based on the outcome, to decide whether to proceed with downstream monitoring. The latter strategy is preferred if the upstream monitoring cost is relatively low, or abatement costs or risk aversion are relatively high.

The above analysis has several limitations, such as exclusion of operation and management costs of monitoring and abatement, and of naturally variable environmental conditions. For the simulations, it is assumed that emission sources are homogeneous, and that either complete or no monitoring of the sources is carried out. Possible future extensions of the analysis could therefore include selective monitoring of heterogeneous potential emission sources and analysis of conditions where not only the outcome of monitoring but also the presence of uncertainty could imply that abatement is not carried out.

Given the highly stylized nature of the present model, it is not suitable for drawing strong policy conclusions. However, the analysis suggests that policymakers should at least evaluate the overall benefits provided by upstream and downstream monitoring, respectively, for water quality policies. This could entail, for example, a comparison of monitoring costs, and improved knowledge by monitoring, across completely different programmes, such as monitoring of nutrient loads at river outlets with monitoring of farmer compliance with agri-environmental programmes. Such monitoring programmes currently operate without any integration, and there are no explicit trade-offs being made between efforts spent on different monitoring activities.

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## Appendix: Development of the Expected Utility Function for a Linear Programming Model

To obtain a function suitable for linear programming, we first note that

$$CV(k_i) = \frac{\sqrt{\mathrm{VAR}(k_i)}}{E(k_i)} \quad \text{and} \quad CV(l_i) = \frac{\sqrt{\mathrm{VAR}(l_i)}}{E(l_i)}$$

We can thus write: $\mathrm{Var}(k_i) = [CV(k_i)E(k_i)]^2$, and $\mathrm{Var}(l_i) = [CV(l_i)E(l_i)]^2$, thereby obtaining

$$E[U] = \sum_i [E(k_i) + E(l_i)]a_i - \frac{\alpha}{2}\sum_i [\mathrm{Var}(k_i) + \mathrm{Var}(l_i)]a_i^2 - \sum_i c_i a_i =$$
$$\sum_i [E(k_i) + E(l_i)]a_i - \frac{\alpha}{2}\sum_i \left[[CV(k_i)E(k_i)]^2 + [CV(l_i)E(l_i)]^2\right]a_i^2 - \sum_i c_i a_i \tag{A1}$$

Making a first-order Taylor series expansion of the second term in the above expected utility function around a chosen point $a_i'$ gives:

$$\begin{aligned} E(U) = & \sum_i [E(k_i) + E(l_i)]a_i \\ & - \frac{\alpha}{2}\sum_i \left[[CV(k_i)E(k_i)]^2 + [CV(l_i)E(l_i)]^2\right]a_i'^2 \\ & + 2\cdot\sum_i \left[[CV(k_i)E(k_i)]^2 + [CV(l_i)E(l_i)]^2 a_i'\left(a_i - a_i'\right)\right] \\ & - \sum_i c_i a_i \end{aligned} \tag{A2}$$

which can be used in a linear programming model.

It is assumed that $a_i = [0, 1]$. We choose to do an approximation of (A2) around $a_i^{'} = 1$. We then assume that $\bar{k}_i = 1$ and $\bar{l}_i = 1$, implying that $E(k_i) = 0.5\bar{k}_i$ and $E(l_i) = 0.25\bar{l}_i$, where $\bar{k}_i$ and $\bar{l}_i$ are the expected benefits achieved if $e_i^0 = 1$, and $s = B$, reflecting the likelihood of $e_i$ and $s$ being 0 or 1. We then get:

$$\begin{aligned} E(U) = & \sum_i [E(k_i) + E(l_i)]a_i \\ & - \frac{\alpha}{2} \sum_i \left[ [CV(k_i)E(k_i)]^2 + [CV(l_i)E(l_i)]^2 \right] a_i'^2 \\ & + 2 \cdot \sum_i \left[ [CV(k_i)E(k_i)]^2 + [CV(l_i)E(l_i)]^2 a_i' \left(a_i - a_i'\right) \right] \\ & - \sum_i c_i a_i \end{aligned}$$

$$\begin{aligned} E(U) = & \sum_i [0.5 + 0.25]a_i \\ & - \frac{\alpha}{2} \sum_i \left\{ \left[ \left[ \frac{\sqrt{0.5}}{0.5} \cdot 0.5 \right]^2 + \left[ \frac{\sqrt{0.5}}{0.25} \cdot 0.25 \right]^2 \right] 1^2 \right\} \\ & + 2 \cdot \sum_i \left[ \left[ \frac{\sqrt{0.5}}{0.5} \cdot 0.5 \right]^2 + \left[ \frac{\sqrt{0.5}}{0.25} \cdot 0.25 \right]^2 \cdot 1 \cdot (a_i - 1) \right] \\ & - \sum_i c_i a_i = \end{aligned}$$

$$E(U) = \sum_i 0.75a_i - \frac{\alpha}{2} \sum_i \left[ (0.5 + 0.5) + 2 \sum_i (0.5 + 0.5)(a_i - 1) \right] - \sum_i c_i a_i, \tag{A3}$$

This collapses to:

$$E(U) = \sum_i 0.75a_i - \sum_i \frac{\alpha}{2} - \sum_i c_i a_i, \tag{A4}$$

where the two first terms express the risk-adjusted net benefit if abatement is carried out, and the last term the certain cost of abatement. Thus, there is a fixed risk penalty equal to $\propto/2$ for each source when there is no monitoring at all.

The expected utility function derived by linear Taylor series approximation can, in a similar manner be derived for the case when only upstream monitoring has been made. If emissions from the sources are positive, we get:

$$E(U) = \sum_i 1.5a_i - \sum_i \frac{\alpha}{4} - \sum_i c_i a_i, \tag{A5}$$

while if they are negative, abatement wl not be carried out. With only downstream monitoring, showing that the recipient is in a bad state, we get:

$$E(U) = \sum_i 1.0a_i - \sum_i \frac{\alpha}{4} - \sum_i c_i a_i, \tag{A6}$$

and when in a good state, we get:

$$E(U) = \sum_i 0.5a_i - \sum_i \frac{\alpha}{4} - \sum_i c_i a_i \tag{A7}$$

Under complete certainty, the expected utility is determined in a straightforward manner, by $\bar{k}_i$., $\bar{l}_i$, and $c_i$, see equation (A1).

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# 5

# Public Policies Towards Marine Protection: Benchmarking Estonia to Finland and Sweden

Tea Nõmmann and Sirje Pädam

## Introduction

The EU Marine Strategy Framework Directive (MSFD) aims at protecting the marine environment across Europe. It requires EU member states to put in place measures to achieve Good Environmental Status (GES by 2020, through the development of national marine strategies. Since eight of the nine coastal countries of the Baltic Sea are EU member states, MSFD provides substantial geographical coverage.

In its objective to protect the marine environment, the Directive also calls for due consideration of sustainable development and the

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assessment of social and economic impacts of proposed measures. MSFD explicitly asks member states to ensure that planned measures are cost-effective and technically viable, and that impact assessments, including cost-effectiveness analysis (CEA) and cost-benefit analysis (CBA), have been carried out prior to their introduction. Based on these analyses, the most cost-effective and beneficial measures can be selected. Since the motivation of the measures is to achieve environmental targets, economic analyses also serve as grounds for the application of exemptions by member states.

The Directive introduces the ecosystem approach to the management of human activities affecting the marine environment. As European seas are shared by many countries, MSFD stresses the importance of cooperation at the regional level and urges the coordination of implementation of MSFD via existing Regional Sea Conventions. For the Baltic Sea region, the Helsinki Convention of 1974 (HELCOM), provides a coordination platform.

Carrying out CEA and CBA on the marine environment is a challenging task. Sea ecosystems are complex and there are substantial knowledge gaps about the impacts on ecosystems due to changes in human activity. Other gaps include the welfare implications of improvements in marine ecosystems. Given that the first cycle of MSFD is presently under way, countries have had to adopt an experimental approach. In order to build knowledge for the next cycle, valuable inputs are expected from reviewing and comparing the CEA and CBA carried out by Estonia, Finland and Sweden, three countries that share the same marine area but have different prerequisites concerning administrative and research capacity.

The purpose of this chapter is to compare the CEA and CBA of the proposed new measures of the National Marine Strategies of Estonia, Finland and Sweden. The analysis is mainly based on the study of documents. The comparison covers each country's process of choosing new measures and the chosen approaches for carrying out CEA and CBA. The framework of the analysis is based on the analytical steps of CEA and CBA.

Section "Theoretical Background" provides a theoretical background and in Sect. "Empirical Background", we present an empirical background of the national Programme of Measures (PoM). Section "Cost-Effectiveness" describes the CEA carried out in the selected countries and in Sect. "Cost-Benefit Analysis", we present the review of CBA. This is followed by a conclusion along with recommendations for the coming cycles of MSFD.

## Theoretical Background

Member states are requested by MSFD to show that the suggested new measures are cost-effective and prior to the introduction of any new measure, member states need to carry out CBA (European Commission 2008). Since these two kinds of economic analysis aid makers while evaluating policy alternatives, the requirements imply sound policy analysis. However, EU legislation does not provide guidance on what CEA and CBA should involve or how to quantify and find values for benefits and costs. The experiences from various countries on implementing environmental CEA and CBA have been reported by background and working group documents (European Commission 2015; Working Group on Economic and Social Assessment 2010).

### Cost-Effectiveness Analysis

CEA is an exercise in comparing the costs and outcomes of different actions, thus assisting policy makers in choosing measures that can reach the policy goal at minimum cost. In order to define a cost-effective allocation of measures, Elofsson (2010: 50) recommends the following three steps. "The first step is to interpret the politically determined environmental target into a measurable target indicator if the target is broadly defined. The second is to calculate costs of measures at the sources and the third to quantify the impact of measures on the target." There is a substantial complication in the first step of a CEA of the Programme of Measures (PoM), owing to the multidimensionality of the environmental objective, i.e. achievement of Good Environmental Status (GES). In order to define GES, the Directive describes 11 qualitative descriptors (see Table 5.1).

**Table 5.1** Qualitative descriptors for determining good environmental status

| Descriptors | Abbreviation |
|---|---|
| Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions | D1 |
| Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystems | D2 |
| Populations of all commercially exploited fish and shellfish are within safe biological limits, exhibiting a population age and size distribution that is indicative of a healthy stock | D3 |
| All elements of marine food webs, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity | D4 |
| Human-induced eutrophication is minimized, especially its adverse effects thereof, such as losses in biodiversity, ecosystem degradation, harmful algae blooms and oxygen deficiency in bottom waters | D5 |
| Seafloor integrity is at a level that ensures that the structure and functions of ecosystems are safeguarded and benthic ecosystems in particular, are not adversely affected | D6 |
| Permanent alteration of hydrographical conditions does not adversely affect marine ecosystems | D7 |
| Concentrations of contaminants are at levels not giving rise to pollution effects | D8 |
| Contaminants in fish and seafood for human consumption do not exceed levels established by Community legislation or other relevant standards | D9 |
| Properties and quantities of marine litter do not cause harm to the coastal and marine environment | D10 |
| Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment | D11 |

*Source* Marine Strategy Framework Directive (MSFD) 2008/56/EC

The descriptors are broadly defined in qualitative terms. For this reason, it is helpful to define measurable target indicators. The Directive has introduced criteria and indicators to help with the interpretation of the descriptors as well as appraising their current status regarding GES targets. The quantitative operationalization of GES has been left to the member states (Oinonen et al. 2016). It is, therefore, expected that prior operationalization of GES is useful for the purpose of CEA.

Estonia, for example, has used a number of indicators among the GES targets that define the gap with respect to each descriptor. There are a total of 44 indicators describing targets. Several descriptors are defined by more than one target. However, this becomes more complicated because of potential interlinkages between the indicators, descriptors and the lack of guidance on how to weigh gaps in the attainment of different GES descriptors (Oinonen et al. 2016).

In addition, Oinonen et al. (2016) point out that uncertainties and the lack of multidisciplinary models of sea ecosystem management, call for expert-based qualitative assessments.

In a review of economic analyses concerning marine and water management, Söderholm et al. (2015) note that a common approach among previous studies has been a focus on measures to reach environmental (GES) targets rather than on implementation. Most often, there is only a weak link between measures and how they are to be put into practice, i.e. the policy instruments. The choice of policy instruments has an influence on CEA as it affects both costs and the behavioural response. The implications of taxes differ to a significant extent from those of information. Experiences from plastic bag regulation show that policy instruments like taxes, bonuses and information imply significant variation in effectiveness (Convery et al. 2007; Collins et al. 2003; Homonoff 2013).

Another observation made by Söderholm et al. (2015) is that the costs of measures are evaluated ex ante rather than based on ex post analysis. Ex ante evaluation is impaired by greater uncertainty as it does not draw from experience.

## Cost-Benefit Analysis

CBA is a tool to assess whether the economic value that is expected to follow from a particular action is in balance with the associated costs. It is a method of social appraisal, and is being used according to criteria derived from welfare economics. The most common purpose of CBA is to provide ex ante policy evaluation. This is also the case with the CBA of PoM.

In the widely used textbook on CBA, Boardman et al. (2011) outline nine steps involved in CBA. Hanley and Spash (2003) describe an alternative structure of CBA in eight steps. Both approaches are similar and the steps overlap to a significant degree. Hanley and Barbier (2009) suggest a six-step approach that has recently been applied by Börger et al. (2016) while comparing CBAs of PoMs in UK, Spain and Finland. In order to be comprehensive, we choose the nine-step structure as the framework for comparing CBAs. The nine steps are as follows:

Step 1: Specify the set of alternative projects;
Step 2: Decide whose benefits and costs count (standing);
Step 3: Catalogue the impacts and select measurement indicators;
Step 4: Predict the impacts quantitatively over the life of the project;
Step 5: Monetize all impacts;
Step 6: Discount benefits and costs to obtain present values;
Step 7: Compute the net present value (NPV) of each alternative;
Step 8: Perform sensitivity analysis;
Step 9: Make a recommendation based on the NPV and sensitivity analysis.

The first step is to define alternative projects or policies. In the case of the PoM, this is applicable to the measures. It is equally important to define the business-as-usual scenario, which outlines the choice of not implementing the project or policy. In their review of CBAs, Söderholm et al. (2015) found that there are substantial challenges related to the definition of the business-as-usual scenario. They point out that unless it is clear what is meant by the choice of "doing nothing", policy alternatives also become indistinct. Since the PoMs are national, it is reasonable to expect that the standing is the population of the country in question (step number 2). At the same time, national population might be too narrow if measures give rise to cross-border benefits or costs.

Identification of the impacts and selecting measurement indicators is the third step of CBA. In this step, the costs and impacts regarding the marine environment are, in principle, available from the CEA. However, determining the benefits to humans from the improvement of marine ecosystem requires additional methods. In order to cover further aspects, for example, indirect costs of measures, criteria for households and businesses should be added. The fourth step involves predicting the impacts and expressing them in quantitative terms. For the same reason as with CEA, CBA will run into difficulties because of the multidimensionality of GES. The fifth step is valuation, i.e. monetization. For the monetization of benefits, it must be possible to measure the value of the improvement of the environment. Issues such as clean beaches, protection from contaminants or any of the descriptors shown in Table 5.1 need to be interpreted, in terms of either willingness to pay or avoidance of degradation costs. Estimates of the monetary costs of measures are available from the CEA. These cost estimates need to be complemented by indirect costs.

Through monetization, all impacts become commensurable. It becomes possible to express the benefits and costs of each choice. However, comprehensive monetization is seldom possible when it comes to environmental impacts. The improvement of the Baltic Sea's marine environment is a non-market good and its value cannot be easily derived from ordinary market activities. In addition, there are knowledge gaps between the impact of the improvement in marine ecosystem services and their implications on welfare indicators. As a result, finding the appropriate monetary values will prove to be either too complex or too costly. For these reasons, CBA is often performed in terms of a qualitative assessment (Söderholm et al. 2015). The steps that follow in the list (discounting, see steps six and seven) require monetization, and are omitted here.

Step 8 includes sensitivity analysis, which is meant to test how variations among uncertain variables affect the result. In qualitative CBA, this can be done by presenting intervals of the outcome or by illustrating how ranking is affected by uncertainty. The final step, the ninth step, is to make recommendations. Doing this on the basis of qualitative CBA is more challenging than on the basis of monetized CBAs. Although Söderholm et al. (2015) point out that there are good quality examples of previous CBAs, they refer to qualitative CBAs which list impacts in various dimensions, without aggregating benefits and costs. In these circumstances CBA provides little or no help in policy choices.

### Beyond Economic Analyses

The economic analyses make up one part of the process of approval of the national marine strategy. Beside CEA and CBA, it includes scientific appraisal, public discussions and consultations with public agencies and ministries. Proposals of new measures originate primarily from scientific gap analyses of the status of marine ecosystems relative to GES. In this work, protection, cleaning up or the reduction of pollutants have been identified as essential in order to reach some target or indicator. The proposals of new measures may also be influenced by expectations of what is acceptable to policymakers. Another factor that contributes to the choice of measures is the tight timelines for approval, which reduce the time available to analyse and design relevant policy instruments.

## Empirical Background

Sources of empirical data include background documents of economic analyses of the Marine Strategies of Estonia, Finland and Sweden as well as the approved programmes of measures of Finland and Sweden (SA Stockholmi Keskkonnainstituudi Tallinna Keskus, Tartu Ülikooli Eesti Mereinstituut ja Tallinna Tehnikaülikooli Meresüsteemide Instituut 2016; Havs-och vattenmyndigheten 2015a, c; HELCOM 2016; Oinonen et al. 2015, 2016; Vretborn 2015).

### Measures by Descriptor

A comparison across national marine strategies shows that measures to protect marine biodiversity and food webs via new marine protected areas or better management of those areas are most frequently suggested (especially in Sweden and Finland) (see Fig. 5.1). Measures to reduce eutrophication are emphasized by Finland. Estonia and Sweden suggest only a few new measures in addition to those of the Water Framework Directive.[1] All countries find it important to impose additional measures on commercially exploited population of fish and shellfish. While

| | Estonia | Finland | Sweden | |
|---|---|---|---|---|
| D1. Biodiversity & D4. Foodwebs | 2 | 6 | 6 | 14 |
| D5. Eutrophication | 2 | 8 | 3 | 13 |
| D3. Populations of commercially exploited fish | 4 | 2 | 6 | 12 |
| D8. Hazardous subst. & D9. Contaminants | 2 | 6 | 4 | 12 |
| D10. Marine litter | 3 | 1 | 5 | 9 |
| D2. Non-indigenous species | 2 | 0 | 3 | 5 |
| D6. Seafloor integrity | 0 | 2 | 2 | 4 |
| D11. Underwater energy & noise | 1 | 3 | 0 | 4 |
| D7.Hydrographical conditions | 0 | 1 | 2 | 3 |
| | 16 | 29 | 31 | 76 |

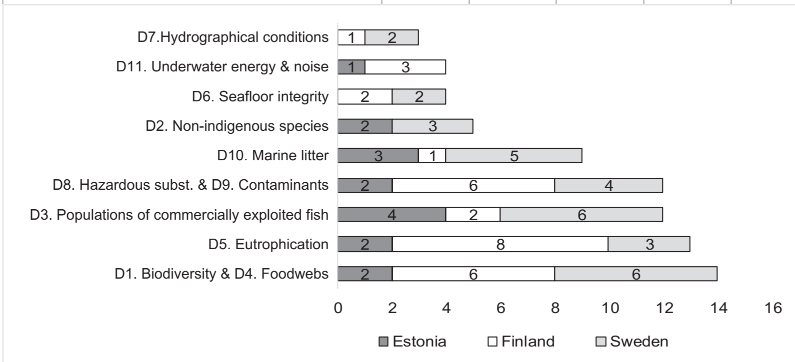


**Fig. 5.1** Distribution of measures of the national marine strategies by descriptor

Sweden and Estonia recommend fishing restrictions, Finland proposes national strategies for several fish species in need of protection.

The management of risks from hazardous substances is emphasized by all countries. While the focus in Estonia is on strengthening preparedness and training for combating possible accidents, as well as on minimizing risks during bunkering, in Sweden and Finland, the emphases is on the identification of hazardous substances (chemicals, pharmaceuticals) in water and sediments and providing guidance to relevant stakeholders. All countries mention the reduction in the use of plastics and plastic bags as important priorities. According to the descriptions of measures, this is to be achieved mainly by awareness-raising information activities. Seafloor integrity measures are developed by Finland and Sweden. As for non-indigenous species, Estonia and Sweden include measures to improve awareness of the problem. Measures for identifying underwater noise issues are being developed by Estonia and Finland. Finally, regarding hydrographical conditions, Sweden suggests measures to prepare guidelines for marine-related impact assessment and guidelines for municipal marine spatial planning, while Finland foresees measures to improve coastal flow conditions.

## Classification of Measures

The classification of measures according to the Directive is rather complex.[2] Based on an analysis of countries' planned measures, the present study employs a simpler classification based on economic theory suggested by the Swedish Environmental Protection Agency (2012): information (e.g. labelling, education, influence public opinion); administrative (e.g. laws, standards, agreements); research and development (R&D e.g. development, demonstration, assessment of technology); and economic (e.g. taxes, subsidies, grants, permit trade). These measures are often based on combinations of various instruments. Söderholm et al. (2015) observe that these measures focus on reaching the environmental targets (GES) rather than on implementation. In the classification of measures, we have merely selected the main types. Table 5.2 shows the division of measures by type in national PoMs.

The preferred focus in Sweden is on information, including education and awareness-raising measures. While this is also common in Estonia and Finland, the majority of measures are administrative in nature. In Sweden, these are the second most preferred choice, followed by R&D-related measures. Countries have different strategies towards

**Table 5.2** Measures by type in the national programme of measures

| Type of measures | Estonia | | Finland | | Sweden | |
|---|---|---|---|---|---|---|
| Information | 6 | 38% | 8 | 28% | 14 | 45% |
| Administrative | 7 | 44% | 10 | 34% | 7 | 23% |
| R&D[a] | 2 | 13% | 10 | 34% | 7 | 23% |
| Economic | 1 | 6% | 1 | 3% | 3 | 10% |
| Total number of measures | 16 | 100% | 29 | 100% | 31 | 100% |

[a]*Note* R&D measures refer mainly to piloting and development activities

presenting further needs of research due to current knowledge gaps, which partly explains the low number of R&D measures in Estonia. Estonia suggests 21 topics for further research in addition to its new measures. Finland proposes research stemming from Water Framework River Management Plans that have implications on marine issues. In Sweden, the discussion on further research needs is broader. Economic measures are in clear minority.

## Cost-Effectiveness

In order to compare the CEA undertaken by Estonia, Finland and Sweden, we first give a brief overview of the approach adopted by each country. The review is based on how the countries have appraised effects, estimated costs and presented the results of CEA.

### Estonia

In Estonia, the process of developing a national PoM was coordinated by the Ministry of Environment and the work was carried out between fall 2014 and summer 2016 by a consortium consisting of experts in the fields of marine environment at the Marine Systems Institute of the Tallinn University of Technology and the Institute of Economics, the Estonian Marine Institute of the University of Tartu and SEI Tallinn (the Tallinn Centre of the Stockholm Environment Institute). At the time of writing, the Estonian PoM is in the process of inter-ministerial approval with the aim of adoption in 2016.

*Assessment of effectiveness*: During the development process of the PoM in 2014–2015, the assessment of the environmental status of the Estonian marine areas carried out in 2012 was revised. Environmental targets ensuring the achievement of GES were specified and pressures affecting the environmental status were assessed together with possible changes until 2020 by the experts. During this revision process, environmental targets of the descriptors were quantified as far as possible. However, the definition of quantifiable targets requires further study.

Based on a gap analysis of existing measures and the distance to GES, a total of 46 new measures were proposed initially by the experts to achieve GES. The 46 candidate measures were further analysed in three separate ad hoc working groups consisting of relevant officers, experts and stakeholders. Working groups were formed based on the grouping of the descriptors: (1) biodiversity, fisheries and invasive species; (2) eutrophication and hazardous substances; and (3) marine litter, underwater noise and energy. The task of the working groups was to assess the technical feasibility and effectiveness of the candidate measures.

The effectiveness of the new measures was assessed against Estonia's 44 GES targets. Participants in the working groups had to give their expert opinions on the extent to which each measure would help achieve GES, i.e. to reduce the gap between the business-as-usual trend and the GES target. The expected impact of the measure was assessed with respect to the relevant subset of the 44 targets. A seven-point scale was used: 1—there is no impact; 2—the impact is very small; 3—the impact is small; 4—the impact is average; 5—the impact is important; 6—the impact is very important; and 7—GES will be achieved fully. The effectiveness of each measure was assessed independently against the individual GES target and no interrelated impacts between measures were assessed. The assessment was done in groups and in the case of difference of opinions among experts, the results were discussed until consensus was achieved. The overall effect of the measures was derived from the highest score given to each measure by the experts. This is because it proved very difficult to determine any meaningful way to describe the contribution of an individual target to overall GES. No weighting or summing up of the scores was used.

In parallel, experts commented and gave feedback on the proposed new measures. In several cases, they recommended further research in order to determine the extent of the problem as well as to define activities or policy instruments suitable for dealing with the problem. In few cases, measures were combined, and finally, some proposed measures were re-classified as existing but not yet implemented—for example, better enforcement—which implied that they were not new measures according to the Directive.

After this assessment process, 21 new measures were pending further analysis. These, together with the results of the initial economic analysis, were presented during public discussions in September–October 2015. After the public discussions and during the final round of internal consultations in the ministries to approve the list of new measures and their planned costs, the number of new measures was further reduced to 16.

*Assessment of costs*: The identification of the cost of measures was carried out via interviews, desktop studies and expert assessment. This work was carried out by a subgroup of the consortium team. Initially, all direct costs of the public institutions were assessed, i.e. personnel costs, subcontracting costs and investment costs. At a later stage, personnel costs were excluded as these were considered as part of the normal work of public officials. Thus, only subcontracting and investments costs, when relevant, were included. The cost estimates were mostly experience-based, and put together in a bottom-up process. In some cases, ex ante studies were consulted. No ex post studies were available.

*Presentation of cost-effectiveness*: Based on the effectiveness score and the costs, cost-effectiveness was assessed and measures were ranked based on the assessment. For purposes of presentation, cost-effectiveness was grouped into three categories: high, average and low. But given the rather short list of measures and their relatively modest costs (excluding two expensive fisheries measures), the grouping does not provide high information value.

## Finland

In Finland, the process of developing the national PoM was coordinated by the Ministry of Environment and the work was carried out in cooperation with the Ministry of Agriculture and Forestry and the Ministry of Transport and Communication as well as other public agencies. Several working groups were established to develop the PoM during 2013–2014. The working groups consisted of planning and other relevant officials from government organizations, researchers and representatives of non-governmental organizations. In all, over 60 people participated in the preparation of the national PoM. The Finnish government adopted the PoM in December 2015.

*Assessment of effectiveness*: The working group whose mandate was to carry out the CEA was established after the candidate measures had been identified by other working groups (Oinonen et al. 2016). The candidate measures were identified on the basis of gap analysis between current status and GES. Further selection of candidate measures was based on their technical feasibility and social acceptability. In all, 31 candidate measures were presented to the working group that was responsible for CEA.

Due to the lack of applicable economic-ecological models for several descriptors, it was decided to employ expert knowledge and structural interviews in order to assess the effects of the measures. According to Oinonen et al. (2016), effectiveness was defined as a probability distribution describing the likelihood that a candidate measure will achieve a given proportion of the gap between the present environmental status and the threshold for the GES. The method was chosen as other qualitative approaches were not supported by the experts in subgroups. When assessing the effectiveness of the measures, it was assumed that impacts are mutually independent, even though in reality the descriptors are interrelated. Data collection was tested in various ways (including pilot email questionnaire), though finally group interviews were conducted in predefined thematic expert working groups.

Questions were asked on the common understanding of the gap to arrive at the GES of the descriptor, understanding of the cause-effect mechanism of the measure, effectiveness and cost of each measure. In a similar way, questions were put forth about the difficulty of assessing effectiveness as well as the joint and cross-effects of candidate measures. Each expert was given seven votes per measure. The results were discussed by the group and the facilitator captured the variation among opinions—the wider the variation, the higher the uncertainty of the effects of the measure. The results were determined by consensus reached within each group after discussion.

The effectiveness of the candidate measures of the Finnish PoM was defined using discrete conditional probability distribution (Oinonen et al. 2016). The distribution and related scores were as follows: 1—the measure does not have impact (score 0), 2—the measure bridges up to 12.5% of the gap (score 0.063), 3—the measure bridges 12.5–25% of the gap (score

0.188), 4—the measure bridges 25–50% of the gap (score 0.375), 5—the measure bridges 50–75% of the gap (score 0.625), 6—the measure bridges 75–100% of the gap (score 0.875) and 7—the measure achieves GES by 2020 (score 1.000). The effects of the measures were appraised in relation to each descriptor. No target indicators were employed.

*Assessment of costs*: The same group of experts was engaged for the assessment of costs. The conditional probability distribution method was used during appraisals. Both direct and indirect costs were included. Costs were distributed into seven classes: €0–0.1 million (score 0.05), €0.1–0.5 million (score 0.3), €0.5–1 million (score 0.75), €1–5 million (score 3), €5–10 million (score 7.5), €10–50 million (score 30), over €50 million (score 50). Each expert had seven votes per measure. The results of the scores on costs were discussed by the group and the facilitator captured the variation in votes. As for the effects, uncertainty was captured based on the range of difference between expert opinions. The cost estimates are solely based on expert elicitation. Other sources were not consulted.

*Presentation of costs-effectiveness*: Ranking of measures based on cost-effectiveness was carried out by using cost-to-effect ratios of the estimates of expected costs and expected effectiveness. Joint effectiveness of two or more measures in closing the gap of a descriptor was calculated for a large number of combinations. Based on these cumulative distributions, various budget constraints were applied in order to identify alternative packages or combinations of measures with high probability of achieving GES. The results showed that dropping two of the least cost-effective measures would not affect the probability of achieving GES. One of the measures that was dropped had a low impact on merely one of the descriptors, and the other was the most expensive among all measures. The analysis also shows that it is possible to capture 60–70% of the maximum joint effect if the budget is cut down from €90 million to €20 million.

## Sweden

In Sweden, the process of developing the national PoM was coordinated by the Swedish Agency for Marine and Water Management (SwAM)[3] in collaboration with relevant authorities and scientific experts. The proposed new measures mainly affect public authorities and municipalities. Most measures are directed towards SwAM's own structure. According to Swedish legislation, SwAM has the authority to regulate fishing and develop guidelines on how marine environments and streams may be used. The national marine strategy was approved by SwAM in December 2015.

*Assessment of effectiveness:* Following the guidelines proposed by MSFD, measurable targets of GES in Swedish marine waters were based on a set of national indicators, including habitats of key species and the input of nutrients to different sea areas. As an additional dimension, Sweden applied environmental standards stated in legislation, which outline the desired condition of the marine environment. In order to appraise the impact of the Swedish PoM on the marine environment, Sweden employed expert assessments. Experts from SwAM and the Swedish Institute for the Marine Environment (SIME) were engaged for this purpose. The appraisal of the impacts of measures was carried out in reference to a business-as-usual scenario until 2020. Experts assessed the level of improvement from the business-as-usual scenario to GES, i.e. complete attainment of the environmental target, as well as the improvement from the business-as-usual scenario as a result of measures. This was done measure by measure on a four-point scale. In order to consider uncertainties, an interval of low and high impact was provided by the experts.

*Assessment of costs*: Costs were put together by another team at SwAM. Most cost estimates were experience based. To some extent, ex post information was available, for example, costs concerning municipal waste collection. The measures in the Swedish PoM entail, to a great extent, direct costs to the public sector, for example, subsidies for beach cleaning projects, the development of tools to make available information on non-indigenous species and costs of personnel. Indirect costs were also collected. Measures that contain indirect costs include, for example, fishing restrictions, which entail indirect costs on commercial and recreational fishing. However, only the direct costs were used in the CEA.

*Presentation of cost-effectiveness*: For the purpose of the analysis, the qualitative expert assessments were compared to direct costs, which were split into four categories. Four-times-four matrices with costs and effects were used to illustrate outcomes. High-cost measures with small impacts were judged not cost-effective and assigned 1 point on a scale of 1–4. High-cost measures with large impacts were assessed as potentially cost-effective and this was also the case with low-cost measures with small impact. Most measures were found to be potentially cost-effective: 19 of 31 measures received 3 points each. Two measures were assigned 2 points each and assessed as possibly cost-effective; another two were appraised based on their cost per hectare. Eight measures lack assessments. These include measures for knowledge and capacity building. The results of the analysis did not lead to the exclusion of measures or any other adjustments.

## Comparison

The broad definition of descriptors was a challenge for all countries. In Finland, assessments were done towards descriptors, while Estonia and Sweden used a richer set of indicators. Although some target indicators were quantitative, the lack of scientific knowledge placed limits to the application of quantitative assessments.

Estonia and Sweden put together the costs of measures in a bottom-up process. The cost estimates were primarily experience-based. Hardly any relevant ex post analysis seems to have been available. The Finnish approach differed, as expert assessments were applied to estimate costs. All countries presented costs in monetary terms, but for the purpose of the analysis, monetary estimates were expressed using points. This is reasonable considering that the effectiveness of measures was assessed qualitatively. The Finnish qualitative approach of probabilistic assessment differed from the other two countries and allowed for much richer analyses.

One challenge faced by all CEAs was the existence of only one or few alternative measures for closing a certain gap. For this reason, it remains uncertain whether the chosen measures provide the lowest cost alternative. Ranking can only be done on an overall level, which implies that GES targets are of equal weight. For example in the Finnish PoM, the measure to concentrate deposition of sediments was ranked highest in terms of cost-effectiveness. In Sweden's and Estonia's CEAs several measures received the same score. The latter approaches only make possible rough classification of cost-effectiveness.

## Cost-Benefit Analysis

In this section, we review the CBA following the steps suggested by Boardman et al. (2011). Preparations for CBA took place during 2014–2016. The review is based on written reports prepared by the CBA teams in Estonia, Finland and Sweden.

### Specification of Alternatives

The CBA conducted by the three countries used different approaches in their specification of alternatives. The Finnish CBA applied aggregation of measures. This implies that there are two policy choices—implementing and not implementing the PoM. The business-as-usual scenario is the same as in the CEA. The Estonian and the Swedish CBAs appraise each measure separately, thus suggesting that there is a policy choice per measure. Both the Estonian and Swedish CBAs describe the business-as-usual scenario and expected developments until 2020. All three CBAs assume a national perspective when it comes to counting costs and benefits.

### Choice of Impact Categories, Predicting the Impacts and Monetization

There is variation between the CBAs regarding the choice of impact categories. While Estonia and Sweden expand the set of impact categories as compared to the CEA, Finland only appraises the aggregate benefits of a subset of descriptors.

The Estonian CBA uses the CEA as an input in the analysis concerning the costs and impacts of measures in the environmental indicators. Valuation is based on expert assessment regarding 12 criteria. These include the impact on different stakeholders and sectors, as well as the complexity of implementation and the time from implementation until the impact of the measure takes effect. The assessment of each criterion is done on a five-point scale. The criteria pertaining to stakeholders and sectors are, to some extent, overlapping, which implies that there is a risk of double counting. Another issue is that the impacts relative to the business-as-usual scenario are not clearly reported. The Estonian CBA makes no attempt to monetize the benefits, as no national background studies on relevant topics were conducted prior to developing a PoM. A small number of international studies that include Estonia have been carried out. The results of these were not available at the start.

The impact categories of the benefits of the Finnish CBA are based on the five descriptors that cannot be achieved in the business-as-usual scenario. These include: biodiversity (D1), marine food webs (D4), human-induced eutrophication (D5), concentrations of contaminants (D8) and contaminants in fish and other seafood (D9). Monetization is based on benefit transfer from prior valuation studies concerning the benefits of coastal habitats (D1 and D4) and nutrient reduction in the Baltic Sea (D5). Monetary values from previous studies are scaled down in accordance to the expected percentage of gap closure vis-à-vis GES. The percentage is based on the expert assessments of CEA.

The Swedish CBA applies the qualitative assessment of CEA regarding the impact of each measure on a set of environmental standards. Benefit transfer of consumer surplus is extracted from Ahtiainen et al. (2014). In a similar vein as the Finnish CBA, benefits are scaled vis-à-vis the percentage estimates of the contribution to gap closure with respect to GES. Additionally, the Swedish CBA assesses benefits from measures on two industries: commercial fishing and marine tourism. Monetization is carried out by connecting improvements in ecosystem services to monetary estimates.

Expert appraisals of impacts on ecosystem services are reported in a background study (Havs-och vattenmyndigheten 2015b). These appraisals concern a subset of measures which are expected to have an impact on either commercial fishing or marine tourism. In a second step, the percentage of gap closure is determined relative to GES (ibid.). This percentage is then used as a weighting factor. The CBA team provides an estimate of the expected increase in fishing activity between the business-as-usual scenario and GES, and how the improvement in ecosystem services affects the demand for marine tourism. There is, however, no discussion in the CBA report on whether the cause-and-effect relationship between improvement in marine ecosystem services and tourism on the one hand, and improvement in marine ecosystem services and commercial fishing on the other, are accurately modelled by the studies that provide inputs for monetization. In addition, there is some confusion about the welfare measures. The value added is applied to commercial fishing and producer surplus to marine tourism.

## Presentation of CBA Results

In the Finnish report, results are discounted to 2014 with a discount rate of 3% during the time period 2016–2021. The results are presented in aggregate terms and reveal that benefits exceed the costs, with reasonable certainty. In order to capture uncertainty, an interval of benefits and costs is presented. On an aggregate level, the CBA shows that benefits will exceed costs if the Finnish PoM is implemented, but there is no information concerning the benefits and costs of specific measures.

The Swedish report presents both measure-by-measure estimates of costs supplemented with qualitative assessments, and discounted benefit-and-cost estimates on the aggregate level. The measure-by-measure summaries use several dimensions, which makes it difficult to compare them. No attempt is being made by the CBA team to provide recommendations at the level of specific measures. On the aggregate level, benefits and costs are discounted during the time period 2016–2030 with a discount rate of 3.5%. Based on the interval of high and low estimates, it is shown that benefits exceed costs with reasonable certainty.

The Estonian CBA applies semi-quantitative expert assessments, making it possible to rank measures. Several information measures receive high scores: they are acceptable, have no indirect costs and their budgetary costs are low, for example, information about non-indigenous species and awareness-raising activities concerning marine litter. High-cost measures, with significant indirect costs and predicted difficulties in the course of implementation, typically receive the lowest scores, including measures to manage storm water discharge in coastal areas. Although the result seems reasonable, the aspect of gap closure relative to GES, remains vague. As an illustration of sensitivity, the CBA team show the relationship between points and costs in a diagram.

There is no aggregate valuation of the monetary benefits. In order to provide a benchmark, the Estonian report refers to recent contingent valuation studies of improved marine environment. The CBA refers to a study (Tuhkanen et al. 2016) that has estimated values for three descriptors using choice experiments: non-indigenous species (D2), water quality improvement (D5 and D8) and oil spills (D8).

## Comparison

A comparison of CBAs reveals that the monetization of benefits has been a great challenge. No monetization has been possible at the level of measures. Only descriptor and aggregate-level benefit transfers are presented. Estonia refers to a relevant contingent valuation study, but does not transfer benefits. Sweden and Finland have transferred benefits from recent contingent valuation studies. In these contingent valuation studies, environmental quality improvements have been significant and scenarios differ from those of the National Marine Strategy. The CBA teams have solved this difference by transferring the share of benefits that matches the expected percentage of the gap closure with respect to GES. This suggests an implicit assumption that benefits are linear with respect to quality improvement. In the Swedish report, benefits from nutrient reduction have been transferred to all dimensions of GES. It is not evident whether this assumption is valid. The authors propose that the improvements from attaining other dimensions of GES have similar qualities.

The business-as-usual scenario is explicitly reported in the Swedish CBA, while in the Estonian CBA, it remains vague. Being at an aggregate level, the business-as-usual scenario of the Finnish CBA lends itself to that of the CEA. The Swedish CBA is the only one that uses an ecosystem service approach for identifying benefits. The linkages between the ecosystem services approach and the connection to business growth of commercial fishing and the increase in marine tourism demand are, however, not transparent. While there is an absence of comparisons between measures in the Swedish and Finnish reports, the Estonian CBA illustrates the ranking of measures on the basis of points received during expert assessments. The ranking provided by the CBA differs to that of the CEA, suggesting that the wider perspective of CBA has added information.

## International Collaboration

During the first cycle of MSFD and the development of the (PoM), international collaboration among the studied countries has been rather modest. This concerns the work of identifying new measures, choosing the methodology for CEA and CBA. Since there already is an institutional body for cooperation to improve the environmental status of the marine environment among the Baltic Sea countries, collaboration on the issues of MSFD would have been expected via HELCOM and through the coordinated Baltic Sea Action Plan (BSAP).[4]

The issue areas and identified marine environmental problem areas of the BSAP fit rather well with MSFD's descriptors. In addition, HELCOM has several working groups that are dealing with selected issues and provide recommendations for participating countries within issues that match those of MSFD (eutrophication, hazardous substances in water and food, as well as accidental pollution at sea, protection of fish resources, biodiversity protection and marine protected areas). The most recent coordinated area is marine litter.[5] The existing platform for cooperation, and the overlap between issue areas between MSFD and HELCOM, suggest that collaboration could be helpful in many ways, including the selection of measures with beneficial cross-border impacts.

So far, HELCOM has not had the competence to assess the socio-economic impact of human activities on the marine environment or to estimate the monetary value of marine ecosystem services and the cost of their degradation. The need for this competence and coordination of methodologies has been recognized and the first steps are being taken to identify issue areas and methodologies for socio-economic assessment during the next cycle of MSFD. An example of this is the planned work of the HELCOM TAPAS project. The assessments this project will carry out are meant to be developed so that national governments can use the results in the 2018 reporting under MSFD. Among several sub-goals, there is also the aim to develop a framework for economic and social analyses in the Baltic Sea region that will contribute to harmonized reporting under MSFD Article 8. This article includes the reporting need for marine uses of the national marine areas. The aim is to extend the collaboration platform used by the project to include the requirements of the second cycle of MSFD, i.e. development of the programme of measures and the coordinated approaches and methodologies for CEA and CBA.

Based on the experience of the first cycle, a regional informal network of national experts on economic analysis has emerged, and the outlook for the next cycle looks more promising in terms of coordinated methodologies and comparable results. It is already clear that the differences among countries in terms of timing of preparatory processes, administrative capacities and financial resources as well as research capacities on economic and social analysis pose a challenge.

## Conclusions and Recommendations

By the logic of the process proposed by MSFD, the countries are required to suggest new measures in response to gaps between the expected status of the marine environment in 2020 and the target of GES. Suggested measures are, in many cases, expressed in terms of what the measure intends to achieve, for example, the restriction on fishing, clean beaches, use of liquefied natural gas (LNG) in shipping and reduction in the use of plastic bags. In line with the observation of Söderholm et al. (2015), the reviewed economic analysis of the national programmes of measures of Estonia, Finland and Sweden focus on measures rather than on implementation. Awareness raising, research and development and other means of information provision are frequent in the first national PoM, but as means of implementation, information alone most often has only a minor impact. Uncertainty about take-up complicates both the appraisal of the impact on the environmental target and the estimation of costs and benefits.

In the work process, gap analysis relative to environmental targets is carried out early. For obvious reasons, gap analysis on the environmental status and targets should be done by natural scientists who are experts on marine ecosystems. It is not clear, though, how the proposals for new measures have been put together. In some cases, these proposals appear to have been suggested by experts on marine ecosystems while in others, they seem to have been put forward by public officials. It seems that measures have been identified mainly based on technical feasibility or social and political acceptability. Experts on economic analysis have been contracted at a later stage, when there is limited or no opportunity to influence the design of measures or to suggest policy instruments for implementation. In addition, as it is the first cycle of MSFD, there is a lack of earlier studies to rely on and this, along with the limited time frame for the PoM process, has affected the depth of the analyses.

The review suggests that there is only a weak link between those who have been involved in the designing of measures and those who have expert knowledge about implementation. The reason for this is obvious in the case of all three countries. The superior capacity concerning background studies and research funding in Sweden and Finland has not made a difference. All three countries suggest measures with vague implications on implementation. In order to prepare for the next cycle, it is important to build up knowledge about policy instruments and implementation. There is a need for reviews of existing ex post studies and further studies that evaluate existing policy instruments to protect the marine environment.

All three countries have chosen expert assessments as the desired mode for carrying out CEA. As a result, the assessments of the impact on the marine GES targets have been qualitative. Due to current gaps in scientific knowledge and quantitative models of sea ecosystems, there are no good alternatives to expert assessments and qualitative appraisal. While Sweden and Estonia have applied standard methods to appraise the effect of a measure on gap closure, Finland has adopted an innovative probabilistic approach. Using this approach has made it possible for the Finnish CEA team to make use of the uncertainties of appraisals within the scientific community. This methodology needs to be further developed in order to allow for a richer set of indicators as the base of appraisal.

Another important aspect of the CEA and CBA is the definition of the business-as-usual scenario. All countries present a business-as-usual scenario for 2020. It is uncertain, though, whether the business-as-usual scenario reflects only current policies or also includes policies that have been adopted but not yet implemented. The addition of policies that have not yet been implemented further increases the requirement of information and calls for extended gap analysis on existing policies (Water Framework Directive, Habitats Directive, etc.) and the implementation of their measures, which should be carried out by officers responsible for regulating and enforcing relevant issue areas.

Cost estimates can be put together using different methods, including ex post studies, collection of information using a bottom-up approach and expert assessments. In the reviewed studies, bottom-up and expert assessments have been used to estimate costs. The precision of cost estimates from expert assessments depends on the knowledge of the participants. Ideally, estimation of costs based on expert assessments should be the task of other experts than those who appraise the impact on the marine ecosystem. The accuracy of the bottom-up methods that have been applied is judged better than expert assessments. For future purposes, it is important to build knowledge about costs, preferably via ex post studies.

The most challenging task of CBA has been the monetization of benefits. No attempt has been made to assign monetary values to the benefits of individual measures. When monetization has been possible, benefit transfer has been used for assigning monetary value at the level of descriptors. Two approaches have been adopted: benefit transfer from recent contingent valuation studies, and appraisal of business implications for commercial fishing and marine tourism based on the improvement of ecosystem services. For the purpose of benefit transfer, CBA teams had to adjust contingent valuation scenarios to the scenarios of national marine strategies. Assumptions have been made about linearity in benefits. This might be a reasonable approximation. However, further studies are required to assess the validity of this assumption. Ecosystem service analysis provides an important link between the improvement of marine ecosystems and welfare measures. The cause-and-effect relations concerning individual welfare implications and business opportunities deserve further research.

Another challenge pertaining to CBA concerns the estimation of indirect costs. This is related both to the lack of ex post studies and the fact that the appraisal concerns measures rather than policy instruments. Finding cost estimates when implementation is unclear implies that less is known about indirect costs and, for this reason, indirect costs might be overlooked. It is, therefore, highly probable that the cost estimates of the CBAs suffer from downward bias.

The reviewed CBAs have presented sensitivity analyses. At the level of recommendations, only the qualitative CBA of Estonia compares the scores and provides a ranking at the level of individual measures. At the same time, it is not possible to conclude whether the benefits of the measures—either separately or at the aggregate level—exceed their costs. This is due to the lack of monetization of benefits. The Finnish and the Swedish CBA provide net present values at the aggregate level. Valuation that can enable the monetization of disaggregate benefits is another area that deserves further research.

Regional coordination of economic analyses has been rather modest during the first cycle of MSFD for different reasons. For the second cycle, HELCOM has initiated activities to coordinate the approaches and methodologies of economic and social assessments of MSFD. In order to achieve GES in the whole regional sea area, it is important to consider cross-country coordination of measures since measures taken by individual countries are not sufficient to achieve GES in their national marine area. Moreover, in the face of limited public resources at the national level to conduct the required valuation studies, coordination opens up opportunities for collaborations at the regional sea level and for valuation studies across neighbouring countries.

## Notes

1. The Marine Strategy Framework Directive calls for additional measures to those relevant to other directives and EU policies and concern the quality of marine waters. The Directive states that: "In so far as particular aspects of the environmental status of the marine environment are not already addressed through Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy (2) (WFD) or other Community legislation, so as to ensure complementarity while avoiding unnecessary overlaps".
2. Annex vi of MSFD presents eight types: input and output controls, spatial and temporal distribution controls, management coordination, traceability measures, economic incentives, mitigation and remediation tools, and communication, stakeholder involvement and public awareness.
3. Havs-och Vattenmyndigheten in Swedish.
4. Baltic Sea Action Plan (BSAP): http://helcom.fi/baltic-sea-action-plan.
5. HELCOM approved the Baltic Sea Marine Litter Action Plan in 2015.

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# 6

# Factors Affecting Choice of Travel Mode in Tallinn

Helen Poltimäe and Mari Jüssi

## Introduction

Together with the economic development and growing welfare, the use of cars is also increasing. People prefer to drive cars for various reasons, mostly related to personal interest. At the same time, there are externalities originating from car use that impose costs on society as a whole. At the local level, the most significant externalities are related to air pollution, congestion and traffic accidents, and these impacts are not taken into account by car drivers. Car use is also contributing to the global problems like climate change and ecosystem damage caused by road infrastructure.

Over the past few decades Estonia has experienced rapid changes in its economy as well as in transport preferences: the number of cars

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in Estonia has increased more than threefold since 1990. Back then the number of cars was 153 per thousand people, while in 2015 it was already 515 (Statistics Estonia 2016). In a relatively short time period, Estonia has become one of the most motorized countries in the European Union. In 2012, the highest motorization rates in the EU were in Italy (621 cars per 1000 inhabitants), Malta (592 cars per 1000 inhabitants) and Germany (530 cars per 1000 inhabitants) (Eurostat 2016). Part of the explanation for the high number of cars in Estonia lies in the low density of settlements, but there is no doubt that lifestyles and transport habits of people have also changed. The mileage of cars has almost doubled in less than 15 years: from 4631 million km in 2000 to 7916 million km in 2014 (Environment Agency 2016). At the national level, there is a clear indication that car ownership and use have grown together with income level (Jüssi et al. 2010). However, at the individual level, the relationship is not so clear: instead of income, the relative decrease of car prices and easy access to loans seem to matter more (Poltimäe 2014).

Energy consumption due to road transport in Estonia is very high: in 2014 it formed 91% of the entire transport sector's energy consumption (Eurostat 2016). The proportional share of road transport energy usage in the total transport sector energy consumption has been quite stable, but the absolute energy consumption of the transport sector in Estonia has increased considerably: while it consumed 24,527 TJ in 2000, in 2014 this figure had grown to 32,673 TJ (Eurostat 2016). In addition, land transport is associated with quite remarkable external costs. In 2007, the total external cost of Estonian road transport was estimated to be approximately 442 million Euros, while the amount collected by taxes on transport was 278 million Euros, of which 75% was spent on national road construction and maintenance (Anspal and Poltimäe 2009).

Estonia has applied a liberal tax policy regarding private transportation. The only tax applied on transport is fuel excise, but it has been shown that this tax instrument has not been effective in altering the upward trend of car use in Estonia (Poltimäe 2014). Although the utilization of fiscal instruments for car ownership and use has been limited in Estonia, there are other measures applied to move people out of cars. The city of Tallinn, home to about one-third of the Estonian population, introduced free public transport in 2013. From 2012 to 2015, the entire public transport fleet in Tallinn was renewed; new bus priority lanes were introduced and parking fees were increased in the Tallinn city centre. However, the increase in demand for public transport has been very modest—according to estimates, only 1.2% could be explained by free public transport (Cats et al. 2014). Some estimates of the impact of free public transport have been higher, but as demonstrated by Cats et al. (2016), walking has been replaced by public transport and also, the average travel distance has increased.

This study is aimed at finding the key factors related to travel mode choice by Tallinn citizens. More specifically, it focusses on factors affecting daily car and public transport use, as the data about daily cyclers is scarce and walking is usually combined with some other mode of transport. Attaining deeper knowledge about these factors could help in understanding whether and how the undesirable trend of increasing car dependence could be reversed and who should be the target group of relevant policy measures. There have been previous studies about socio-demographic attributes that affect the travel mode choice (for example, Buehler 2011; de Palma and Rochat 2000; Schwanen et al. 2002). These studies have been carried out in countries like Germany, the USA and Switzerland. As discussed above, the Estonian context is different. Moreover, it is mostly socio-demographic data of studied households or individuals that have been taken into account. In this study, we are able to complement socio-demographic background data with information about the motivations driving the choice of transport mode and possible alternatives to these choices. Another interesting piece of information is the compensation of car costs and parking costs by employers. As car compensation is quite extensively used in Estonia, this might have a bearing on the decision of driving a car vis-a-vis opting for other travel modes. Similar information is not available in other studies—perhaps the role of compensation is not as substantial in other countries, or not enough data is available.

## Literature Review

In order to identify the factors affecting transport mode choice, it is useful to first discuss the different motivations for travel. Motivation to travel can be of intrinsic or extrinsic nature. Traditionally, travel has been regarded as extrinsic, i.e. people commute to achieve objectives like shopping, going to work, meeting friends, etc. (Mokhtarian et al. 2015). These can be termed instrumental or utilitarian motives and can be operationalized, for example, by financial costs, travel time, convenience and physical effort (Gardner and Abraham 2007). Recently, more attention has been given to intrinsic motivation, i.e. people move because of enjoyment of the activity itself—for example, driving a car might be related to independence, control, status and safety (Gardner and Abraham 2007; Mokhtarian et al. 2015). The concepts of intrinsic and extrinsic motivation can not only explain why and where people travel, but also the mode of transport they choose.

If it were only extrinsic motivation, then travel demand could hardly be managed. However, there is no doubt that, at least to some degree, it could be affected by policies targeted at costs and convenience of travel. Gardner and Abraham (2007) bring out five motives in sustaining car use that are to a large extent related to utilitarian considerations: journey time, physical and psychological effort related to transport, personal space and financial expenditure. However, utilitarian considerations are closely related to affective concerns, which pertain to emotions experienced during travel. Steg (2005) classifies the motives for car use as instrumental, affective and symbolic. Symbolic motives are related to the identity of a person and allowing self-expression. The study by Steg (2005) finds that for males and members of young age groups, symbolic motives are more important than for females and older age groups.

While the motives for travel form part of a very recent branch of literature, there are other factors that have deserved attention for decades. One of the main factors associated with increasing car ownership and use is income. Dargay and Gately (1999) model the S-shaped relationship between income and car ownership based on data from 26 countries, i.e., car ownership responds moderately to income increase at the lowest and highest income levels, but quite abruptly in the middle-income group. Metz (2010) finds that in the Great Britain the average number of yearly trips has been stable for more than 30 years, but the distance travelled has increased. Eriksson et al. (2008) demonstrate the increasing use of cars in work-related commuting in Sweden and see no sign of saturation. Kuhnimhof et al. (2012) state that in industrialized countries, travel demand is stagnating, but the trend differs across socio-economic groups: in Germany, car use by young male adults is decreasing and without it, the general trend of car use might still be increasing. Hence, there are some fairly controversial results emerging out of literature regarding the relation between income and car use.

There are also other socio-demographic characteristics at the individual or household level that have been found to have a bearing on the choice of mode for daily travel, for example gender, education, household type, settlement type, etc. Women are found to use cars less frequently than men (Giuliano and Dargay 2006; Schwanen et al. 2002). As for age, different relations have been reported: Kuhnimhof et al. (2012) show that car use is increasing slightly among older age groups. On the contrary, a negative relationship between age and car use has been reported by de Palma and Rochet (de Palma and Rochat 2000) and Giuliano and Dargay (2006).

The concentration of car ownership has been found to be an important factor affecting transport mode choice (Buehler 2011; Carse et al. 2013; de Palma and Rochat 2000; Giuliano and Dargay 2006; Schwanen et al. 2002). Others have discussed the role of high socio-economic status in giving rise to lifestyles that involve more travel, and the association between travel and income (Metz 2010; Schwanen et al. 2002). Closely related to this indicator is employment status: employed people tend to use cars more (Buehler 2011; Giuliano and Dargay 2006). All three measures, socio-economic status, income and car ownership, are perhaps closely related.

Body mass index was found to be a relevant factor while explaining the preference for cars over bicycles in a study conducted by Carse et al. (2013). The authors of this chapter consider this relationship as an opposite one: choosing a bicycle means regular activity and hence the body mass index of these people can be expected to be lower compared to those who opt for cars.

Another set of factors affecting transport mode choice is related to physical environment or city structure. It matters which part of a city a household is located in, how densely it is populated and what is the general urban structure of the region. A negative relationship between population density and car use has been demonstrated by Buehler (2011) and Giuliano and Dargay (2006). Public transport and journeys on foot are more frequent in the denser city centre and car use is dominant in suburban areas (Monzon et al. 2011). Similarly, the greater the mix of residences and workplaces, the lower the probability of car use (Buehler 2011). Carse et al. (2013) have claimed the obvious relation that for rural households, the preference for cars is much greater than for urban households.

If there are limitations on workplace parking, it is less likely that people will use cars, but if there are no limitations, car use is more likely (Carse et al. 2013). Also, Schwanen et al. (2002) have shown that in the case of congestion and parking problems, people are not very prone to using cars, because the travel time is greater.

An alternative measure found to be relevant is the difference in journey time between cars and public transport: a larger time difference leads to lower usage of public transport (Monzon et al. 2011). Eriksson et al. (2008) have analysed the main reasons or motivators for reducing car use. They asked people what would make them use their car less for commuting to work. The most common reasons were "working from home on some days" and "improved public transport" (Eriksson et al. 2008: 429). Similarly, Kingham et al. (2001) claim that people would reduce the use of cars if public transport was frequent, reliable, convenient and cheap.

It is not only physical factors that matter when it comes to transport mode choice, but people's perceptions: for example, how safe they perceive driving a car, walking, cycling, etc. (Iftekhar and Tapsuwan 2010).

As has been discussed, the role of various measures in increasing the cost of car use or encouraging the use of public transport affect the decision to opt for cars (Carse et al. 2013; Kuhnimhof et al. 2012; Schwanen et al. 2002). Other researchers have shown that car use is quite inelastic to cost (de Palma and Rochat 2000; Kingham et al. 2001). While this inelasticity seems to be greater in the case of fuel and car price, the costs and inconvenience related to parking and congestion have proved to be relevant factors when it comes to discouraging daily car use.

Hence, the two broad groups of factors affecting transport mode choice can be classified as socio-demographic factors and physical/transport system factors. Climatic factors have also been found to matter in travel decisions, but the present study does not take those into consideration. In addition, the cultural context of a country may be intertwined with the application of transport policy measures. For example, Buehler (2011) has demonstrated that controlling for all explanatory variables for car use (namely socio-economic, demographic, spatial and land-use), Germans are still much more likely to walk and use bikes and public transport than the people of USA. These factors may be related to the extrinsic and intrinsic motivation factors discussed above.

## Data and Methodology

To study the different factors that affect travel mode, we used data from a household travel survey conducted by TNS Emor in Tallinn in 2015 (commissioned by Kredex) and a survey carried out in 2012 by Eesti Uuringukeskus (commissioned by the Tallinn City Government). The data not includes information about households but also a very detailed registry of trips of all the observed household members. More than 2000 households were observed, resulting in a unique database of persons' mobility patterns, including time, costs, destinations, purpose of trips and reasons for mode choice as well as willingness to change mode of travel.

We constructed two logistic models with a binary dependent variable. The first model pertains to car use and the dependent variable takes value "one" if a respondent claims to be a daily car user and "zero" if a person uses a car less frequently. The second model characterizes public transport users and the dependent variable takes value "one" if respondent is a daily public transport user. An alternative approach would be to use a multinomial logit model, where the dependent variable reflects several transport modes, including, for example, cycling, walking, etc. As in the case of Tallinn's survey data, where the proportion of daily cyclists is quite small, this mode choice cannot be taken into account in our model. On the contrary, the proportion of surveyed people claiming to be daily walkers is very high. This may be due to several reasons: first, walking is usually combined with some other travel mode, for example, public transport, and hence it is not possible to decide the respondent's main travel mode based on this data. Secondly, walking could be an objective in itself, as in physical exercise and not as a means of getting somewhere. This question is beyond the scope of this research and hence, walking as a travel mode is not considered in our model.

The independent variables included in the models are selected according to the literature review presented above, after adding some variables that are possible due to survey data. The socio-demographic indicators included in the model are gender, income, age, number of children, education, social status, driving license holding and car ownership level, which have been found relevant by different authors. An additional variable considered in our model is compensation of car costs by employers. Given that there have been no limits on car costs compensation in Estonia, such a scheme is often used as part of the employee motivation package by companies and could affect the travel mode decision. The survey also includes information about the possibility of working remotely, which could also affect the general travel pattern of a person: if this is possible from time to time, it might reduce the need for daily car use. Also the number of total trips made by a person per day was calculated and included in the model as it can be suspected that people who make more trips per day, be it for work or to take other family members, have a higher chance of being daily car users.

The variables related to city structure or physical environment are density of population in a city district where the respondent lives and the relevance of parking costs. Parking costs are estimated using a survey question regarding whether the respondent had to pay parking costs during each trip; if so, how much; and whether these were compensated by their employer.

In addition, the model for daily car users includes a variable to determine each respondent's main motivation—extrinsic or intrinsic. This is constructed based on the question, "What is the main reason you use a car as your main transport mode". The main reason is usually given spontaneously by the respondent, but has been classified by the survey administrators into the following categories:

(1) Reasons related to destination (distance, accessibility);
(2) Not satisfied with public transport;
(3) Time-saving;
(4) Independence, I can decide myself;
(5) Environmentally friendly;
(6) Habit;
(7) Comfort;
(8) Affordability;
(9) Work assignments;
(10) Necessity to transport people;
(11) Necessity to transport things; and
(12) Travel in other ways is complicated (disability).

Of these named categories, independence and comfort could be classified as intrinsic motivations for driving a car, and hence a binary variable of intrinsic motivation is formed.

The second model for public transport users is constructed very similarly, excluding only variable—motivation for car driving.

## Results

Before analysing the factors that determine whether a person chooses to travel by car or some other mode, we analyse whether the travel behaviour of people has changed and to what degree. The time period for this analysis is, unfortunately, not very long: the data about working people's commuting modes in Tallinn is available for 2000–2015. The share of private car use has increased, from 35% in 2000 to 44% in 2014 (Fig. 6.1). The increase is particularly steep in the period 2003–2008.

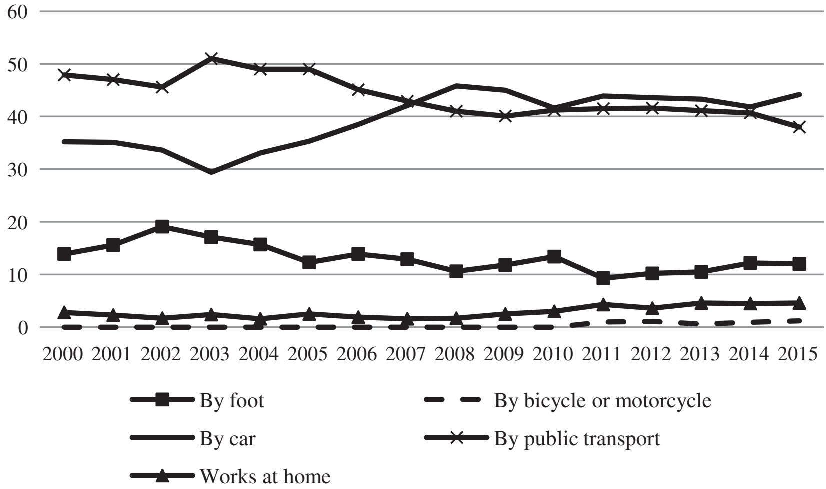


**Fig. 6.1** Main modes of commuting to work, percentage of occupied in Tallinn, 2000–2015. *Source* Statistics Estonia (2016)

The share of public transport users has decreased notably from 4 to 38%. The percentage of people who walk to work has decreased a little, from 14 to 12%. Hence, a significant change in travel behaviour has taken place in last 15 years, and we suspect that if data for a longer time period had been available, the change would perhaps have been even more drastic.

According to travel survey data, the number of households with no car has dropped from 41% in 2012 to 37% in 2015. The average number of cars per household in 2015 is 0.8; if one calculates the average number of cars only for households that have a car, it is 1.3. 17% of households own two or more cars.

The number of trips made by Tallinn inhabitants is still lower than in countries like USA and Great Britain: according to Giuliano and Dargay (2006), 4.4 trips per day are made by people in USA and 3 trips in Great Britain. The average time spent in travelling is about an hour in both countries. In Tallinn, these numbers are smaller: an average of 2.6 trips was made per day done by respondents of the survey in 2015. However, this number has increased considerably even in the last three years: in 2012, the average number of trips made by surveyed people in Tallinn was 1.98. The average time spent in travelling is about half an hour for surveyed households in 2015. The number of trips is the highest among the self-employed (3.1 trips per day) and the lowest among homekeeping people (1.8). There is also a linkage with family type: in the case of children, the number of trips is 3.1 per day, while if there are no children, the average number of trips is 2.5 per day.

In 2015, about 40% of the sample claimed to be daily car users. In the 2012 survey, the question was posed in a slightly different form: respondents were asked to specify their transport mode choice for going to work, school or other destinations. About 32% of the respondents said that they travel mostly by car. Although the question was put slightly differently, we can still affirm that there is quite a significant increase in the number of car users and also the number of trips made by Tallinn inhabitants.

In 2015, 36% of the respondents in Tallinn claimed to be daily public transport users. This number is higher than in the case of Estonia in general, as the public transport system in Tallinn is better due to the large number of inhabitants in the city and high population density in most city districts.

The number of daily car users differs greatly according to social status. The maximum car users can be found among self-employed people: about 70%. Among those with full-time employment, 47% are daily car users. Among the retired, students and homekeepers, the proportion of daily car users is the smallest (9, 20 and 25%, respectively). However, in these last named categories, the probability of being a daily car user increases abruptly in the case of employment: among working retired people the proportion of daily car users is 22.6%; among working students it is 30.8%.

To better understand people's travel motivations, the stated reasons for daily car use were also analysed. The motivations were classified as intrinsic if the respondent claimed to use cars mainly for independence and comfort. All other responses were classified as extrinsic as these are not associated with enjoyment but with necessity—for example, reasons related to destination, time-saving, etc. In total, 20% of the sample cited intrinsic motivation for daily car use. There are slight differences across gender: 20% of men and 15% of women drive for intrinsic motives. A positive relation can be observed with income level: only 12% of people with income up to 400 Euros cite intrinsic motivations while among higher income levels, this proportion is more than 18%.

The most frequent motivation was "reasons related to destination (distance, accessibility)", which was cited by 35% of the respondents as the main reason for daily car use. Other popular reasons were "comfort" (24%) and "time-saving" (17%). Only 3% of daily car users said that they drive cars because they are not satisfied with public transport. The stated reasons among daily public transport users follow a similar distribution: 41% of daily public transport users offer "reasons related to destination" as the main reason; 25% state the fact that it is free of charge as the main reason; and 13% cite "comfort" as the main reason. Only 8% of daily public transport users say that the main reason is related to not having a car and/or driving license.

The results of the logit models that predict daily car use and public transport use are presented in Table 6.1, in the form of odds ratio. Both the models are statistically significant. The first one, which looks at daily car use, predicts 80% of cases correctly, while the second model, which looks at daily public transport use, predicts 76% of cases correctly.

According to the results of these models, females are less likely to use cars every day compared to males. The number of children increases the odds of being a daily car user. The number of cars owned and income level also increase the likelihood of daily car use. As for labour market status, in the case of working people (either employees or entrepreneurs), the odds of being a daily car user are three to four times higher than in the case of homekeepers, the retired or students. Age and education of respondents did not have a significant relation with daily car use.

The possibility of working from a distance is correlated with income level: the proportion of people who can work remotely is increasing significantly with income level. About 23% of people earning less than 400 Euros per month claim that they can work remotely on some days, but among these who earn more than 700 Euros per month, this percentage is already 49%. As income effect is already reflected in the model, we decided to exclude the variable of the possibility of remote working.

**Table 6.1** Results of logit models in the form of odds ratio

| | Model 1 (daily car user) | | Model 2 (daily public transport user) | |
|---|---|---|---|---|
| | Odds ratio | *p*-value | Odds ratio | *p*-value |
| Gender (reference group: males) | 0.62 | 0.000 | 1.78 | 0.000 |
| Number of children | 1.18 | 0.037 | | |
| Labour status (reference group: employee) | | | | |
| *Entrepreneur* | | | 0.51 | 0.003 |
| *Homekeeping* | 0.34 | 0.000 | 0.39 | 0.001 |
| *Retired* | 0.38 | 0.000 | 0.20 | 0.000 |
| *Student* | 0.33 | 0.011 | 3.07 | 0.002 |
| Income per household member (reference group: up to 400 Euros) | | | | |
| *400–700 Euros* | 1.59 | 0.021 | | |
| *More than 700 Euros* | 1.99 | 0.001 | 0.55 | 0.001 |
| Holding of driving license | 5.81 | 0.000 | 0.32 | 0.000 |
| Car ownership level | 2.63 | 0.000 | 0.53 | 0.000 |
| Car costs compensation (reference group: does not get compensation) | 1.74 | 0.001 | 0.47 | 0.000 |
| Number of trips per day | 1.08 | 0.059 | | |
| Intrinsic motivation (independence and comfort) | 2.23 | 0.000 | | |
| | LR chi$^2$ = 869.6<br>Prob = 0.000<br>PseudoR$^2$ = 0.351<br>Correctly classified: 79.6% | | LR chi$^2$ = 420.44<br>Prob = 0.000<br>PseudoR$^2$ = 0.185<br>Correctly classified: 75.9% | |

The number of trips made per day increases the odds that a person is a daily car driver. In this case, it is difficult to assess the direction of the relation: it is possible that the decision to drive a car allows for more trips.

The physical environment variables do not show any significant relation to daily car use. This means that neither the population density of a city district where a respondent lives nor that the city district itself can help explain the preference for car use. However, the same is not the case with the income level of different city districts: in districts where income levels are higher (districts with private houses), the proportion of people using cars daily is also higher. A more relevant factor than population density would be the distance to the closest public transportation stop and the frequency of public transport, but unfortunately, this information is not available.

Parking costs also do not show any significant relation to the decision to be a daily car user. One of the explanations for this is that parking compensations might be included in car costs compensation. Some parking costs might be missing from dataset as these whose destinations are related to high parking costs or limited parking availability, are perhaps not daily car users and we cannot observe their parking costs.

As for car compensation, it has clear positive relation to daily car use. Among daily car users, there are 31% of respondents who get car compensation from employer. Car compensation seems to be related to income level also: of these who get compensation, 9% earn less than 400 Euros per month, 17% earn between 400 and 700 Euros and 74% earn more than 700 Euros per month.

Intrinsic motivation for car use is a relevant factor in our model. People who mention "comfort" and "independence" as the main reason for car use are very likely daily car drivers. Comparing the profiles of these people with those who drive for extrinsic reasons, there appears no difference across gender, education, number of children and social status: the distribution across these is similar for both groups. The differences are largely based on age and income level. About 65% of car users driving for intrinsic motives are aged between 15 and 44 years, while among these who drive for other reasons, this percentage is 48%. Most of this difference can be seen in the age group 25–34. Among those who drive for intrinsic reasons, only 19% of the respondents earn less than 400 Euros per month, while this proportion is 28% for those who drive for other reasons. As expected, there are more people belonging to higher income groups among those who drive for intrinsic reasons compared to the rest. To see the readiness of such people to move out of cars, we looked at their willingness to change their main travel mode. The following question was employed: "If you could choose, which travel mode would you prefer to get to your main destinations?" Most drivers who had cited intrinsic motivations stated that their preferred mode would still be a car (59%), while 16% said they would use public transport and 14% would prefer to walk. Only 5% chose biking.

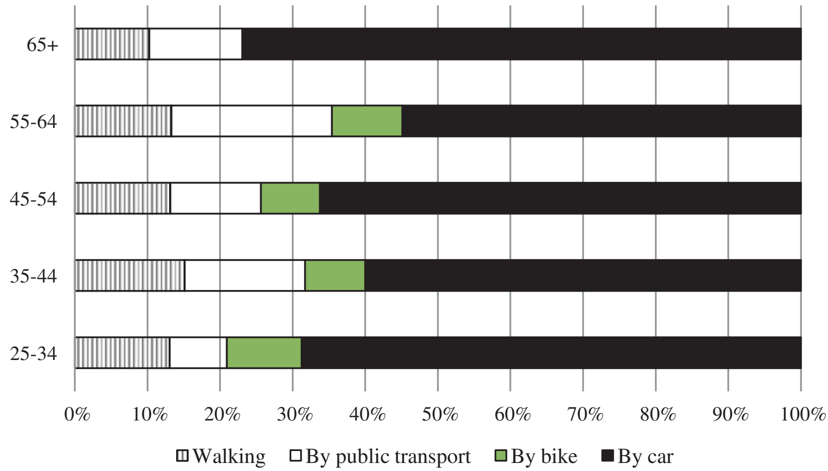


**Fig. 6.2** Willingness to change main travel mode of daily car users

The willingness among daily car users to change their main travel mode differs by age group (see Fig. 6.2). In every age group, there is quite a remarkable proportion of people who would still prefer to drive a car: these ranges from 55% in the age group 55–64 and up to 77% in the oldest age group. The willingness to switch from driving a car to biking is quite similar across age groups, in the range of 8–10%. The readiness to use public transport is highest in the age group 55–64 (22%) and lowest in the youngest age group (about 8%).

When turning to factors that affect the likelihood of being a daily public transport user, most of the linkages work in the reverse direction to those discussed in the case of daily car use. Females are more likely to be daily public transport users. Being a student increases the odds of being a daily public transport user by more than three times compared to being an employee, but in the case of employees, the odds of being public transport users are still higher than in the case of entrepreneurs, homekeeping and retired people. At the highest income level (more than 700 Euros per household member), the odds of being a daily public transport user are about two times lower compared to the lowest income group (less than 400 Euros per household member). As expected, holding a driving license, owning a car and getting car costs compensation from employer decrease the likelihood of being a daily public transport user. The role of education, number of children and number of trips made are not significant in this model.

## Conclusions

This chapter identifies the key factors that are relevant to travel mode choice in a city that has made available different instruments for sustainable transport. For example, free public transport for residents, introduction of bus priority lanes, paid parking areas at the city centre, etc. Still, car use continues to increase and streets are becoming more congested. As expected, travel mode choice is affected by different factors. Income has a positive effect on the decision to be a daily car driver, which is in line with previous studies. However, we also found that the role of car compensation by employers is significant. This instrument used by companies and enabled by the Estonian tax system works in the opposite direction to sustainable transport policy and if one wants to reverse the trend of increasing car use, more stringent policies need to be applied to such compensation schemes. There is not much motivation for giving up car use if one's employer is compensating these costs, and the more a person earns, the more likely it is that some or all of their car costs will be compensated.

Daily car use is very much related to social status: if a person works, it is more likely that they also drive a car. It is not too clear whether this is due to possibility (because they can afford it) or desire (because they enjoy driving). According to the reasons cited for daily car use, the most common is related to destination (distance and accessibility). We can assume that these people could be weaned off car use by attracting them with a good public transport system or cycling network that can satisfy their needs.

The role of intrinsic motivation in car use has proven to be significant one, but not a very large share of people cited them as their primary motivation. As expected, those who did consist of younger and middle-aged people in higher income categories and perhaps this group is difficult to wean off cars.

As for public transport, it is not only because it is free that people use it, but a surprising number claim to do so because it is comfortable. This segment should be developed further by improving the quality and accessibility of public transport. In doing so, urban planning plays a big role: if cities expand without integrating public transport and mobility planning—as has been happening in certain areas of Tallinn—the travel mode choices of inhabitants in remote areas become limited as public transport is not always available and the travel time in the case of public transport increases.

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# 7

# Environmental Impacts of Rural Landscape Change During the Post-Communist Period in the Baltic Sea Region

Kari Lehtilä and Patrik Dinnetz

## Introduction

Change in landscape is one of the main anthropocentric processes affecting terrestrial ecosystems (Vitousek et al. 1997; Lindenmayer and Fischer 2013; Mitchell et al. 2015). It has a major impact on patterns of biodiversity and biogeochemical cycles, i.e. flow of nutrients, carbon and water through ecosystems. Destruction of habitats due to landscape change has been identified as one of the most important factors behind species extinction (Pimm et al. 2014). On the other hand, landscape change can also involve development towards conditions that are regarded as environmentally favourable, such as raised water tables or new forests. These changes can either occur through passive processes, where the land is left for natural development, or through active management.

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The most extensive landscape change process in Europe in recent decades has been due to agricultural land abandonment after the break-up of the Soviet Union (Henebry 2009). The break-up of the communistic system fundamentally changed the conditions for agricultural economy through changes in land ownership, production methods, markets and state support. The extent of land abandonment in some regions was close to 60% of the agricultural area. In a longer time perspective, abandonment of agricultural land has been going on in East European countries and elsewhere in Europe throughout the twentieth century (Kuemmerle et al. 2015; Keenleyside and Tucker 2010). Up until the late nineteenth century, before urbanization and modernization of agriculture reduced the demand for agricultural land, agricultural area continued to grow due to population increase (Kana et al. 2008). When the production of animal fodder moved from hay meadows to agricultural fields, the total area of meadows decreased dramatically, with negative consequences for biodiversity because hay meadows are very species rich. After 1990, the new trend was the closure of collective and state-owned farms. By this time, most hay meadows were already gone. Instead, it was agricultural fields that were abandoned and either left as fallow or transformed, intentionally or unintentionally, into a forest succession. Post-Soviet land abandonment took place most extensively in the early 1990s. After 2000, the rates of abandonment have declined, and in some areas, abandoned land has been recultivated (Estel et al. 2015).

The process of agricultural land abandonment demonstrates how large-scale landscape change can occur not only as a transformation of natural habitats to human use, but also in the opposite direction, where land is transformed from intensive management to a more natural state with less anthropogenic features. It is evident that agricultural land abandonment in East Europe took place to such a large extent that it had substantial environmental effects. We will review studies of land abandonment on biodiversity, nutrient leakage to aquatic ecosystems, carbon sequestration and ground water supply. As our review shows, there is a mixture of environmental effects following land abandonment, some of which can be considered as having a positive and others a negative impact on the environment. Classifying environmental changes as positive or negative not only depends on the outcome for species and ecosystems, but also on our perspectives on rural environments and rural development. Based on a review of ecological patterns and processes, we discuss the socio-economic implications of land abandonment to present alternative views on how to interpret research findings on ecological change and to use them as basis for landscape management.

## The Extent of Agricultural Land Abandonment After the Break-up of the Soviet Union

Abandonment rates have been studied using many different methods and the estimates depend on the time period covered, so comparisons between studies should be made with care. Nevertheless, it is clear that large areas of agricultural land have been abandoned in East Europe after the break-up of the Soviet Union (Alcantara et al. 2013; Prishchepov et al. 2013). The trend has been especially strong in the Baltic states, Russia, Ukraine and Moldova (Alcantara et al. 2013). Schierhorn et al. (2013) report that cropland abandonment rate was 39% on the European side of Russia between 1990 and 2009. The Baltic countries have undergone the same process, Estonia being one of the hot spots of agricultural land abandonment. According to the Food and Agriculture Organization (FAO) of the United Nations, 54% of the arable land of Estonia was abandoned between 1992 and 2005 (FAO 2012). Although agricultural land abandonment also existed before that period, it was most widespread in Estonia during the early years of the 1990s, after the Soviet collapse (Mander 1994; Kana et al. 2008). In the late 1990s and onwards, abandonment and recultivation have approximately balanced each other out in Estonia (FAO 2012; Estel et al. 2015; Statistics Estonia 2016).

This pattern repeats itself, and at present, abandonment rates are slowing down and recultivation has started in many areas of Eastern Europe. In Western Ukraine, abandonment rates were up to 56% between 1986 and 2008 (Baumann et al. 2011). Similar to Estonia, abandonment was at its maximum in Ukraine in the early 1990s, and during the following decade, recultivation rates of unused agricultural land were more than 50% in some districts (Smaliychuk et al. 2016). Agricultural abandonment in the Carpathian ecoregion was also most extensive in the 1990s and occurred mainly in marginal areas, whereas recultivation of abandoned farmland has been common in agricultural areas with high productivity after 2000 (Griffiths et al. 2013). In the period between 2001 and 2012, farmland abandonment was strongest in north-eastern Poland and in Southern Finland (Estel et al. 2015). In Poland, it corresponded with changes in the economy of the agricultural sector. Finland has invested in long fallows and in supporting the introduction of grassland vegetation in agricultural fields.

Western Europe has also experienced extensive agricultural land abandonment (Strijker 2005). However, there are important differences between the land abandonment processes in West and East Europe. In West Europe, land abandonment has been taking place since the middle of the nineteenth century, and the main causal factors are urbanization and modernization of agriculture. Eastern Europe underwent a rapid political, economic and social transformation after the collapse of communism, resulting in very high rates of land abandonment over large areas during a short period of time (Baumann et al. 2011). Land abandonment in Western Europe has been high in areas distant from population centres and marginal in agricultural production (MacDonald et al. 2000). The same pattern can be found in Russia (Prishchepov et al. 2013), but in Ukraine the abandonment rates have been higher in more central regions (Baumann et al. 2011). Land abandonment in the former Soviet Union is not only the result of the structural rationalization of the agricultural sector, but in countries like Estonia, initially, an effect of extensive land reforms and neoliberal agricultural politics (Macours and Swinnen 2000). In later years, interaction with EU legislation has further affected the use of agricultural land, especially in Estonia but also in many other European countries (Holt-Jensen and Raagmaa 2010; Renwick et al. 2013).

## Ecological Succession After Agricultural Land Abandonment

Abandonment of agricultural land implies a change in management regime that unavoidably affects habitats and ecosystems. When land is abandoned and the earlier management regimes are discontinued, the most likely scenario involves encroachment of open land by bushes and trees. In Northern Europe, land abandonment naturally results in afforestation (Cramer et al. 2008; Verburg and Overmars 2009), but the course of succession and the resulting forest type depends on local environmental conditions and local and regional species pools. Afforestation following land abandonment can either rely on natural recruitment of tree species from the surrounding landscape, or take the form of active plantation with the aim of commercial forest production, or for the purpose of environmental management (Bremer and Farley 2010). Afforestation can also be a consequence of warfare or other political changes that can lead to abandonment of agricultural land (Witmer and O'Loughlin 2009), with open fields, meadows and pastures being subjected to natural succession or transferred to forestry (Rudel et al. 2000). Management actions, e.g. tree planting, and abiotic and biotic legacy from agriculture are important determinants for successional trajectories and for the rate of succession (Cramer et al. 2008). In cases where afforestation is a result of active plantation, native species in commercially unproductive open land like abandoned fields, mires, grasslands and other non-forested land are often replaced with faster growing species, not uncommonly alien species (Bremer and Farley 2010). The successional pattern of afforestation in the Baltic region has been very diverse (Ruskule et al. 2012; Matuszkiewicz et al. 2013).

## Changes in Biodiversity After Land Abandonment

Many studies on afforestation have found that a transition from open land to closed forest has a negative effect on biodiversity. A common feature in case studies showing negative effects is that the landscape transition proceeds from small-scale agricultural fields or open semi-natural grasslands to closed forests. Afforestation of semi-natural grasslands or natural shrublands and mires will, at least initially, have negative biodiversity effects, whereas forest plantation on degraded land or species-poor intensively managed agricultural fields may contribute positively to biodiversity (Bremer and Farley 2010). This implies that the effect of afforestation on biodiversity patterns depends on the biodiversity of the habitat that is replaced by forest (Bremer and Farley 2010; Amici et al. 2012; Graham et al. in press).

When open species-rich habitats are overgrown with shrubs and forest trees they lose many light-demanding plant species, causing a decline in species richness (Bremer and Farley 2010). The negative effect has been shown for different taxa such as birds (Allan et al. 1997; Lachance et al. 2005), vascular plants (Buscardo et al. 2008; Lachance et al. 2005) and bryophytes (Buscardo et al. 2008). Environmental changes with negative impacts for some organisms always leave room for others to improve their position. In a forest ecosystem the closing canopy results in shading but also in a more humid microclimate that may be beneficial for fungi, lichens and bryophytes. Afforestation has been shown to have positive effects on soil fungal biodiversity (Carson et al. 2010). Lichens and many species of bryophytes are organisms that have large affinity for old-growth forests. The response of epiphytic lichens to environmental change is suggested to be strongly dependent on available habitat (Kubiak 2013), and is rather slow (Johansson et al. 2013). However, lichen community structure in first-generation forests following natural afforestation in Estonia did not differ from long-term managed forests (Lõhmus and Lõhmus 2008), hence indicating a positive afforestation effect in the new forests. Afforestation following land abandonment may therefore lead to increased fungal and lichen biodiversity. Positive effects of afforestation have also been reported for spiders (Oxbrough et al. 2007), beetles (Oxbrough et al. 2010; Komonen et al. 2015) and ants (Komonen et al. 2015). However, changes in invertebrate species richness also depend on the stage of the abandoned habitat. Forest management can have different effects on different groups of invertebrates. Rotation forestry could be positive for carabid beetles, but negative for spiders (Oxbrough et al. 2010).

## Changes in Nutrient Emissions, Carbon Balance and Groundwater Supply

In connection with abandonment of agriculture at the time of the break-up of the Soviet Union, the use of fertilizers was reduced substantially. The application of mineral fertilizers decreased in Estonia by 80–85% between 1989 and 1994 (Löfgren et al. 1999). According to official Estonian statistics, ammonia and nitrous oxide emissions of Estonian agriculture decreased by 60% between 1990 and 1998 and have been approximately stable after that (Statistics Estonia 2016). The decrease of agricultural land area in the catchment of Lake Peipsi, at the border of Estonia and Russia, has considerably decreased the nitrogen and phosphorus emissions to the aquatic ecosystem (Mourad et al. 2005; Mourad et al. 2006; Iital et al. 2005). Van Rompaey et al. (2007) found that soil erosion reduced and sediment delivery to water bodies decreased with up to 75% in the Czech Republic in the 1990s due to agricultural land abandonment. Other studies from Estonia, Latvia, Lithuania, and Hungary have shown inconsistent results with a mixture of reduction of nutrient emissions and the absence of downward trends (Tumas 2000; Stålnacke et al. 2003; Stålnacke et al. 2004). It has been suggested that there is a long-time lag between the abandonment of agricultural land and the reduction in aquatic emissions (Grimvall et al. 2000; Tumas 2000). Changes in nutrient loads in natural ecosystems are, of course, also affected by new environmental policies and treaties. However, nutrient load reduction in the Baltic states was found to be much higher than in EU countries lacking recent land abandonment.

Many areas in Eastern Europe are carbon sinks due to afforestation following agricultural land abandonment. From this perspective, afforestation offers an opportunity for large-scale climate-change mitigation (Vuichard et al. 2008; Kuemmerle et al. 2011; Kuemmerle et al. 2015). After agricultural land abandonment, the carbon balance is at first stable or negative due to carbon loss from former arable land and low productivity in early succession, but turns to substantial carbon sequestration after 7–8 years (Schierhorn et al. 2013). Most of the carbon accumulation occurs in the soil. Accumulation in wood happens later in succession, when bush vegetation develops to a more mature forest, which has not yet occurred in the regions with recent land abandonment after the Soviet collapse. Due to agricultural land abandonment, carbon sequestration in Russia during the period 1990–2009 represented 4% of all global carbon emissions due to deforestation and land use changes (Schierhorn et al. 2013; Kurganova et al. 2014). Similarly, Romania (Olofsson et al. 2011) and Ukraine (Kuemmerle et al. 2011) are net terrestrial carbon sinks.

Water scarcity is a global problem and of large importance in the Baltic region. Large areas around the Baltic Sea are facing groundwater shortage, and there is a risk of the situation becoming worse due to global warming (Luoma and Okkonen 2014). Afforestation is an important factor for groundwater supply, especially in arid and semi-arid conditions (Lu et al. in press). The presence of a forest can potentially secure and sustain the water supply of a region (Ellison et al. 2012). However, there is always a fine balance between the forest's own water consumption and the humidification of the habitat. In China there exist large-scale forest plantations in arid regions to stop desertification (Lu et al. in press). The tree species selected for these programmes were not selected to suit local environmental conditions. The evapotranspiration of trees was higher than precipitation, resulting in a negative water budget. Loss of groundwater is a major threat to the socio-economic development of China, and the situation is similar in other regions of the world.

Forest trees may have a remediation effect on polluted soils. There are good examples of phytoremediation of heavy metal pollutions such as lead and cadmium (Pei et al. 2015). Phytoremediation using green plants to remove pollutants is an affordable and simple technique to remediate polluted areas, especially when time is not a factor (Pei et al. 2015). At contaminated sites it is mainly the soil that is contaminated. Allowing trees to grow can decrease or stop soil erosion (van Dijk and Keenan 2007; Ponette-González et al. 2015), and phytostabilization can in turn decrease the leakage of pollutants into groundwater and run-off to streams and larger water bodies.

## Agricultural Land Abandonment and Views of Landscape Change

How should the research findings of environmental effects of agricultural abandonment be applied in the management of rural landscapes? One possible interpretation of the research results is that there is no reason to be especially worried about land abandonment, because closing vegetation and growing forests can contribute to carbon sequestration, which helps mitigate climate change (Schierhorn et al. 2013; Kurganova et al. 2014). Land abandonment may also decrease nutrient emissions to water bodies, although its contribution in comparison to other emission sources is not clear (Stålnacke et al. 2003, 2004). When it comes to biodiversity, there is no straightforward answer to whether and when the effect of land abandonment is positive or negative (Queiroz et al. 2014). Land abandonment can create more natural forest habitats, which are beneficial for forest species. There are instances where afforestation has had negative environmental effects, such as the study of Lu et al. (in press) on the negative effect of forests on groundwater supply in arid parts of China. Agricultural habitats with high biodiversity value, especially hay meadows, are disappearing (Johansson et al. 2008; Pitkänen et al. 2016). A decrease in the area of semi-natural grasslands is regarded as a high biodiversity concern in Europe (Silva et al. 2008). However, a decrease in biodiversity due to loss of traditional farming can be seen as a problem that is distinct from post-Soviet agricultural land abandonment. In Estonia, in the 1930s, a third of the land surface consisted of semi-natural grasslands (Kukk and Kull 1997). Most of these grasslands have now disappeared, and many species of these habitats are threatened. Semi-natural grasslands have lost their function as a fodder source and cannot be recovered, irrespective of economic system. Old human-made grasslands are still used as pastures for grazing, but to a much smaller extent than in the historical farming landscape of Northern Europe. Although pastures have high biodiversity, they differ from meadows (Saarinen and Jantunen 2005). This implies that in the modern farming landscape, hay meadows can only be saved as cultural artefacts through external support (Kleijn and Sutherland 2003).

The problem of deriving conclusions for management from these kinds of studies is that they usually lack a critical discussion of the societal and cultural values embedded in different types of cultural landscapes and trajectories of rural development. While an extensive analysis of the management of agricultural land abandonment is outside the scope of this chapter, because it would require a broader discussion of rural change, we do point out links between environmental effects of agricultural land abandonment and the goals of socio-economic development of landscapes. We use here Soliva's (2007) ideal type narratives for rural studies. These narratives are based on values and assumptions of stakeholders in the Alpine landscapes, but this methodology is also useful in the context of Eastern Europe. The four narratives are: (1) a wilderness narrative that sees the decline of rural economy as a possibility for new wilderness reserves, (2) a modernization narrative emphasizing industrialized agriculture, (3) a subsistence narrative seeing the future in self-sufficiency, and (4) an endogenous development narrative that promotes new economic activities through local initiatives and participation.

These narratives cover different aspects of socio-economic development, land abandonment and biodiversity. In the wilderness narrative, management should be directed to increase the value of prioritized biodiversity. Agricultural land abandonment is seen as a possibility to increase the cover of wilderness reserves. Ongoing discourse on the rewilding of Europe (Navarro and Pereira 2012) fits well into this narrative. On the one hand, grassland conservation goals are in accordance with the wilderness narrative, while on the other, grasslands are not wilderness but a part of traditional cultural landscapes where active management is needed. The wilderness narrative has a potential problem, that is, abandoned agricultural land will not necessarily become wilderness. In Sweden and Finland, agricultural land that was abandoned during the twentieth century mostly became production forests with low biodiversity value, and the Baltic states have followed the same afforestation strategy (Jõgiste et al. 2015). The endorsement of the use of abandoned areas for commercial forest management is compatible with the modernization narrative that views abandonment of marginal land as a necessary consequence of economic development of agriculture. In the modernization narrative, agricultural land with high productivity should be used for large-scale intensive agriculture, often requiring subsidies. A foreseeable consequence of this narrative is that biodiversity conservation is left to play only a marginal role.

The subsistence and the endogenous development narratives are anthropocentric, but they include a utilitarian view where biodiversity, and diversity in general, are seen as positive for local livelihoods. Conservation of grasslands is compatible with these narratives. The subsistence narrative strongly opposes land abandonment and sees it as a loss of cultural heritage and traditions. According to this narrative, it is not important if some aspects of agricultural land abandonment, such as nutrient retention and carbon sequestration, can be regarded as environmentally positive. The cultural aspects are in focus and override environmental concerns. The views following this narrative may be enhanced by the common belief that traditional methods are more environmentally friendly than modern farming techniques. The endogenous development narrative is more open to dynamic changes in the landscape. A focus on local, small-scale sustainable development includes the investments in tourism and recreation using a landscape that can be a mixture of old and new elements and also contain wilderness patches. The endogenous development narrative is thus more flexible from a viewpoint of environmental management and not necessarily hostile to new technology.

Agricultural land abandonment is a large-scale and complicated phenomenon, with local, regional and global effects. In discussions about the environmental governance measures needed to respond to land abandonment, the stated goals are often unclear and conflicting. The implications for biodiversity and ecosystem processes may be multifaceted, especially when broader issues of rural land use are added to the discussion. As is often the case with environmental issues, facts and values are inextricably linked. The different perspectives offered by the four narratives mentioned above may help to illuminate possible outcomes of different management strategies and hence help decision-makers navigate the issue of abandonment.

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# 8

# An Analysis of Permission Processes for Wind Power in Sweden

Stig Blomskog

## Introduction

Wind power is an important sustainable and renewable energy resource in the Baltic Sea Area as well as in the rest of the world. However, the expansion of wind power gives rise to value conflicts with other environmental values. Besides these value conflicts, coined as 'green versus green', expansion of wind power gives rise to conflicts with other competing interest for exploiting land-areas suitable for wind power. These value and interest conflicts give rise to complex and long lasting planning and legal permission processes concerning wind power. The processes are regulated by Environmental Codes. Since all countries in the Baltic Sea Area are EU members, it can be assumed that environmental legislation and permission processes in these countries have a similar content. In this chapter, we focus on a typical permission process as it is carried out in Sweden.

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A permission process starts when a wind power entrepreneur has submitted an application for a permit to establish a wind power station on a specific land-area. The outcome of a permission process is that the authority *issues* or *rejects* a permit for the planned wind power station. A permit may be rejected if the authority assesses that some of the expected negative effects of a planned wind power station give rise to significant value losses or, in other words, non-acceptable value losses. Such significant value losses might be due to an expected negative impact on landscape and wildlife, an increased noise, a detrimental effect on tourism, a decreased property values and so on.[1] The rejection of a permit is thus based on some kind of *critical threshold values,* which the authority has to determine and apply in the case of each permission process. However, even if no significant value losses arise, a permit may nevertheless be rejected if the authority assesses that the overall value losses *outweigh* the overall value gains as an increased production of 'green' electricity. This means that a permit to establish a wind power station seems to require, at least, an assessment that the overall value gains *outweigh* the overall value losses.

Needless to say, these permission processes give rise to complex decision problems as well as intricate conceptual problems. However, the extensive academic analyses of these processes are, at least in Sweden, performed in terms of an informal everyday language (see Bergek 2010; Petterson 2008). Using an informal language may hamper the correct interpretation of various fundamental conceptual problems, which arise in these permission processes. An example of such a defective analysis is that the weighing of value gains and value losses is commonly analysed in terms of a weight-scale metaphor. Such a metaphor does not, of course, contribute to an understanding of what is at stake in the very important weighing decisions, the purpose of which is to solve intricate and complex value conflicts. Other conceptual problems are how to specify the basic decision problem in a permission process and the adherent value conflicts, which are expected to arise at a wind power establishment. Further, the use of critical threshold values gives rise to intricate conceptual problem, which is difficult to understand within an informal conceptual framework.

Starting from these critical observations the purpose of this chapter is to, in the *first place,* reconstruct a typical permission process by means of formal conceptual framework, which is applied in Multi-criteria Decision Analysis (a classical reference is Keeney and Raiffa 1993, a more up to date reference is Figueira et al. 2005). Based on this formal reconstruction we will interpret and specify the basic decision problem and the adherent value conflicts. *Secondly*, we will interpret the application of critical threshold values and weighing decisions, which are used in order to solve the value conflicts. We end the analysis by applying the formal conceptual framework on an example of a permission process.

We do not know of any previous studies that have attempted to reconstruct or, in other words, to translate these legal permission processes into a more formal conceptual framework. Our attempt can therefore be regarded as an experiment. This means that the chapter will therefore have an introductory character of a very complicated legal decision problem. We therefore have to disregard from several important and intricate legal, formal and technical questions. The result of the chapter might be a starting point for further and more extensive analyses of these legal permission processes based on multi-criteria decision analysis. Our hypothesis is that a formal reconstruction can improve the understanding of various fundamental conceptual problems, which arise in these permission processes. A correct interpretation of these conceptual problems is of course important in order for the outcome of a permission process to be based on rational and well-argued judgements and decisions.

The chapter is organized as follows. In the *next* section, we give a summary of a typical permission processes and how it is regulated by the Swedish Environmental Code. In the *third* section, we give a brief introduction of the Multi-Criteria Decision Theory (MCDT) and some formal concepts, which we will use in the reconstruction. In the *fourth* section, we reconstruct a typical permission process in five stages. In the last stage, we apply the concepts of the reconstruction on a concrete and to a certain extent realistic example of a permission process. In the *fifth* section, we summarize the discussion in the chapter.

## Permission Processes for Wind Power: The Case of Sweden

According to the Swedish Environmental Code, a wind power entrepreneur who intends to establish a wind power station on a piece of land has to apply for a permit.[2] The application for a permit must be submitted to the County Board Administration. An important part of the application is the Environmental Impact Assessment (EIA), which has to be designed by the wind power entrepreneur. The content of an EIA is regulated by the Swedish Environmental Code (Swedish Ministry of the Environment and Energy 2000; see chapter 6 in the Swedish Environmental Code; Canter 1996).[3] The purpose of an EIA is to describe the direct and indirect impact of a planned wind power station on people, animals, plants, the landscape and cultural environment. It also contains a description of possible alternative sites for the wind power station and the reason for the choice of the land-area specified in the application. An important formal requirement of an EIA is that it must contain all information that makes it possible for the authority to make a rational and well-informed decision concerning a permit. Not fulfilling all formal requirements of EIA is a common reason for the rejection of permits.

The assessment of the application and the adherent EIA is delegated by the Count Board to an Environmental Assessment Committee. The implementation of the permission process is regulated by guidelines stated in the Environmental Code, which includes general and introductory guidelines in Chapter 1. Section 1, "Objectives and area of application of the Environmental Code", is commonly named the "portal paragraph". It begins with the following statement: "The purpose of this Code is to *promote sustainable development* which will assure a healthy and sound environment for present and future generations" (emphasis added).

This general statement provides grounds for issuing a permit to a wind power station, because wind power is defined as a sustainable energy resource, which promotes sustainable development. However, other statements in the section provide grounds for rejecting a permit. The section continues as follows:

"The Environmental Code shall be applied in such a way as to ensure that:

1. human health and the environment are protected against damage and detriment, whether caused by pollutants or other impacts.
2. valuable natural and cultural environments are protected and preserved.
3. biological diversity is preserved
4. the use of land, water and the physical environmental in general is such as to secure a long term good management in ecological, social, cultural and economic terms.
5. Reuse and recycling, as well as other management of materials, raw materials and energy are encouraged with a view to establishing and maintaining natural cycles."

The first three points are obviously grounds for rejecting the issuance of a permit. For example, a wind power station can have a detrimental impact on human health due to the disturbing noise and shadowing effects. It is also likely that a wind power station will have a negative impact on the cultural and aesthetic value of the landscape. Further, point three indicates a common reason for rejecting a permit. For example, a wind power station located close to an eagle's nest might increase the risk of mortality of eagles, which is a species protected by the European Commission (European Commission 2016). The five points stated in the portal paragraph summarize the basic and intricate decision problems, which the Committee has to solve. The basic decision problem is to solve the value conflict which will emerge when the points stated in the portal paragraph are applied.

The Environmental Code contains guidelines for the Committee to solve these intricate value conflicts. According to the Code, value conflicts should be solved in two stages. First, the Committee shall assess if the planned wind power station is expected to cause any kind of "significant damages". If the Committee judges that some sort of significant damage will arise, the permit shall be rejected. This means that the Committee has to determine and apply "critical threshold values". However, if the Committee pronounces that no significant damages will arise, this leads to the second stage of the permission process. In the second stage, the Committee has to weigh overall value losses against overall value gains which are expected to arise if the planned wind power station receives a permit. This weighing process is, of course, a demanding cognitive task. The permit will be rejected if the Committee finds that overall value losses *outweigh* the overall value gains. This means that a necessary condition for a permit to be issued is for value gains to outweigh value losses. But this condition is only necessary since a number of formal conditions also have to be fulfilled in order for the Committee to issue a permit. For example, in Sweden the municipals have a veto right against the establishment of wind power stations. These two intricate decision stages—application of critical threshold values and weighing decisions—the purpose of which is to solve value conflicts, are analysed below through formal reconstruction.

We conclude by pointing out that appeals against the decision of the Environmental Assessment Committee are common. This means that the permission process has to be repeated in the Land and Environmental Court, which examines the application anew. Appeals against decisions of the Land and Environmental Court are also common, which means that the application will be re-examined in the Land and Environmental Court appeal. The difference between the three permission processes is that various facts may be added or deleted. However, from a formal point of view, there is no difference between the permission processes in these three instances. The intricate decisions about weighing reasons in order to issue a permit against reasons for rejecting a permit will have the same structure in all these instances.

## Multi-criteria Analysis: A Brief Introduction

In this section, we introduce some basic concepts which are applied in Multi-criteria Analysis (MCA) and MCDT. A complete and detailed explanation of the basic concepts requires a more advanced mathematical and logic conceptual framework, which is beyond the scope of this chapter (for a survey of various models and applications based on MCDT, see Figueira et al. 2005).

A very brief summary of a multi-criteria decision process is as follows. A decision-maker (DM) judges or decides how good or suitable a number of alternatives $a_1, a_2, \ldots, a_n$ are for a certain purpose. The overall judgement of the alternatives is based on facts about the alternatives, which are described in terms of various aspects (attributes or criteria), which we denote as: $\alpha_1, \alpha_2, \ldots, \alpha_k$. The final judgement is reached after a number of stages. In the first stage, the DM evaluates the alternatives to each of the aspects. These basic and partial value judgements are then aggregated in a number of stages into an overall value judgement of the alternatives. The aggregation process can be complicated, particularly if there are many alternatives and aspects involved in the process. The overall judgement can be interpreted as a judgement of an aggregated aspect, which we denote as $\alpha_o$. It is essential to understand that the aspect $\alpha_o$ is *not* a descriptive aspect but has to be interpreted as an *overall value aspect*, which represents the overall goodness of the alternatives. The outcome of the process is a *rank-order* of the alternatives regarding aspect $\alpha_o$. If an alternative $a_i$ is *overall better than* an alternative, $a_j$ can formally be represented as:

$$a_i \succ_{\alpha_o} a_j,$$

where the expression "$\succ_{\alpha_o}$" is to be read as "overall better than".

In aggregation processes in multi-criteria problems, the following three concepts are frequently used: *ordinal value judgements*, *cardinal value judgements* and *inter-factorial comparisons.* We employ these concepts in our formal reconstruction of the permission process. In order to explain the functioning of these concepts we use a simple example. Let us assume that the DM has to make a choice between two alternatives, *A* and *B*. The DM has to decide which of the alternatives is best for a certain purpose. The overall evaluation of the alternatives is based on the two descriptive aspects, $\alpha_1$ and $\alpha_2$, which can state facts about the alternatives. In the first stage of the decision process, the DM evaluates the alternatives with respect to each of the aspects. The outcome of the partial evaluations is that regarding aspect $\alpha_1$ alternative *A* is better than alternative *B*, which we formally denote as

$$A \succ_{\alpha_1} B$$

and regarding aspect $\alpha_2$, alternative *B* is better than *A*, i.e.

$$B \succ_{\alpha_2} A.$$

The first partial and *ordinal value judgement* regarding $\alpha_1$ is reason for a claim that *A* is *overall better than B*. It is important to note that this ordinal value judgement does not imply anything about the value differences between *A* and *B* regarding aspect $\alpha_1$. The second partial and ordinal value judgement regarding aspect $\alpha_2$ is a reason for the claim that *B* is overall better than *A*. Obviously, the DM has to solve a value conflict. This means that the DM has to take a weighing decision. We assume that the outcome of the DM's weighing decision is that *A* is overall better than *B*, i.e.

$$A \succ_{\alpha_o} B.$$

This weighing decision implies that the DM has judged that the *value difference* between *A* and *B* regarding $\alpha_1$, which we denote as

$$\Delta_{\alpha_1}(A, B)$$

is *greater than* the value difference between *B* and *A* regarding $\alpha_2$, which we denote as

$$\Delta_{\alpha_2}(B, A).$$

The weighing decision in terms of a so-called *inter-factorial comparison* can be stated as

$$\Delta_{\alpha_1}(A, B) \succ^* \Delta_{\alpha_2}(B, A),$$

where "$\succ$*" is to be read as "is greater than". This judgement is a so-called *cardinal value judgement,* which has to be distinguished from ordinal value judgements. A cardinal value judgement implies comparison between two value differences. This example demonstrates that in order to solve this kind of value conflict, it is not sufficient to merely make ordinal value judgements, which may be less cognitively demanding than cardinal value judgements.

The outcome of the weighing decision can be represented by the implication:If $\Delta_{\alpha_1}(A, B) \succ^* \Delta_{\alpha_2}(B, A)$, then $A \succ_{\alpha_o} B$.

We conclude the example by commenting on a very common mistaken interpretation of weighing decisions. In this example, it would be that the outcome of the weighing decision is that the DM has judged that aspect $\alpha_1$ is *more important* than $\alpha_2$. But using the notion of importance in order to justify or explain the outcome, a weighing decision in this context leads only to confusions. The justification of the outcome of the weighing decision has to be based on a comparison of a value difference in aspect $\alpha_1$ with a value difference in aspect $\alpha_2$. This means that to base the weighing on some kind of intuition about the relative importance of aspects, without considering the value differences in the specific decision situation, could give rise to not-very-well-argued weighing decisions. The famous decision theorist R.L. Keenye proposed the concept of "importance" in multi-criteria decision problems as "the most common mistake" (Keenye 2002). There is also extensive research in MCDT about the weighing behaviour of subjects and common mistakes made in the weighing processes (a seminal reference of this research is Weber and Borcheding 1993).

## Formal Reconstruction of a Permission Process

In this section, we reconstruct a typical permission process, taking an application for a permit to establish a wind power station. The reconstruction is based on the formal concepts introduced above. Reconstruction is carried out in five stages. In the *first* stage, the basic decision problem is specified. In the *second* stage, we explain the value conflict, which has to be solved by the Environmental Assessment Committee (named as DM in the reconstruction). In the *third* section, we reconstruct the first way of solving the value conflict based on so-called *critical threshold values*. In the *fourth* stage, we reconstruct the second way of solving the value conflict, based on *weighing decisions*. In the *fifth* stage, we apply the conceptual framework to an example of a permission process.

## Stage 1: Specification of the Basic Decision Problem in a Permission Process

The basic decision problem in a permission process is whether the use of the land-area, which is specified in the application, is suitable for wind power, compared to alternative uses. The decision problem can be specified as follows:

There are two alternative uses of the land-area:

1. Alternative *W* = "*using* the land-area for wind power",

and

2. Alternative *Not-W* = "*not using* the land-area for wind power".

The alternative *not-W* can be regarded as a status quo or null-alternative. The decision problem can be formally stated as:

Is it the case that: $W \succ_o not-W$?

$('' \succ_0'')$ means "is overall better than")

Besides a decision such that $W \succ_o not-W$, a permit requires that a number of formal conditions are fulfilled. The conditional norm for a permit can therefore be stated as:

If it is the case that *the formal conditions are fulfilled* and that $W \succ_o not-W$, then the authority *shall* give a permit to the wind power entrepreneur.

Given that the conditions in the antecedent of the conditional norm are fulfilled we assume that no further decisions are required in order for a permit to be issued.

The intricate and complex question, which is the focus of this chapter, is how to understand the decision process, the outcome of which might be that $W \succ_o not-W$ or that $not-W \succ_o W$.[4] What kind of decisions and judgements are involved in reaching these decisions? What are the empirical and non-empirical grounds for the decisions and judgements taken by the DM? In what way are these various decisions and judgements related to each other? There is, however, no possibility in this chapter to make a more profound analysis of all these complicated questions. Such analysis requires a more formal language than that used in this chapter.

As pointed out in the introduction, this chapter should be regarded as a first attempt to reconstruct or, in other words, to translate a typical permission process into more formal language. Our hypothesis and motivation for the reconstruction is that by using formal language, the knowledge of the decision problems in the permission processes might be improved.

We conclude the first stage by pointing out that it is essential to understand that the Swedish Environmental Code only contains more or less general guidelines for the DM to conduct the permission process. This means that the permission process is permeated by so-called *discretionary* judgements made by the DM.

## Stage 2: Specification of the Value Conflict

The origin of value conflicts that need to be solved in the permission process lies in the possible consequences of establishing a wind power station, i.e. by choosing alternative *W*. Consequences that give rise to value gains from both a public and private point of view can be named *positive* consequences. By this, we refer to the impact of alternative *W* on an aspect (criterion or attribute), denoted as $\alpha_i^+$. An example of such an $\alpha_i^+$- aspect is the *production of "green" electricity*. Choosing alternative *W* will obviously have an impact on the aspect production of "green" electricity, which will give rise to public value gains. Other possible $\alpha_i^+$- aspects are *local employment rate* and *property value*, which might be influenced by alternative *W* such that value gains arise. However, the Environmental Code does not specify relevant $\alpha_i^+$- aspects in any detail. The Code gives only very general guidelines saying that measures that promote sustainable development should be supported (see chapter 1, Sect. 1 in the Code). It therefore seems to be an "open" question for the DM to decide what kind of $\alpha_i^+$-aspects should be considered in the permission process.

A value gain due to the possible impact of alternative *W* on aspect $\alpha_i^+$ can be more precisely stated as an *ordinal value statement* in the following manner:

Regarding an aspect $\alpha_i^+$, alternative *W* is *better than* alternative *not-W*, $i = 1, 2 \ldots m$.

This ordinal value statement can be formally stated as:

$$W \succ_{\alpha_i^+} not\!-\!W,$$

where $\succ_{\alpha_i^+}$ is to be read as "Regarding aspect $\alpha_i^+$ …. is better than …."

These ordinal value statements regarding all identified $\alpha_i^+$- aspects are reasons for the statement that $W \succ_o^v not\!-\!W$. This can be formally stated as:

If—ceteris paribus—$W \succ_{\alpha_i^+} not\!-\!W$, then $W \succ_o^v not\!-\!W$, where the expression "ceteris paribus" represents the assumption that *W* and *not-W* are *equal* with respect to all other relevant aspects.

A *negative* consequence is defined as a consequence in terms of the impact of alternative *W* on an aspect, denoted as $\beta_i^-$, such that a value loss arises. Examples of $\beta_i^-$-aspects are *noise*, *risk for bird mortality rate*, *view of the landscape*, *competing interests for exploiting the land-area* and so on. The Swedish Environmental Code provides a relatively detailed specification of aspects related to value losses which should be considered in the permission process. Which $\beta_i^-$- aspects are assessed as relevant in various permission processes is, of course, situation dependent. For example, an aspect as interest related to reindeer husbandry would not be relevant if there is no reindeer husbandry in the surroundings of the land-area meant for the planned wind power station.

The value losses due to the impact on aspect $\beta_i^-$ of alternative *W* can be stated as:

Regarding an aspect $\beta_i^-$, alternative *W is worse than* alternative *not*-W, $i = 1, 2, \ldots, n$.

The value losses can also be stated as:

Regarding an aspect $\beta_i^-$, alternative *not-W is better than* alternative *W,* $i = 1 \ldots n$.

In subsequent analysis, we will use this latter way to represent value loss-related $\beta_i^-$- aspects. The value loss can be formally stated as:

$$Not-W \succ_{\beta_i^-} W,$$

which is a reason for the statement that $Not-W \succ_o W$. This can be formally stated as:

If—ceteris paribus—$not-W \succ_{\beta_i^-} W$, then $not-W \succ_o W$.

When the DM has identified all relevant $\alpha_i^+$- and $\beta_i^-$- aspects we can say that the DM has specified the value conflict. The next stage in the decision process is to solve the value conflict. As mentioned above, the Swedish Environmental Code supports two ways of solving the value conflict—application of *critical threshold values* and *weighing decisions*. In the next stage, we reconstruct the application of critical threshold values.

## Stage 3: Solving the Value Conflict by Means of Critical Threshold Values

In multi-criteria decision analysis, it is common for critical threshold values to be applied. The application of critical threshold values is an effective way to exclude from the decision process alternatives that, in the DM's assessment, give rise to severe value losses (see Mendoza et al. 2002, and Belton and Stewart 2002). To use critical threshold values seems, at first glance, to be less cognitively demanding than taking weighing decisions. The DM needs only to make *ordinal value judgements* as to what extent an alternative might give rise to value losses below or not below the determined critical threshold values. We claim that the application of critical threshold values in permission processes gives rise to complicated conceptual problems. However, a more profound analysis of these conceptual problems requires the use of formalized semantic theories, which is beyond the scope of this chapter. The purpose of our reconstruction below is to give a first intuition of the conceptual problems related to the application of critical threshold values in permission processes.

The guidelines of the Swedish Environmental Code state that the DM should apply critical threshold values concerning the impact on the $\beta_i^-$- aspects, which gives rise to value losses. The application of a critical threshold value can be expressed as a conditional statement in the following manner:

1. If the impact of alternative *W* on a $\beta_i^-$- aspect is expected to give rise to *a non-acceptable value loss*, which is below *a critical threshold value*, then the authority *shall* reject a permit.

This means that a sufficient condition for rejecting a permit might be that non-acceptable value losses arise regarding only one $\beta_i^-$- aspect. On the other hand, this means that a necessary condition for granting a permit is that *no* non-acceptable value losses arise for any of the $\beta_i^-$- aspects. But it can, of course, be the case that a permit is rejected even if all necessary conditions are fulfilled. The DM might argue that the overall value loss will give rise to an overall non-acceptable value loss. This means that besides a critical threshold value for each $\beta_i^-$- aspect, the DM can also apply an overall critical threshold value. At least according to our reading, such argumentation for rejecting a permit seems not be ruled out by the Environmental Code.

However, the intricate issue with the conditional statement (1) above, representing an application of critical threshold values, is that it is not an *operational norm*, because the antecedent in the conditional is a *value statement* with *no empirical content*. This means that the conditional statement cannot give any guidance to the DM. It can only be regarded as a partial definition of the expression "to give rise to non-acceptable value losses", which implies, due to the meaning of the expression, the norm that the authority shall decline the permit.

To make such a critical threshold value so-to-speak operational, a crucial question arises: Is it possible to define *empirical criteria* as grounds for application of critical threshold values regarding the $\beta_i^-$- aspects? The possibility to stipulate such empirical criteria depends on the extent to which the aspects in question are measurable. If an aspect is measurable using an objective measurement method, it would be easy to stipulate an empirical criterion in terms of a measure, which would make it simpler for the DM to interpret and communicate it to other parties involved in the permission process. Of course, the normative question remains to decide at which level the measure would be suitable as an empirical criterion for a critical threshold value.

However, the problem is that most of the aspects in the permission process related to value losses are not measurable in an objective sense. A possible solution to this problem is to stipulate empirical criteria that can be interpreted as *indicators* for when the impact on the aspects give rise to non-acceptable value losses. As an example of an indicator, we can try to stipulate an indicator for the aspect *noise*. It is common praxis in permission processes that *loud measure* in the nearest housing area to the planned wind power station is used as indicator of the noise that might be caused by the wind power station. A common praxis in Swedish legal permission processes is that loud levels beyond 40 dB(A) should be regarded as disturbing noise, i.e. the loud level 40 dB(A) is an indicator for a critical threshold value. This reasoning gives rise to the following operational norm:

2. If a wind power station is expected to cause a *loud level above 40 dB(A)* in the nearest housing area, then the impact on the aspect noise gives rise to a *non-acceptable value loss*.

If we combine conditional statement (1) above with conditional statement (2) the following operational norm is implied:

3. If a wind power station is expected to cause a *loud level above 40 dB(A)* in the nearest housing area, then the authority *shall* reject a permit to establish a wind power station.

An obvious problem with a threshold level like 40 dB(A) is that the aspect noise and aspect loud do not share the same meaning. Noise is a kind of subjective aspect, whereas loud is an objective aspect which can be measured by objective measurement methods. The measurement of noise is ultimately based on subjective experiences of loud. The problem with an empirical criterion like 40 dB(A) is that the subjective experience of loud interact with other circumstances. For example, a loud level of 40 dB(A) caused by one wind power station, given certain circumstances, can be experienced subjectively as disturbing noise, whereas 40 dB(A) caused by another wind power station under other circumstances might *not* be experienced by the same subjective as disturbing noise. For example, it seems to be the case that the experience of loud as noise depends on whether the wind power station is visible or not from the nearest housing area (see Bakker et al. 2012). This means that an empirical criterion, such as 40 dB(A), used as an indicator for the critical threshold value regarding the aspect noise has to be extended by some kind of clause, which we can name "under normal conditions". The conditional norm can now be stated as:

4. If—*under normal circumstances*—a wind power station is expected to cause *a loud level above 40 dB(A)* in the nearest housing area, then the impact on the aspect noise gives rise to a non-acceptable value loss.

However, this extension of the conditional statement gives rise to the next problem, which is how to interpret "under normal circumstances". The clause "under normal circumstances" cannot reasonably be interpreted as a general rule that can, in each permission process, yield a decisive answer if the circumstances are normal or not normal. We claim that it is more reasonable to interpret a clause like "normal circumstances" as some kind of reminder that in each permission process, the DM has to make discretionary judgements as to what extent the circumstances are regarded to be normal or not normal. Such a line of reasoning indicates that the application of critical threshold values is ultimately more or less based on discretionary and situation-dependent judgements. This means that before taking a decision in a specific permission process, if some value losses are expected to be below critical threshold values, the DM has to assess the way in which the value losses interact with the situation-dependent circumstances.

Another problem regarding the possibility of defining empirical criteria for critical threshold values is that many of the aspects in the permission process are constituted by a number of sub-aspects, which are unlikely to be objectively measurable. One example of such an aspect is *view of the landscape*. There seems to be a serious demand to stipulate empirical criteria for a critical threshold value for such a multidimensional aspect. In the first place, for each sub-aspect, an empirical criterion has to be stipulated. And these empirical criteria have to be regarded as indicators for the critical threshold values regarding the sub-aspects, which are in turn the grounds for critical threshold value regarding the multidimensional aspect. Further, even if the DM succeeds in defining such a complicated and multidimensional empirical criterion, the application of critical threshold value would be—similar to the aspect noise—sensitive to the specific circumstances of the land-area where the wind power station is planned.

This line of reasoning strengthens our claim that the application of critical threshold values is ultimately more or less based on discretionary and situation-dependent judgements. This means that stipulating certain standardized critical threshold values which can be applied in all or many various permission processes does not seem to be a possibility. The application of standardized critical threshold values would of course make the permission process less cognitively demanding for the DM and make the outcome of the permission process more predictable, which would be in the interest of wind power entrepreneurs.

We conclude that the application of critical threshold values is ultimately based on discretionary and situation-dependent judgements. Solving the value conflict by means of critical threshold values seems to be a cognitively demanding process in parity with weighing decisions, which we discuss in the next stage.

## Stage 4: Solving the Value Conflict by Means of Weighing Decisions

We start the formal reconstruction of a weighing process by discussing an important and fundamental formal difference between applying critical threshold values and taking weighing decisions. When critical threshold values are applied, it is sufficient to base the decision on ordinal value judgements. The DM has only to decide if the value loss is below or above a certain critical threshold value. However, a weighing decision has to be based on cardinal value judgements, because the

DM's weighing decision has to be based on a comparison of value difference across aspects. Such comparisons are named inter-factorial comparisons in MCA (see Section "Multi-criteria Analysis: A Brief Introduction").

A common mistake, also mentioned in Section "Multi-criteria Analysis: A Brief Introduction", is for a DM to base a weighing decision on some intuition about the relative importance of certain aspects without considering value differences across aspects. Such a weighing decision is based on a profound conceptual mistake and should be avoided. We now know the frequency of such mistakes in the permission process. However, there are reasons to believe that a DM, i.e. an Environmental Assessment Committee, makes this kind of conceptual mistake due to the fact that weighing decisions are represented in an informal everyday language. As pointed out in the introduction, it is common to represent weighing decisions in terms of a weight-scale metaphor, which can be very misleading and could compromise the quality of weighing decisions.

In order to properly reconstruct a weighing decision, we introduce notations for cardinal value statements. Further, to make the reconstruction of a weighing process tractable, we assume that there is only one relevant $\alpha_i^+$- aspect, denoted as $\alpha_1^+$, and two relevant $\beta_i^-$- aspects, denoted as $\beta_1^-$ and $\beta_2^-$.[5] The reason for this simplification is that reconstruction of a weighing process based on an arbitrary number of aspects becomes very cumbersome. We argue that the principle problems arising in a weighing process can be explained by means of this simplification.

The cardinal value statement regarding an $\alpha_1^+$- aspect can be denoted as:

$$\Delta_{\alpha_1^+}(W, not{-}W).$$

This cardinal value statement shall be read as follows: "regarding aspect $\alpha_1^+$, the *value gain* that arises if alternative *W* is chosen for alternative *not-W*". Regarding the $\beta_i^-$- aspect we denote the cardinal value statements as:

$$\Delta_{\beta_i^-}(not{-}W, W)$$

A somewhat tricky question is how to read this cardinal value statement without causing too much confusion. One reading is: "Regarding the aspect $\beta_i^-$, the *value gain* that arises if alternative *not-W* is chosen for alternative *W*." Another reading is: "Regarding the aspect $\beta_i^-$, the *value loss* that arises if alternative *W* is chosen for alternative *not-W*." Below we discuss why we prefer the latter reading.

The aggregation or combination of these two value losses into an overall value loss is denoted as:

$$\Delta_o(not-W, W) = \Delta_{\beta_1^-}(not-W, W) \oplus \Delta_{\beta_2^-}(not-W, W),$$

where "$\oplus$" = the concatenation operator used in MCA (function in analogy to the operator "+" applied on numbers).[6] We make a crucial assumption that negative interactions do not arise between the two partial value differences. This means that we exclude cardinal value statements as:

$$\Delta_{\beta_1^-}(not-W, W) \succ^* \Delta_{\beta_1^-}(not-W, W) \oplus \Delta_{\beta_2^-}(not-W, W)$$

and

$$\Delta_{\beta_2^-}(not-W, W) \succ^* \Delta_{\beta_1^-}(not-W, W) \oplus \Delta_{\beta_2^-}(not-W, W).$$

However, we have no knowledge about the extent to which such interactions would arise in permission processes. We suspect that due to the use of informal everyday language in permission processes, a DM, i.e. an Environmental Assessment Committee, is unable to express and consider such intricate relations as negative interactions between various kinds of value losses.

The value conflict that we stated in terms of ordinal value statements at the end of stage 2 can now be stated in terms of cardinal value statements in the following manner:

The value gain $\Delta_{\alpha_1^+}(W, not-W)$ is a reason for the statement that $W \succ_o not-W$.

And the overall value loss $\Delta_{\beta_1^-}(not-W, W) \oplus \Delta_{\beta_2^-}(not-W, W)$ is a reason for the statement that $not-W \succ_o W$.

To solve the value conflict a weighing decision has to be taken. This means that the value gain

$$\Delta_{\alpha_1^+}(W, not{-}W)$$

has to be compared with the overall value loss

$$\Delta_{\beta_1^-}(not{-}W, W) \oplus \Delta_{\beta_2^-}(not{-}W, W).$$

The possible outcome of the comparison is that the value gain outweighs the overall value losses, i.e.

$$\Delta_{\alpha_1^+}(W, not{-}W) \succ^* \Delta_{\beta_1^-}(not{-}W, W) \oplus \Delta_{\beta_2^-}(not{-}W, W)$$

or that the over value loss outweighs the value gain, i.e.

$$\Delta_{\beta_1^-}(not{-}W, W) \oplus \Delta_{\beta_2^-}(not{-}W, W) \succ^* \Delta_{\alpha_1^+}(W, not{-}W).$$

Such comparisons are obviously cognitively demanding tasks. One reason is that the overall value loss is an aggregation of partial value losses based on very different kinds of aspects, for example, the aspect noise and the aspect view of the landscape. And in realistic situations the number of aspects is usually significantly greater than two, as in this example. One way to mitigate the cognitive burden is to follow a stepwise weighing process. In the first step, the DM compares the value gain only against each partial value loss. It is sufficient in a simple example as this, to consider only two possible outcomes of these partial weighing decisions.

1. If the outcome of the weighing is that:

$$\Delta_{\beta_1^-}(W, not{-}W) \succ^* \Delta_{\alpha_1^+}(W, not{-}W)$$

or

$$\Delta_{\beta_1^-}(W, not{-}W) \succ^* \Delta_{\alpha_1^+}(W, not{-}W)$$

are—assuming that there are no negative interactions—sufficient reasons for the statement that

$$\Delta_{\beta_1^-}(not-W, W) \oplus \Delta_{\beta_2^-}(not-W, W) \succ^* \Delta_{\alpha_1^+}(W, not-W),$$

which implies that:

$$not-W \succ_o W,$$

which implies that:
the authority *shall* reject a permit.

2. However, if the outcome of the weighing is that

$$\Delta_{\alpha_1^+}(W, not-W) \succ^* \Delta_{\beta_1^-}(W, not-W) \text{ and } \Delta_{\alpha_1^+}(W, not-W) \succ^* \Delta_{\beta_1^-}(W, not-W)$$

and are only necessary conditions for the statement that $W \succ_o not-W$, the means that the DM has to in second stage weigh the value gain

$$\Delta_{\alpha_1^+}(W, not-W)$$

against the overall value loss

$$\Delta_{\beta_1^-}(not-W, W) \oplus \Delta_{\beta_2^-}(not-W, W).$$

If the outcome of the weighing is that:

$$\Delta_{\alpha_1^+}(W, not-W) \succ^* \Delta_{\beta_1^-}(not-W, W) \oplus \Delta_{\beta_2^-}(not-W, W),$$

this implies that

$$W \succ_o not-W,$$

which implies—given that all formal conditions are fulfilled—that
the authority *shall* issue a permit.

Obviously, a permit will be declined if the outcome of the weighing is

$$\Delta_{\beta_1^-}(not-W, W) \oplus \Delta_{\beta_2^-}(not-W, W) \succ^* \Delta_{\alpha_1^+}(W, not-W).$$

Our conclusion, based on this simplified reconstruction, is that the weighing processes, which is part of the permission process, is obviously cognitively demanding. There are therefore reasons to believe that the outcomes of weighing decisions based on informal everyday language are not necessarily well-argued. A possible way to improve this stage of the permission process is to employ tools that have been developed in MCDT and that can support the DMs, i.e. the Environmental Assessment Committees, in a systematic way.[7]

## Stage 5: Formal Reconstruction of a Realistic Example

We end the chapter by applying the formal concepts introduced above in order to reconstruct a realistic example. The example can be regarded as a very compact summary of a report delivered by an Environmental Assessment Committee. We begin by assuming that a wind power entrepreneur has submitted an application for a permit to establish a wind power station on a land-area specified in the application. The Committee decides to focus on the four $\beta_i^-$- aspects, i.e. aspects related to value losses: *noise*, *risk for bird mortality rate*, *view of the landscape,* and *competing interest of using the land-area.* The only $\alpha_i^+$- aspect is *production of green electricity*.

The aspects and adherent value gains and value losses are represented as follows:

*Aspects giving rise to value gains: Value gain*

- Production of "green" electricity (E) $\Delta_E^+(W, not{-}W)$

*Aspects giving rise to value losses: Value loss*

- Noise at the nearest housing area (N) $\Delta_N^-(not{-}W, W)$
- View of the landscape (V) $\Delta_L^-(not{-}W, W)$
- Risk for Bird mortality rate (R) $\Delta_B^-(not{-}W, W)$
- Competing interest as reindeer husbandry (I) $\Delta_I^-(not{-}W, W)$

The Committee initiates the permission process by stating the decision problem:

From an overall public point of view, is the alternative *W* better than the alternative *not-W*? Based on the guidelines of the Environmental Code, the Committee examines, first, if there are reasons to believe that non-acceptable value losses arise for any of the $\beta_i^-$-aspects. This means that the Committee tries to solve the value conflict by applying critical threshold values. Each aspect is examined as follows:

*Noise*: The Committee found it reasonable to stipulate a threshold level corresponding to 35 dB(A), which is lower than in common praxis. It justified this stipulation by citing the specific circumstance that the nearest housing area is a summer house area, defined as a quiet area. The Committee therefore argued that a higher loud level than 35 dB(A) was not suitable under the circumstances. The entrepreneur predicted this requirement and therefore reduced the number of wind turbines in the planned wind power station. This adjustment in the number of wind turbines lowered the expected loud level to 35 dB(A). The Committee therefore arrived at the judgement that regarding noise, no non-acceptable value loss would arise.

*Risk for bird mortality rate*: The entrepreneur identified an eagle's nest in the surroundings of the land-area specified in the application, and explicitly mentioned it in the EIA. The risk of eagles being killed by the wind power turbines depends, among other things, on the distance between the planned wind power station and the eagle's nest. The Committee stipulated the shortest distance between the wind power and eagle's nest as an indicator for a critical threshold value. If the nest was closer to the planned wind power station than the stipulated distance, a non-acceptable value loss would arise and the permit would be rejected. The Committee assessed that the distance between the nest and the planned wind power station would approximately be equal to the distance stipulated as a threshold level. The Committee therefore concluded that *no* non-acceptable value loss would arise concerning the risk for bird mortality rate.

*View of the landscape*: For this aspect, the Committee did not find it meaningful to stipulate any empirical criterion. It realized that the view of the landscape is a multidimensional aspect and it is therefore very cumbersome or even meaningless to stipulate an empirical threshold level. The Committee's discretionary judgement was that the landscape is rather sensitive for the intrusion of enterprises of an industry character. But the Committee assessed, despite the sensitivity of the landscape, that a wind power station would not have a sufficient negative impact on the landscape such that non-acceptable value losses arise.

*Reindeer husbandry*: The Committee judged that a wind power station may, to a certain extent, compete with reindeer husbandry, which would cause some economic losses for the reindeer owner. The entrepreneur agreed to compensate the reindeer owner for some of the economic losses, but it was not possible to compensate, according to the reindeer owner, for the loss of cultural values related to reindeer husbandry. The Committee assessed that the impact on the interest of reindeer husbandry would give rise to considerable value losses, but would not give rise to non-acceptable value losses.

The Committee´s conclusion, based on the application of critical threshold values related the four $\beta_i^-$-aspects, was that *no* non-acceptable value losses would arise. Further, the Committee did not find that the overall value loss would be below an overall critical threshold value. This means that the Committee must solve the value conflict by taking weighing decisions. The Committee therefore compared the value gain arising from the production of green electricity, denoted as:

$$\Delta_E(W, not{-}W),$$

with the overall value loss, which is an aggregation of the four partial value losses, denoted as:

$$\begin{aligned}\Delta_o(not\text{-}W, W) = \Delta_N(not\text{-}W, W) \oplus \Delta_L(not\text{-}W, W)\\ \oplus\, \Delta_B(not\text{-}W, W) \oplus \Delta_I(not\text{-}W, W).\end{aligned}$$

The granting of a permit requires that the Committee judges that $\Delta_E(W, not{-}W) \succ^* \Delta_o(not{-}W, W)$. However, the Committee decided, after a stepwise comparison of the value gain against the various value losses, that:

$$\Delta_o(not{-}W, W) \succ^* \Delta_E(W, not{-}W),$$

i.e., the expected overall value loss was assessed to be greater than the overall value gain.

The Committee justified its decision in the following manner: even if the value losses regarding each of the aspect did not give rise to non-acceptable value loss, the value loss was nevertheless considerable and outweighed the value gain. The Committee also underscored the fact that the wind power entrepreneur had to decrease the number of turbines in order to fulfil the threshold level for noise stipulated to 35 dB(A). This meant that the original value gain of the planned wind power station would decrease due to a decreased capacity. This decrease in capacity also contributed to the Committee's final decision that the use of the land-area was not suitable for wind power.

We conclude the story by adding that the entrepreneur objected to the decision by claiming that the Committee had not considered the impact of the wind power station on the aspect *local employment rate*. The entrepreneur claimed having presented strong evidence that local employment rate would increase at a wind power investment. And such an increase would be of high local value, because the local unemployment rate happened to be high. One reason for the Committee not to consider such an objection may be that it did not know, due to the lack of a formal conceptual framework, how to aggregate the various value gains into an overall value gain. Secondly, the Committee could not handle a complicated process in terms of weighing an aggregate of value gains against an aggregate of value losses. Such a weighing process could be supported by tools developed within MCDT.

## Endnotes

1. An extensive description of values which have to be considered at planning of wind power stations are in Planning and Review of Wind Power Plants ("Planering och prövning av vindkraftverk på land och i kustnära vattenområden") issued by the National Board of Housing, Building and Planning, 2009, Stockholm.

2. In Sweden, a permit is required from the County Board for Wind Power Stations with more than seven wind turbines higher than 120 ms or with at least one wind turbine higher than 120 m (see endnote 2).
3. Planning and Review of Wind Power Plants contains instructions for how to design an EIA (see endnote 2).
4. We ignore the possible outcome that alternatives W and not-W are overall equally good. This possibility is not, as far as we can see, treated in the Swedish Environmental Code.
5. It seems common in permission processes that the only $\alpha_i^+$- aspect which is included is that of the production of "green" electricity. However, the number of $\beta_i^-$- aspects related to value losses is usually more considerable than two.
6. The meaning and application of concatenation operations are explained in chapter 3 of Figueira (Figueira et al. 2005).
7. To implement models developed under MCDT in these permission processes requires applied research in terms of case studies where decision analysts have to cooperate with members of the Environmental Assessment Committee.

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# 9

# Attitudes Towards Paying for Environmental Protection in the Baltic Sea Region

**Sirje Pädam and Ranjula Bali Swain**

## Introduction

There are severe environmental threats to the Baltic Sea, and the cost of improving its ecosystem is high. One of the challenges of public environmental policies relates to the raising of funds for environmental protection. For the Baltic Sea, further challenges arise because the sea is a common pool resource to its nine littoral countries. Unless national policies are coordinated, measures undertaken by one country run the risk of being nullified by less stringent limits placed by other countries. Due to the income disparity across the Baltic region, it has been hypothesized that there is a substantial variation in the Willingness to pay for environmental protection. Other differences between countries

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include cultural background and line of policy, which may further influence the commitment towards environmental issues.

The country-level differences in priority accorded to environmental protection seem to be large, even among seemingly similar countries. In a cross-country comparison about the use of EU cohesion funding for the environment in the Baltic states, it was found that Estonia will devote a significantly larger share of its funding to environmental protection in comparison to Latvia and Lithuania, (Pädam et al. 2010). Similar differences between Estonia, on the one hand, and Latvia and Lithuania, on the other hand, have been reported elsewhere (Czajkowski et al. 2015; Ahtiainen et al. 2014). Recent research shows growing empirical evidence of substantial heterogeneity across countries in the Baltic region in terms of approaches towards environmental protection.

The purpose of this chapter is to examine the main determinants of public attitudes towards the willingness to pay for environmental protection in countries surrounding the Baltic Sea. Support to environmental protection is measured as the willingness of individuals to make financial sacrifices through higher prices and taxes or accepting cuts to one's standard of living to protect the environment. Furthermore, the chapter adds to existing literature by empirically analysing the determinants of general attitudes to environmental protection in Estonia and comparing it with its neighbouring countries around the Baltic Sea.

The analysis is conducted using data from the International Social Science Program (ISSP Research Group 2012) and a data set collected for Estonia in 2010, based on a selection of ISSP survey questions (Estonian Environmental Survey 2010). ISSP has organized a series of international attitude surveys on various issues covering more than 50 countries. The survey on the environment, which was carried out in 2010 (Environment III), includes questions about peoples' attitudes towards environmental protection in 33 countries. Seven of the nine littoral countries of the Baltic Sea, with the exception of Poland and Estonia, are included. Although the Estonian survey was conducted during the same time period as the third wave of ISSP Environment, the Estonian survey results are analysed for the first time and are compared to Environment III and the coastal countries of the Baltic Sea. Besides individual socio-economic variables of the respondents, the analysis covers country-specific data.

The next section reviews literature on the willingness to pay for environmental protection. The one subsequent to it describes the Estonian and ISSP data. Section "Comparing Environmental Protection Attitudes in Estonia and Other Baltic States" introduces the results of the Estonian survey in a cross-Baltic perspective, and in Section "Empirical Results", we present the empirical evidence and compare the Estonian attitudes to other countries in the Baltic region. The policy implications and differences between the countries with respect to Estonia are discussed in the concluding section of this chapter.

## Literary Review

In order to present a review of literature consisting of previous analyses of attitudes in terms of willingness to pay from a cross-country perspective, we begin by describing the concepts of attitudes and willingness to pay.

### Attitudes and Willingness to Pay

ISSP uses an attitude approach to the concept of willingness to pay in its survey. Respondents are asked to state their level of agreement or disagreement to statements about their willingness to pay to improve the environment. Willingness to pay is also used in contingent value (CV) surveys. In CV surveys, respondents are asked about their willingness to pay a specific amount of money for an explicit change in environmental quality. Clarity about the environmental quality change is important because each respondent needs to consider whether the financial sacrifice outweighs the improvement. This is because CV studies measure preferences for the environmental good in question.

Compared to CV surveys, the questions of ISSP are relatively non-specific. Rather than asking respondents about detailed aspects of environmental improvement, ISSP questions consider the environment as a whole. For this reason, the questions of the ISSP survey are too general to hold information about individual preferences. Instead, the aim is to examine people's attitudes. While preferences are based on an individual's ranking of alternatives constrained by his or her resources, attitudes express agreement or disagreement to alternatives. From this, it follows that a positive attitude towards an option does not necessarily imply that the individual prefers that alternative (Pouta 2003). Furthermore, by connecting the attitudes about environmental protection to personal sacrifices, the ISSP questions differ from attitude questions in general. Chaisty and Whitefield (2015) point out that the motivation behind this is to prevent respondents from merely paying lip service to environmental concerns.

## Surveys on the Baltic Sea Region

In the empirical analyses of attitude data and the CV surveys, several common socio-economic variables become focus areas of study. Most often, CV surveys find a positive relationship between the environmental good in question and income (see, e.g. Ahtiainen et al. 2014; Flores and Carson 1997; Hökby and Söderqvist 2003). In a handbook of the Swedish Environmental Protection Agency, a positive correlation between environmental good and individual income is specified as one criterion for assessing the quality of an environmental CV study (Naturvårdsverket 2005). Since most environmental goods are likely to be normal goods, adjusting for income is an important assumption when taking a value estimate from one context to another. Due to the high costs of conducting CV studies, there is often a need to bring existing value estimates to new contexts. The practice of benefit transfer, i.e. the transfer of willingness-to-pay estimates from one country to another, typically rely on adjustments for GDP per capita (see, for instance, Turner et al. 1999; Huhtala 2009).

The CV survey conducted by Ahtiainen et al. (2014) is interesting in this context. The authors implement a coordinated and identical CV study in all nine coastal countries of the Baltic Sea in 2011. The respondents are presented with two improvement scenarios with respect to the Baltic Sea in 2050 and asked about their willingness to pay for policies that reduce its eutrophication. The resulting country-level estimates on willingness to pay for nutrient reduction are compared to the benefit transfer carried out by Turner et al. (1999). In a study that dates from the 1990s, the authors used the estimates of a Swedish and Polish CV survey to transfer the willingness-to-pay estimates to the rest of the coastal countries of the Baltic Sea by adjusting for GDP per capita. Ahtiainen et al. (2014) note that the new estimates for willingness to pay are lower than those suggested by Turner et al. (1999). One reason is that the benefits of the scenarios are expected to appear in 40 years while the previous study from the 1990s expected benefits to be in place in 20 years. Another is that the approach of benefit transfer, i.e. GDP per capita adjustment, overestimates the benefits. This is the case for all countries, except Estonia. The average willingness to pay in Estonia is about two times higher than the transferred estimate.

Ahtiainen et al. (2014) analyse the data collected at the country level with interval regressions. Their results reveal that personal income is significant for all countries but Germany. Age is significant and negative in three out of nine countries. Higher education implies higher willingness to pay in Finland, Germany, Poland and Russia. Furthermore, personal experience of eutrophication and concern for the environment of the Baltic Sea have significant explanatory power for the willingness to pay in eight of the nine countries. Although income is found to be an important determinant, the authors conclude that other socio-economic variables, experience, knowledge and concern for the environment play a more significant role.

## Cross-Country Attitude Surveys

Empirical studies on cross-country comparison of attitudes make a wider set of sources available. There are three well-known, internationally coordinated surveys that allow cross-country comparison of environmental attitudes: ISSP, the World Value Study (WVS) and the European Values Study (EVS). Franzen and Vogl (2012) use all three sources to investigate previously reported opposite results: Some studies find positive significance between country-level wealth while others report higher environmental concern in poor countries. In these studies, environmental concern is expressed on a scale of agreement or non-agreement to the willingness to pay for environmental improvement. Based on multilevel regression analysis, the authors show that wealth is a positive determinant of environmental concern.[1] Franzen and Vogl (2012) also note that this result is much stronger at the individual level, as compared to the country level. In another study that uses the data set of ISSP, Franzen and Meyer (2010) find that variation in the agreement to pay higher prices for environmental protection is, to a large degree, explained by individual difference in wealth rather than country-level income: 85 versus 15%. Among the individual variables in the multilevel analysis, income, years of education, post-materialism, perceived environmental quality and being a woman significantly increase environmental concern. The notion of post-materialism originates from environmental sociology and is based on the proposition that environmental concern in society grows with prosperity (Inglehart 1995). When people have met their basic material needs for food, shelter and safety, they tend to shift focus to post-materialist quality-of-life issues from materialist issues. The influence of age on environmental concern is also significant, but negative. Similar results on the individual level have also been reported by authors who have analysed data from EVS and WVS (Gelissen 2007; Torgler and Garcia-Valinas 2007).

Gelissen (2007: 401) proposes a model with individual variables, supplemented stepwise by country-level variables. The dependent variable reflects agreement to the survey questions: "I would give part of my income if I were certain that the money would be used to prevent environmental pollution," and "I would agree to an increase in taxes if the extra money is used to prevent environmental pollution". Gelissen (2007) finds that countries with high levels of national wealth appear to be less willing to pay for environmental protection than those with lower levels of national wealth. At the same time, the results reveal that high economic growth and country-level post-material values are related to higher support for environmental protection. Gelissen (2007) does not find a significant correlation between willingness to pay and environmental quality variables (population density, air and water quality). The results at individual level reveal that environmental support is positively and directly related to income, post-materialism, educational level and environmental engagement, and negatively related to age.

Torgler and Garcia-Valinas (2007) employ Spanish data from three waves of WVS (1990, 1995 and 2000) and use the level of agreement to the survey question “I would agree to an increase in taxes if the extra money is used to prevent environmental damage,” as the dependent variable. “I don’t know” answers and missing values are omitted from the analysis. The authors use ordered probit estimation with and without weighting factors (for pooling purposes and to correct the sample in order to reflect national distribution). The results reveal that education, political interest, green ideology, financial satisfaction and social capital significantly increase willingness to pay higher taxes. Furthermore, the authors find that the higher the level of public expenditure on the environment (per $km^2$), the lower is the willingness to pay taxes for the environment in the Spanish regions.

Using multivariate analyses on ISSP data, Ivanova and Tranter (2008) show that the main determinants for expressing positive willingness to pay for the environment include education, value orientations, perception of environmental risk and concern over the implications of global warming. However, the authors do not include individual income in their regressions. There is no discussion about this omission, but in a separate regression, using data from the Australian Survey of Social Attitudes, they find that income is an insignificant determinant with respect to the willingness to pay higher taxes.

Pädam and Ehrlich (2011) compare the results of the Estonian Environmental Survey with the second wave of ISSP environment. There is no analysis of individual characteristics. The authors compare data at the country level and find a positive correlation between GDP per capita and the willingness to pay higher prices as well as acceptance of cuts in living standards. No correlation is found between GDP per capita and the willingness to pay higher taxes.

In a recent study, Chaisty and Whitefield (2015) compare three waves of the ISSP Environment (1993, 2000 and 2010). They find that at the individual level, environmental attitudes depend positively on education, being a left-wing voter, social class and expressing post-materialist values. The negative impact from age is not significant for the whole data set, only for a subset of post-communist countries. Another interesting result is that post-communist countries remain less supportive of environmentalism than advanced democracies. Chaisty and Whitefield (2015) suggest that it is likely that there is a stability of attitudes shaped during the communist regimes, and that these are possibly sustained by negative experiences of transition. An alternative explanation offered is that the connection between environmental issues with the political discourse is weak. So far, environmentalism has not emerged as a distinct dimension of political competition in post-communist countries.

## Data

In this chapter, we investigate the attitudes towards making sacrifices for environmental protection in the Baltic Region. Two data sets form the empirical basis of the analysis: ISSP (ISSP Research Group 2012) and the Estonian Environmental Survey (2010). The pooling of common questions in ISSP and the Estonian Environmental Survey makes it possible to account for attitudes towards the environmental protection of eight coastal countries of the Baltic Sea. We compare the attitudes of the aggregate of seven countries in the Baltic Region to those of Estonia. The attitudinal dependent variables covered by both data sets include willingness to accept cuts in living standards, willingness to pay higher taxes and willingness to pay higher prices in order to protect the environment.

### Data Sources and Details

ISSP is a continuing annual programme of cross-national collaboration on surveys covering topics important for social science research, including on environmental attitudes. The latest survey on the environment was carried out in 2010 and covers 32 countries, providing about 45,000 observations (ISSP Research group 2012). Of the nine littoral countries, neither Poland nor Estonia was included.

In another survey, a selection of ISSP survey questions were collected for Estonia. The Estonian Environmental Survey was conducted during December 2009 to February 2010. About 1200 respondents were contacted by interviewers in Tallinn and in rural areas of Estonia. Almost 850 filled-in questionnaires were returned, yielding a response rate of about 70%. Pädam and Ehrlich (2011) conducted the Estonian Environmental Survey with the intention of mapping the attitudes towards environmental protection and protected areas, to be presented at the celebrations of the 100th anniversary of nature protection in Estonia.

The empirical analysis covers the responses to the following three questions included in both the Estonian Environmental Survey and ISSP: "How willing would **you** be to **accept cuts** in your **standard of living** in order to protect the environment?", "How willing would **you** be to pay **much higher taxes** in order to protect the environment?" and "How willing would **you** be to pay **much higher prices** in order to protect the environment?" (Bold emphasis appears in the original survey questions). Responses to these questions are given on a six-point scale: "very willing", "fairly willing", "neither nor", "fairly unwilling", "very willing" and "don't know". Missing values and "I don't know" answers are omitted.

Although positive responses towards making sacrifices for the sake of the environment do not necessarily imply that individuals prefer this action, the analysis will still capture the determinants of positive attitudes towards the action. Based on earlier research, responses to attitude questions are consistent with those of willingness to pay (Pouta 2003), thus suggesting that the Estonian Environmental Survey and ISSP data could provide inputs for further implications. Additionally, the questions about hypothesized sacrifices in terms of payment and lower standard of living provide semblance to the CV questions.

## Data Adjustment for Comparability

To make comparisons between Estonia and other countries in the Baltic region, the data needed to be pooled. This created some additional challenges. The age variable of ISSP had to be coded according to the seven age categories in the Estonian Environmental Survey. The education variable of Estonia is similar but not the same as that of ISSP. Lack of formal education was not reported for Estonia. Secondary special category was classified as intermediate category, whereas the university incomplete category had to be mixed with the secondary education category for Estonia. University and postgraduate categories were collected separately for Estonia, but were merged together. Income was converted to euros using average annual currency exchange rates for 2010. Data on personal income differed within the sample. Some countries had collected data on gross income (Denmark, Finland, Latvia, Lithuania, and Sweden) while others did so on net income (Estonia, Germany and Russia). In addition, two countries (Denmark and Estonia) presented income in income categories rather than actual amounts. Converting net to gross income required some simplifying assumptions. Internet-based personal income tax calculators were used for Estonia and Germany to calculate gross incomes[2] in 2010. The general tax level was chosen. For Russia, a further simplification had to be made assuming that gross income would correspond to an upward adjustment by the 2010 flat tax of 13%.

In order to compare gross incomes, countries with progressive income taxation were organized according to the original nine Danish categories (Denmark, Finland, Germany and Sweden), while flat tax countries used the nine Estonian income categories (Estonia, Latvia, Lithuania, and Russia). For the purpose of cross-country comparisons, GDP per capita data based on purchasing-power-parity (PPP) from the International Monetary Fund were employed (IMF 2015).

## Comparing Environmental Protection Attitudes in Estonia and Other Baltic States

Table 9.1 presents the descriptive statistics of the variables used in the analyses for Estonia and the other littoral states in the Baltic region (except Poland). Compared to the other countries in the Baltic region,

**Table 9.1** Environmental attitudes in Estonia and Baltic region (Estonia and Poland excluded)

| | Baltic region | Estonia |
|---|---|---|
| Protect environment: lower your standard of living (%) | | |
| Very willing | 4.4 | 3.4 |
| Fairly willing | 26.6 | 15.3 |
| Neither willing nor unwilling | 25.5 | 22.2 |
| Fairly unwilling | 24.1 | 35.7 |
| Very unwilling | 19.5 | 23.3 |
| Protect environment: pay much higher taxes (%) | | |
| Very willing | 2.6 | 2.9 |
| Fairly willing | 18.0 | 21.4 |
| Neither willing nor unwilling | 23.5 | 18.5 |
| Fairly unwilling | 29.4 | 40.6 |
| Very unwilling | 26.5 | 16.5 |
| Protect environment: pay much higher prices (%) | | |
| Very willing | 3.3 | 4.2 |
| Fairly willing | 24.8 | 28.7 |
| Neither willing nor unwilling | 26.0 | 17.9 |
| Fairly unwilling | 26.0 | 36.2 |
| Very unwilling | 19.8 | 12.9 |
| Males (%) | 41.7 | 40.5 |
| GDP per capita 2010 (ppp USD) | 32,653 | 21,613 |
| Age category (in years) | | |
| 18–23 | 8.5 | 22.6 |
| 24–29 | 8.5 | 24.1 |
| 30–39 | 14.6 | 18.3 |
| 40–49 | 18.2 | 16.1 |
| 50–59 | 19.5 | 11.5 |
| 60–69 | 17.5 | 4.9 |
| 70 or more | 13.2 | 2.5 |
| Education degree | | |
| No formal | 1.2 | – |
| Lowest formal | 12.2 | 3.3 |
| Intermediate | 19.2 | 28.3 |
| Higher secondary | 29.8 | 22.2 |
| University incomplete | 17.6 | – |
| University complete | 20.0 | 46.2 |
| Income category (monthly gross income in euros) | | |
| 1. (0–131) and (0–1118) | 29.6 | 6.5 |
| 2. (132–377) and (1119–1678) | 24.6 | 17.9 |
| 3. (378–627) and (1679–2238) | 15.1 | 22.4 |
| 4. (628–876) and (2239–2796) | 10.4 | 22.0 |
| 5. (877–1125) and (2797–3357) | 6.4 | 15.3 |
| 6. (1126–1375) and (3358–4475) | 7.1 | 8.7 |

(continued)

**Table 9.1** (continued)

| | Baltic region | Estonia |
|---|---|---|
| 7. (1376–1625) and (4476–5595) | 3.4 | 4.0 |
| 8. (1626–1875) and (5596–6714) | 1.3 | 2.1 |
| 9. (1876–…) and (6715–…) | 2.2 | 1.2 |
| Total observations | 7280 | 756 |

*Source* Estonian Environmental Survey (2010) and Environment III (ISSP Research Group 2012)

Estonia seems to have a lower acceptance for cuts in the standard of living for environmental protection. The country-level data suggest that Estonia is similar to Latvia, Lithuania, and Russia in the willingness to accept cuts in the standard of living for environmental protection. On the other hand, Estonia's willingness to pay higher taxes and higher prices for environmental protection places it at a similar level as the other countries of the Baltic region (Denmark, Finland, Germany and Sweden). This implies that Estonia seems to lie on the dividing line between former planned economies and western countries.

On average, the GDP per capita of Estonia ($21,613) is much lower than the average of the other countries in the Baltic region ($32,653). In the sample, the Nordic countries and Germany have an average GDP per capita of about $40,000. The country-level GDP per capita of Estonia is about half of this and close to the GDP per capita of Lithuania,, Russia and Latvia. About 40–42% of the sample respondents were male. In this respect, there is a similar bias in Estonia and the other countries in the Baltic region.

The sample from Estonia is younger as compared to the other Baltic countries. Weighting Estonia's attitude questions to the age structure of the population has only a minor impact on the outcome. In the weighted sample, attitudes tend to become more positive, but the magnitude of this change is less than one percentage point.

The sample respondents in Estonia have a higher university level education, with 46.2% falling in this category, as compared to 20% in other Baltic countries. It is thus not surprising that the more educated Estonian sample may have a bias towards more positive attitudes towards paying higher taxes and prices for environmental protection.

According to Ahtiainen et al. (2014: 285), the largest shares of higher education in the population are found in Sweden, Estonia and Finland (33, 31 and 29%, respectively). The share of higher education in the population is about 25% in Denmark, Germany, Latvia, Lithuania, and Russia.

## Empirical Results

Public attitudes in terms of the willingness to accept cuts in the standard of living and the willingness to pay higher taxes and prices depend on several socio-economic factors. In the data, the response to questions determining these attitudes is a category variable, where the response was ordered as follows: very willing, fairly willing, neither, fairly unwilling and very unwilling. Thus, the ordered logistic regression is employed, and the marginal effects are estimated for all the reported categories (Greene 2012).

### Results on Attitudes Towards Environmental Protection

In this section, the marginal effects for the "very willing" and "fairly willing" categories are reported for Estonia and the other littoral states in the Baltic region. Table 9.2 presents the marginal effects of the ordered logit for the pooled "very willing" and "fairly willing" categories to accept cuts in their standard of living to protect the environment. For other countries in Baltic region, an increase in the GDP per capita leads to an increase in willingness to accept cuts in the standard of living. Similarly, university education increases the tendency to accept cuts in the standard of living to protect the environmental quality. The increase in income also implies a disposition towards greater acceptance to cuts in the standard of living for the environment. The willingness to accept cuts in the standard of living is lower among men in the Baltic Sea region. However, while university education is related to greater willingness to accept cuts in living standards for other Baltic countries, Estonia does not reflect this trend.

**Table 9.2** Ordered logit marginal effect for the respondents who stated they were very and fairly willing to accept cuts in their standard of living (standard errors in parenthesis)

| | Other countries in Baltic region | Estonia |
|---|---|---|
| GDP per capita | 1.3 e−6 (5.14 e−7)[c] | |
| Male | −0.022 (0.009)[b] | |
| Age category | | |
| 70 or more | −0.038 (0.019)[b] | 0.193 (0.096)[b] |
| Education degree | | |
| University incomplete | 0.074 (0.034)[b] | |
| University complete | 0.143 (0.035)[c] | |
| Income category | | |
| 3 (378–627) and (1119–1678) | 0.038 (0.014)[c] | |
| 4 (628–876) and (2239–2796) | 0.047 (0.017)[c] | |
| 5 (877–1125) and (2797–3357) | 0.053 (0.021)[b] | |
| 7 (1376–1625) and (4476–5595) | 0.050 (0.028)[a] | |
| 8 (1626–1875) and (5596–6714) | | 0.201 (0.118)[a] |
| 9 (1876–…) and (6715–…) | | |

*Source* Estonian Environmental Survey (2020) and ISSP Environment III (IISP Research Group 2012)
*Notes* Only significant results reported for age and income categories
[a], [b] and [c] represent the statistical significance at 10, 5 and 1% confidence interval, respectively

Older respondents (70 years and above) are especially sensitive to a decline in the standard of living for environmental protection. Contrary to other countries in the Baltic region, older respondents (70 years and above) in Estonia were more willing to accept cuts in the standard of living for the protection of the environment. Similar to other countries in the region, an increase in personal income (monthly gross income exceeding €1625) raises the willingness of Estonian respondents to accept cuts in the standard of living.

The estimates for the willingness to pay a much higher tax are presented in Table 9.3. An increase in the GDP per capita, income (in most income categories) and higher education implies a positive attitude towards paying higher taxes to protect the environment in the other Baltic region countries. Completion of university studies has a larger influence on the attitude to pay higher taxes as compared to the reference categories (lower education). Estonians within the lower

**Table 9.3** Ordered logit marginal effect for the respondents who stated they were very and fairly willing to pay much higher taxes (standard errors in parenthesis)

| | Other countries in Baltic region | Estonia |
|---|---|---|
| GDP per capita | 6.97 e-6 (3.82 e-7)[c] | |
| Age category | | |
| 30–39 | −0.033 (0.015)[b] | |
| 40–49 | −0.036 (0.015)[b] | |
| 50–59 | −0.030 (0.015)[b] | |
| 60–69 | −0.035 (0.015)[b] | |
| 70 or more | −0.052 (0.015)[c] | 0.186 (0.110)[a] |
| Education degree | | |
| University incomplete | 0.085 (0.024)[c] | |
| University complete | 0.120 (0.024)[c] | 0.142 (0.060)[b] |
| Income category | | |
| 2 (132–377) and (1119–1678) | 0.024 (0.024)[c] | −0.138 (0.063)[b] |
| 3 (378–627) and (1119–1678) | 0.038 (0.010)[c] | |
| 4 (628–876) and (2239–2796) | 0.046 (0.012)[c] | |
| 5 (877–1125) and (2797–3357) | 0.082 (0.017)[c] | |
| 6 (1126–1375) and (3358–4475) | 0.060 (0.016)[c] | |
| 7 (1376–1625) and (4476–5595) | 0.058 (0.212)[c] | |
| 8 (1626–1875) and (5596–6714) | 0.105 (0.038)[c] | |
| 9 (1876–…) and (6715–…) | | |

*Source* Estonian Environmental Survey (2010) and ISSP Environment III (ISSP Research Group 2012)
*Notes* Only significant results reported for age and income categories
[a, b] and [c] represent the statistical significance at 10, 5 and 1% confidence interval, respectively

income brackets (€132–1678 gross monthly income) do not display an attitude of willingness to pay higher taxes. University-educated Estonians tend to display greater willingness to pay higher taxes for the environment. This is also the case among older respondents (70 years and above) than younger ones.

Table 9.4 presents the marginal effects of the ordered logit for those who are “very willing” and “fairly willing” to pay much higher prices to protect the environment. For other countries in the Baltic region, an increase in the GDP per capita of a country leads to greater willingness to pay higher prices. Similarly, university education increases

**Table 9.4** Ordered logit marginal effects for the respondents who were very and fairly willing to pay much higher prices (standard errors in parentheses)

| | Other countries in Baltic region |
|---|---|
| GDP per capita | 1.1 e-6 (4.77 e-7)[c] |
| Age category | |
| 40–49 | −0.043 (0.018)[b] |
| 70 or more | −0.058 (0.018)[c] |
| Education degree | |
| University incomplete | 0.098 (0.024)[c] |
| University complete | 0.157 (0.030)[c] |
| Income category | |
| 2 (132–377) and (1119–1678) | 0.028 (0.010)[b] |
| 3 (378–627) and (1119–1678) | 0.051 (0.013)[c] |
| 4 (628–876) and (2239–2796) | 0.045 (0.015)[c] |
| 5 (877–1125) and (2797–3357) | 0.080 (0.020)[c] |
| 6 (1126–1375) and (3358–4475) | 0.076 (0.020)[c] |
| 7 (1376–1625) and (4476–5595) | 0.084 (0.027)[c] |
| 8 (1626–1875) and (5596–6714) | 0.136 (0.046)[c] |
| 9 (1876–…) and (6715–…) | 0.128 (0.036)[c] |

*Source* Estonian Environmental Survey (2010) and ISSP Environment III (ISSP Research Group 2012)
Notes: Only significant results reported for age and income categories
[a], [b] and [c] represent the statistical significance at 10, 5 and 1% confidence interval, respectively. None of the variables were significant for Estonia

the tendency to contribute through higher prices to protect the environmental quality. The Increase in income also implies a disposition towards greater contribution through paying higher prices for the environment. Middle-aged respondents (40 years and above) are, however, less willing to pay higher prices for environmental protection. Estonia is an outlier. Contrary to other countries in the Baltic region, no similar trend is observed in Estonia in terms of higher prices for the protection of the environment.

The trends of significant variables are, in most cases, in accordance with expectations and results reported elsewhere in the literature. It is interesting to note that the results of studies in other countries in the Baltic region reflect the findings of Franzen and Vogl (2012)—that of a stronger positive influence from personal income than from country-level wealth. Furthermore, several previous studies have reported age being a negative determinant of attitudes and preference to pay for environmental improvement. This tendency is present in other countries in the Baltic region and is most visible with regard to willingness to pay higher taxes to protect the environment. Estonia, on the other hand, does not reflect this.

Most previous studies have found that education significantly improves attitudes towards environmental protection. This is the case in the Baltic region as well. Higher education (including university incomplete) is a positive determinant for other countries in the Baltic region with respect to all three attitude questions. In Estonia, higher education significantly improves attitudes towards paying higher taxes, while higher education is not an important determinant for accepting cuts in living standards or the willingness to pay higher prices.

## Conclusions

This chapter is one of the first to compare public attitudes with respect to the willingness to pay for environmental protection in Estonia cross-nationally. Three questions covered by the Estonian Environmental Survey and by ISSP have been compared and further analysed by ordered logit regressions. The cross-Baltic comparison places Estonia in the middle position. Estonia seems to display lower acceptance to cuts in the standard of living for environmental protection in comparison to the average of the other countries in the Baltic Sea region. The country-level data suggest that Estonia is similar to Latvia, Lithuania, and Russia in terms of willingness to accept cuts in the standard of living for the purpose of environmental protection. On the other hand, Estonia's willingness to pay higher taxes and prices for environmental protection is higher than the average for the pooled set of other countries in the Baltic region, and at a similar level as that of the Nordic countries and Germany. The initial observations of this chapter about differences between Estonia, on the one hand, and Latvia and Lithuania, on the other hand, thus seem to find support when it comes to the willingness to pay higher taxes and higher prices.

This chapter also finds support for the hypothesis that demand for the environment tends to increase with income. This is true for both personal income and country-level income. Some differences can be detected between the public attitudes towards willingness to accept cuts in the standard of living and the willingness to pay higher taxes and prices. Attitudes concerning monetary sacrifices show a larger number of significant income categories than attitudes towards cuts in the living standards. It is also interesting to note that the results reflect earlier findings of a stronger positive influence from personal income than from country-level wealth. This, together with previous research, indicates that adjustments of GDP per capita do not perform well for the purposes of benefit transfer, which implies that greater attention should be paid to other variables when value estimates are brought from one context to another.

Higher education, in addition to personal income, is the other main determinant of support to environmental protection. Completion of university studies has a significant influence on the willingness to pay for environmental protection in the other countries in the Baltic region. In Estonia, higher education significantly improves attitudes towards paying higher taxes. These results suggest that there is support among the general public to pay higher taxes for the purpose of environmental protection.

## Notes

1. They take into account the difference triggered by the number of answering categories, the five-point scale of the ISSP and the four- point scale of the WVS and EVS. This is done by calculating scale-dependent agreement indices, which are used for correcting the scale-dependent difference.
2. The sources for conversion from net gross income are as follows: http://palk.crew.ee/ and http://www.parmentier.de/steuer/steuer.htm?wagetax.htm (accessed 26 April 2016).

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# 10

# Is International Cooperation in the Baltic Sea Drainage Basin Possible?

Tomasz Zylicz

## Introduction

Baltic countries can and do cooperate in several fields—economic, environmental, cultural and over a number of other issues. In particular, they have agreed to reduce the eutrophication of the sea they all share borders with. Discharges of nitrogen and phosphorus did decline after the signing of the Helsinki Convention in 1992, but the pace of this progress has been much too slow, according to many environmentalists. The geographical distribution of costs and benefits calls for concerted action from all countries in the drainage basin and adequate participation in abatement expenditures.

This chapter outlines hypothetical cooperation programmes necessary in order to reduce by half the inflow of nitrogen and phosphorus—nutrients identified as crucial to restore the sea to decent environmental

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conditions. In order to be cost-effective, an abatement programme requires the implementation of measures that are as inexpensive as possible, given the overall discharge reduction level. For this, it is important that some countries set more ambitious targets than justified by their domestic priorities. In contrast, other countries will find it profitable to contribute financially to activities undertaken elsewhere rather than at home.

However, the last several decades demonstrate that neither group is ready to participate in such a cost-effective allocation of abatement efforts. Countries, where inexpensive methods are available, are reluctant to go beyond what is justified by their domestic considerations. The others hesitate to pay for activities that are located abroad. Consequently, the pace of the improvement process is slower than is justified by comparing region-wide benefits with region-wide costs. This chapter addresses the question of what are the prerequisites to invigorate this development.

It is important to first recognize that international cooperation in the Baltic Sea drainage basin is possible. Indeed, countries surrounding the Baltic Sea have been involved in many cooperative projects. Nevertheless, sceptics claim that the level of cooperation has failed to restore the sea to a decent environmental quality characteristic in the first half of the twentieth century. In particular, countries have failed to curb the eutrophication caused by the excessive inflow of nitrogen and phosphorus. Strictly speaking, neither the former nor the latter are contaminants. Both nitrogen and phosphorus are essential components of fertilizers applied in agriculture. The problem starts when their concentration exceeds what an ecosystem can absorb.

## The Predicament

According to assessments carried out at the turn of the 1980s and the 1990s, the annual discharge of nitrogen was around 900,000 tonnes, and that of phosphorus was 40,000 tonnes. Both have moderate downward trends now. The 2008 estimates (HELCOM 2011) are 700,000 tonnes for nitrogen and 30,000 tonnes for phosphorus.

Nitrogen is considered the limiting factor in the Baltic Sea ecosystem. In other words, phosphorus is abundant enough that its further growth will not lead to higher biological production, while its decline will not lead to lower production. Production is determined by the availability of nitrogen. The eutrophication of the Baltic Sea has been caused by the excessive inflow of nitrogen and phosphorus from land. Hence, if the sea is to be protected, land-based activities need to be controlled.

Efforts to protect the Baltic Sea have a long history. In 1973, the Gdansk Convention on protecting living resources in the Baltic Sea was signed. The International Baltic Sea Fishery Commission was established in Warsaw to set fishing quotas. However, it soon became apparent that the condition of the sea depends on land activities rather than harvests. Consequently, in 1974, the Helsinki Convention on protecting the marine environment of the Baltic Sea was signed. A completely amended agreement was signed in 1992, which is sometimes called the New Helsinki Convention. Contrary to the Gdansk Convention, the Helsinki Convention addresses land-based activities that are crucial for the protection of the Baltic Sea.

In economists' jargon, Baltic Sea protection is a public good. Such goods are characterized by the principles of non-exclusion and non-rivalry. The latter implies that socially justified supply of such goods is higher than the quantity resulting from the preference of any individual user. At the same time—because of the former—no one volunteers to finance the cost of supply, knowing that nobody can be excluded from the set of beneficiaries (once the supply is financed by somebody else). Hence, everybody would profit from the protection of the Baltic Sea but nobody is keen on financing it, since perhaps somebody else will pay anyway. As a result, signatories of the Helsinki Convention keep contemplating ever-improved protection programmes but their actual spending on such programmes is less than what is necessary to effectively reduce eutrophication.

The problem is further complicated by the fact that the abatement of nitrogen and phosphorus (along their way to the sea) also has local benefits, but domestic priorities are not necessarily aligned with wider regional ones. For instance, from the Polish domestic point of view, the most urgent concern is the abatement of sewage discharged into tributaries of the upper Vistula. For example, protecting the Dunajec river (a tributary to the upper Vistula) is certainly beneficial for the Baltic Sea but its contribution is relatively small. At the same time, from the point of view of providing people with safe potable water and guarding local ecosystems, the protection of the Dunajec river is of paramount importance. Consequently, if the Polish government were sensitive to local benefits, it would support the building of sewage treatment plants in the southern parts of the country. Yet the nutrients discharged in upstream regions are partially retained along their way to the sea. Out of, say, 12 tonnes of nitrogen discharged into the Dunajec river, only 4 tonnes will reach the Baltic Sea. If the discharge were reduced by 75%, only 1 tonne would reach the sea. In other words, while the total discharge is reduced by 9 tonnes, the Baltic Sea receives only 3 tonnes less. If a similar exercise were to be carried out for the lower Vistula, say, for the city of Tczew, where almost the entire load reaches the sea, the same abatement effort (discharge of 3 tonnes instead of 12 tonnes) lowers the inflow into the Baltic Sea by 9 tonnes. Thus, from the point of view of protecting the Baltic Sea, the most effective sewage treatment plants are those located in northern, rather than southern, Poland. (Zylicz 2003 provides a theoretical framework for this problem.)

All signatories to the Helsinki Convention are aware of this. All of them protect water ecosystems, but they do it to reap domestic benefits. The Baltic Sea attracts a lot of attention, but most activities are undertaken in order to protect inland ecosystems rather than the sea.

There is a large variety of abatement measures available. From the point of view of reducing eutrophication, it is certainly effective to equip housing districts with adequate sewage infrastructure. Yet it is also beneficial to limit car traffic, which is responsible for part of the nitrogen compounds reaching the sea. Should a decreased inflow of phosphorus be caused by lower consumption of detergents, or lower application of fertilizers in agriculture? Is eutrophication to be reduced by relying on highly efficient tertiary installations instead of cheaper secondary ones, or by restoring coastal wetlands which serve as “nutrient traps”?

These sorts of problems are analysed by economists who study the costs of reaching a given target—in this case, the reduced eutrophication of the Baltic Sea. They start with the cheapest option. In this context, that would be to restore the coastal wetlands that were once drained. But the amount of nitrogen and phosphorus that can be abated by this method is small. Therefore, they also take the next abatement method. Let us assume that this is reduced fertilization in agriculture. Applying this method is not without a cost, as it implies lower crop yields, i.e. a loss that needs to be compensated in some way. Alternatively, it can be reached by investing in better manure storage. But even if all farmers were to switch to better technology, the eutrophication problem of the Baltic Sea would not be solved. This leads us to the next abatement method, say, secondary treatment, followed by tertiary treatment, and so on, until the expected degree of reduced eutrophication has been reached.

It is evident that the cost of protecting the Baltic Sea depends on the level of ambition. A moderate programme to reduce eutrophication is cheap. Further improvement requires growing unit costs. The framework of the Helsinki Convention envisions a 50% reduction in the inflow of nitrogen and phosphorus. The annual cost of such a programme for nitrogen was estimated at $4 billion in the 1990s. Of course, it could be higher, if unnecessarily expensive measures are implemented—for instance, reduced fertilizer application in agriculture before restoring coastal wetlands that deliver the same result at a lower cost; or applying tertiary treatment, even though secondary treatment would be sufficient in certain cases, and so on. Economists say that reaching the 50% eutrophication target requires $4 billion per year if a cost-effective programme is implemented (otherwise, it could be more expensive). An early assessment of these costs was carried out by Gren et al. (1995) (see Table 10.1).

Similar assessments were repeatedly carried out later, and most recently by Wulff et al. (2014) and Hasler et al. (2014). These are higher in nominal terms—more than €4 billion in 2011—but given the

**Table 10.1** Cost-effective abatement of nitrogen in the Baltic drainage basin
*Source* Gren et al. (1995)

| Reduction rate (%) | 5 | 10 | 15 | 20 | 25 | 30 | 35 | 40 | 45 | 50 |
|---|---|---|---|---|---|---|---|---|---|---|
| Cost ($10^9$ $/year) | <0.1 | 0.1 | 0.1 | 0.2 | 0.3 | 0.5 | 0.8 | 1.2 | 1.9 | 4.1 |

time that has elapsed since the earlier estimations, they are somewhat lower in real terms (e.g. because the baseline loads are lower than those observed in the 1990s (Ahlvik et al. 2014); some progress has already been achieved).

A team based in the University of Warsaw made a pioneering attempt to estimate the benefits from reduced eutrophication. Coordinated research programmes were carried out in Poland, Sweden and Lithuania, in the mid-1990s. The Polish programme was the most extensive. This allowed for the consistent interpretation of Swedish and Lithuanian data that would otherwise be difficult to compare. As a result of this research, it was estimated that the annual benefit from the 50% reduction in the eutrophication of the Baltic Sea is $56 for an average adult Pole, $28 for a Lithuanian and $229 for a Swede (Markowska and Zylicz 1999). Following a complicated extrapolation procedure, these average figures allowed researchers to estimate the total benefit from the 50% abatement programme at $6 billion per year. Newer and more precise estimates are somewhat lower, but difficult to compare, as they are based on different programmes (Ahtiainen et al. 2014).

Estimated costs and benefits of a cleaner Baltic Sea suggest that the programme contemplated by environmentalists is economically justified. In other words, when the entire drainage basin is taken into account, the benefit from the 50% reduction of nutrient discharges is higher than its cost. But costs and benefits need to be compared at the drainage basin level. If balances were to be struck at the level of individual countries, the 50% target would turn out to be too ambitious. The cost is the same as accounted for in the drainage-basin-wide case, but the benefit from improving the quality of local ecosystems is not attractive enough. Some of the benefits are external in the sense that they accrue to other countries.

## Chander–Tulkens Model of Cooperation

While Baltic countries have made efforts to protect the sea, the scale of their activities has been geared to serve domestic priorities. The Swedes do whatever they consider justified and complain that the Poles do too little. The Poles also do whatever they feel is required and declare that they could have done more if other beneficiaries participated in the expenditure. There seems to be no way out of this situation: There will be no progress beyond what everybody does according to their individual priorities, unless a mechanism is put in place to finance the additional efforts required. Such efforts may prove necessary in order to protect the sea as a whole (as a public good), but they are not justified from the point of view of individual countries.

A specific mechanism for financing a public good was first suggested long ago, but it was formalized as late as the turn of the 1980s and 1990s. It is now called the Chander–Tulkens (1992) model, named after the economists who developed it using very sophisticated game theory techniques. Based on their theoretical model, one can derive the following equation to determine money transfers between the users of a common public good:

$$T_i = \gamma_i p_i - (\pi_i : \pi_N) \cdot \Sigma_j \gamma_j p_j,$$

where $T_i$ is money transfer to country $i$, $\gamma_i$ is marginal abatement cost in country $i$, $p_i$ is pollution abatement in country $i$, $\pi_i$ are benefits in country $i$ from the region-wide abatement and $\pi_N$ is the sum of benefits from the region-wide abatement ($\pi_N = \Sigma_j \pi_j$).

Despite the complicated method involved in deriving this formula, its intuitive interpretation is quite simple. Every country gets its abatement cost financed ($\gamma_i p_i$) and, at the same time, contributes to the total regional abatement cost ($\Sigma_j \gamma_j p_j$) in proportion to its share in the total benefit ($\pi_i$:$\pi_N$). A negative amount means that a country must pay rather than receive money. It can be easily demonstrated that the sum of transfers is zero ($\Sigma_j T_j = 0$). This mechanism allows for the financing of the economically justified supply of a public good, while beneficiaries participate in this endeavour in proportion to the benefits enjoyed.

For the Baltic Sea clean-up programme—understood here, for simplicity, with the 50% nitrogen abatement—$\pi_N$ is estimated at \$6 billion, and $\Sigma_j \gamma_j p_j$ is estimated at \$4 billion. The Chander–Tulkens model makes it possible to calculate the amount of hypothetical money transfers that would allow countries to reach such a state. Table 10.2 was compiled at the University of Warsaw.

**Table 10.2** Hypothetical Baltic transfers *Source* Markowska and Zylicz (1999)

| Country ($i$) | $\pi_i$:$\pi_N$ (%) | $T_i$ ($10^6$ \$) |
|---|---|---|
| Finland | 14.4 | −216.9 |
| Sweden | 26.7 | −395.6 |
| Denmark | 16.5 | −292.3 |
| Germany | 11.2 | 67.2 |
| Poland | 24.1 | 280.8 |
| Lithuania | 1.2 | 280.0 |
| Latvia | 0.8 | 208.8 |
| Estonia | 0.6 | 177.2 |
| Russia | 4.6 | −109.2 |
| Total | 100.0 | 0.0 |

As could have been anticipated, Sweden emerges as the largest net payer and Poland as the largest net beneficiary. Rather unexpectedly, Germany turns out to be financed in net terms, while Russia turns out to be a net payer. The paradox can be easily explained by the benefit extrapolation method applied. The benefits are assumed to be proportional to the number of people living in the Baltic drainage basin (in Poland, it is virtually the entire population). The benefits to Germany have been extrapolated from Swedish data, but the total is small given that only 3 million adult Germans live in the Baltic drainage basin. In particular, the inhabitants of Berlin are excluded from the calculation (as they live in the North Sea drainage basin), even though they certainly stand to benefit from the Baltic clean-up programme. Thus, if the benefits to Germany were to be calculated more accurately, they would probably turn out to be higher than the 11.2% number in the table 10.2. For Russia, the benefits have been extrapolated from Lithuanian data, which could have resulted in overestimation (the Russian share in total benefits is probably lower than 4.6%). Lithuanians are proud to be a coastal nation; they seem to appreciate the sea and, on average, presumably appreciate it more than Russians.

The hypothetical transfers described in the table are disputable and can certainly be corrected once more accurate information on the benefits from reduced eutrophication of the Baltic Sea becomes available. The order, however, is remarkable. The benefits are likely to be over \$1 billion per year. This is many times more than the amount Nordic countries allocate as environmental assistance for their eastern and southern neighbours, to be spent on water protection. Therefore, the largest net beneficiaries finance the reduced eutrophication of the Baltic Sea to a much lower extent than required in order to achieve an efficient scale. On the other hand, countries such as Poland, which could have done more, keep acting in accordance with domestic considerations. In order to abate nitrogen and phosphorus, and bring them to the levels justified by benefits accruing to all Baltic countries, a regional transfer mechanism needs to be established. At present, it seems that the signatories of the Helsinki Convention are not ready to take such a step.

One reason that countries hesitate to establish a transfer mechanism is uncertainty as to their share in the total benefits. While the average willingness to pay (WTP) for a given country is relatively easy to compute, a convincing extrapolation poses a challenge. The method adopted in calculations summarized in the table above takes into account the population living in the Baltic Sea drainage basin. Two littoral countries, Germany and Russia, have very small areas included in the drainage basin, which leads to uncertainty as to how to account for their citizens who live in other drainage basins. In the case of Germany, extrapolation limited to those who live in the Baltic drainage basin has probably led to massive underestimation. On the other hand, the approach adopted by Ahtiainen et al. (2014)—where the entire German population was considered—is probably also inappropriate. Their study assumed that the populations of littoral states are homogeneous with respect to their WTP for the Baltic clean-up programme. Thus, the German average WTP was multiplied by the entire German population. Table 10.3 summarizes the transfers implied by benefit shares based on Ahtiainen et al. (2014).

The transfers implied by this breakdown of benefits are completely different from the earlier pattern. First of all, the total sum of transfers is more than twice as high, which reflects the fact that the distribution of abatement costs and benefits is entirely different. German abatement costs are very low since a very small area falls in the drainage basin. At the same time, the benefits are substantial since the German population is much higher than that of any other Baltic country, except for Russia, which is assumed to be another large beneficiary (and net payer). Ahtiainen et al. (2014) take into account the population of Western

**Table 10.3** Alternative hypothetical Baltic transfers *Source* Author's calculations based on Ahtiainen et al. (2014)

| Country ($i$) | $\pi_i$:$\pi_N$ (%) | $T_i$ ($10^6$ $) |
|---|---|---|
| Finland | 4.2 | 206 |
| Sweden | 15.9 | 52 |
| Denmark | 3.5 | 245 |
| Germany | 47.3 | −1428 |
| Poland | 8.3 | 936 |
| Lithuania | 0.6 | 305 |
| Latvia | 0.3 | 228 |
| Estonia | 0.7 | 171 |
| Russia | 19.2 | −715 |
| Total | 100.0 | 0 |

Russia only, which nevertheless dominates over the rest. Table 10.3 makes Germany and Russia net payers, and every other country net beneficiaries.

The idea of including entire countries instead of merely the parts included in the drainage basin is sound; hypothetical agreements are to be signed by states rather than smaller territorial units. On the other hand, it is questionable whether extrapolation of the computed WTP onto the entire German population is justified. Extrapolation of the Russian WTP is also debatable. It is obvious that many Russians are aware of the Baltic Sea and do care about its environmental predicament. Yet it is doubtful whether they would agree to participate in a transfer mechanism leading to their subsidization of Baltic clean-up efforts in Finland, Sweden and Denmark.

## Policy Conclusions

Exercises based on alternative models of distributing benefits from a less eutrophicated Baltic Sea demonstrate that hypothetical transfers are difficult to ascertain. They show that the wider the gap between distribution of benefits and distribution of abatement efforts, the more extensive will be the transfers that are called for. If both were distributed in the same manner, there would be no need for transfers. Another lesson learnt is that transfers are sensitive to countries' WTP for the clean-up of the Baltic Sea. Even a small change in benefit assessment may result in a substantive change in hypothetical payments and in a country moving from the net payer to the net beneficiary group, or vice versa. While computing individual WTP for the Baltic Sea clean-up seems to be a relatively easy task, extrapolating its results onto wider populations poses a substantial challenge.

Hypothetical transfers based on the Chander–Tulkens model reflect side payments required to trigger cooperation between Baltic countries necessary to escape from the trap they are stuck in when they ignore the public good nature of the sea. They are rough estimates of what may provide incentives for cooperation, and do not have to be understood as direct payments.

One institutional design which may be contemplated is nutrient trading. As is widely known in environmental economics, tradable permits allow for the cost-effective allocation of an abatement effort. In other words, if the sum of permits is equal to what is agreed upon as the permissible level of discharges, their tradability will lead to minimizing the total cost of reaching this level. Cost-effectiveness, however, is independent of burden sharing. In particular, if the permits initially allocated replicate the cost-effective apportionment of abatement targets (so that no trading will occur), there are no transfers. If a country receives more permits than implied by these targets, it will be a net beneficiary of trading; otherwise, it will be a net payer. Consequently, by manipulating the initial allocation of permits, one can arrive at the transfer pattern identified by the Chander–Tulkens model.

An alternative way to implement the transfer mechanism is development assistance as practised by the Nordic countries. Within the framework of regional environmental aid, these countries have been involved in bilateral programmes to invest in abatement capacity in the Southern Baltic states. As mentioned above, the scale of these activities is at least one order of magnitude lower than the flows envisaged in the Chander–Tulkens model. In order to become an incentive, they should be increased many times, but they already form the required mechanism.

Yet another way to implement transfers—at least with respect to EU countries—could involve making use of the European structural and cohesion funds. Two problems need to be sorted out. One is the statutory function of these funds. They are meant to reduce development disparities among regions and member states. At the same time, the purpose of Baltic cooperation is different, even though, to some extent, undertaking abatement projects can be seen as a development objective. The second problem is "additionality", both on the collection and the disbursement side. European fund revenues are built through contributions made by member states, which are motivated by certain needs articulated by the European Commission. Planned disbursements are carefully assessed by "cohesion" countries that negotiate their totals. Hence, if the Chander–Tulkens transfers were to be channelled by European funds, the additionality of their budgets has to be checked.

Finally, it should be noted that the idea of side payments is inconsistent with the "polluter pays" principle. Some environmentalists object to this on these grounds and expect that all countries undertake whatever abatement activities are justified by supranational considerations. While consistent with some theoretical deliberations, the principle is not followed in international agreements, which often follow the "victim pays" principle. Are Baltic countries ready to sacrifice ideological rules for a more pragmatic approach?

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