Baltic-wide and Swedish Nutrient Reduction Targets

An Evaluation of Cost-effective Strategies

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Preface

The condition of the Baltic Sea has deteriorated over a long period. Frequently we receive reports about extensive and potentially toxic algal blooms, widespread dead sea-beds and depletion of fish stocks. A major force behind this development is the eutrophication of the Baltic Sea. Despite substantial reductions of the discharges of phosphorous and nitrogen during the last two decades (albeit smaller than those stipulated in international agreements) the situation of the Baltic Sea is still severe. In 2007, the countries surrounding the Baltic Sea adopted the so-called Baltic Sea Action Plan (BSAP) which sets up ambitious reduction targets regarding the yearly input of nutrients. Whether or not these targets actually will be met depends among other things on the costs of doing so.

In this report to the Expert Group on Environmental Studies, fil. dr. Katarina Elofsson, Swedish University of Agricultural Sciences, discusses how the costs of attaining the objectives of the BSAP may be reduced by nutrient-input permits trading and assesses the potential cost savings of such trade amongst the Parties of the BSAP. She also discusses how the Swedish domestic policy may be improved upon. It is the Expert Groups hope that the report will contribute to the policy process.

The author is solely responsible for the content, the analysis and conclusions presented in the report.

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Sammanfattning

Genom Baltic Sea Action Plan (BSAP) har länderna runt Östersjön nyligen kommit överens om en ny uppsättning gemensamma mål för minskade kväve- och fosforutsläpp. Det främsta skälet till att man ändrar målen är att det idag finns bättre kunskap om sambandet mellan kväve och fosfor å ena sidan och siktdjup å den andra. Siktdjupet anses vara en god indikator på övergödning.

Det finns två uppsättningar mål i BSAP:

1. *Bassängmål* som definierar vilka utsläppsminskningar som krävs för var och en av Östersjöns sju bassänger och
2. *Mål för avrinningsområden* som anger vilka utsläppsminskningar som krävs *i varje land* för var och en av de sju bassängerna.

Bassängmålen har beräknats så att en viss given förbättring av siktdjupet ska uppnås. De utsläppsminskningar som krävs har fördelats mellan länderna som omger respektive bassäng i form av mål för avrinningsområden. När länderna uppfyller målen för avrinningsområdena medför detta alltså också att man når bassängmålen. I princip skulle emellertid bassängmålen kunna uppnås genom andra sätt att fördela kraven på belastningsminskningar mellan länderna. BSAP-målen för kväve och fosfor har ännu inte antagits av Sveriges Riksdag. Istället bygger de gällande svenska kväve-och fosformålen på en äldre internationell överenskommelse som gjordes på 1980-talet.

I den här rapporten diskuteras:

a) Hur de nya internationella målen kan nås till lägsta kostnad, exempelvis genom att man tillåter utsläppshandel mellan Östersjöländerna och
b) Kostnadseffektivitet i svensk miljöpolitik för minskade kväve- och fosforutsläpp.

En ekonomisk modell över Östersjöregionen används för att analysera kostnaderna för att minska närsaltsutsläppen, dvs. kväve- och fosforutsläppen, till Östersjön under olika tänkbara miljöpolitiska scenarier. Modellen innehåller information om kostnader för och effekter av ett stort antal utsläppsminskande åtgärder i alla länder runt Östersjön. Analysen visar att:

- *Den potentiella vinsten av bassängvis handel med utsläppsrätter för kväve och fosfor är stor. Väl fungerande handel med utsläppsrätter beräknas kunna minska ländernas samlade årliga kostnad för att nå BSAP-målen med 16 procent, vilket motsvarar 724 miljoner €.*

Skälet till att man kan uppnå denna besparing är att målen för avrinningsområdena medför geografiska restriktioner för var åtgärder får vidtas. Därigenom kan man inte tillvarata alla möjligheter att vidta åtgärder till låg kostnad. Handel med utsläppsrätter mellan regeringar kan lösa detta problem och samtidigt minska kostnaderna för alla länder.

- *Utveckling av en samarbetsmodell liknande Clean Development Mechanism, som tillämpas under Kyotoprotokollet, kan bidra till att ytterligare minska kostnaderna för att nå kväve- och fosformålen för Östersjön.*

En nackdel med BSAP-överenskommelsen är att den inte inkluderar alla länder som bidrar till övergödningen av Östersjön. Vitryssland och Ukraina bidrar båda med betydande närsaltsutsläpp till Östersjön men har inget åtagande genom BSAP.

Clean Development Mechanism är en samarbetsform som definierats i Kyotoprotokollet och som möjliggör bilateralt samarbete mellan länder som har kvantitativa åtaganden och länder som inte har detta. Den ger incitament att genomföra lågkostnadsåtgärder i länder utan utsläppsåtagande istället för dyrare åtgärder i länder med åtagande. Ett liknande system skulle kunna tillämpas för Vitryssland och Ukraina.

- *Ett kostnadseffektivt åtgärdsprogram för att nå BSAP-målen kan innebära att siktdjupet ökar mer än vad länderna har som mål.*

Bassängmålen kräver omfattande minskningar av fosforbelastningen. I många fall måste fosformålen uppnås med åtgärder som samtidigt minskar både kväve- och fosforutsläpp, vilket medför att kvävebelastningen minskar mer än vad BSAP-länderna har som mål. Därmed minskar också siktdjupet mer än vad länderna gemensamt har satt upp som mål. Detta gäller i synnerhet Egentliga Östersjön. Den extra ökning av siktdjupet som kan följa på de nuvarande BSAP-målen kan vara förknippad med en liten ökning av nyttan men betydande kostnader. I dessa fall kan kostnaderna minskas genom en anpassning av utsläppsmålen samtidigt som de uppsatta målen för siktdjup fortfarande uppnås. En revidering av utsläppsmålen som tar hänsyn till ovanstående måste baseras på modeller där man både tar hänsyn till kostnader och kväve- och fosforutsläppens effekt på siktdjupet.

- *De svenska kväve- och fosformålen kan nås till lägre kostnad.*

En analys av den svenska politiken för minskade kväve- och fosforutsläpp indikerar att ett uppfyllande av BSAPs tillrinningsmål kommer att kosta ungefär dubbelt så mycket som de nuvarande svenska målen, om målen nås till lägsta kostnad. Detta beror huvudsakligen på de geografiska restriktioner som BSAP-målen medför när det gäller var utsläppen ska minskas. Exempelvis kommer BSAPs krav på att koncentrera svenska fosforåtgärder till avrinningsområdet till Egentliga Östersjön innebära att dyrare åtgärder i Egentliga Östersjöns avrinningsområde måste ersätta mindre dyra åtgärder i övriga avrinningsområden. Detta ger skäl att fråga sig huruvida man kan åstadkomma samma fosforminskning till Egentliga Östersjön till lägre kostnad genom att tillåta att man räknar in fosforåtgärder i andra avrinningsområden, samtidigt som hänsyn tas till att effekten av dessa åtgärder på Egentliga Östersjön kan vara lägre.

Analysen som presenteras nedan visar dessutom att ytterligare åtgärder för att minska kväveoxidutsläppen från energisektorn och, i de flesta delar av landet, ytterligare konstruktion av så kallade skyddszoner i lantbruket samt fortsatta ansträngningar att minska utsläppen från enskilda hushåll som inte är anslutna till vattenreningsverk inte är kostnadseffektiva åtgärder satt i relation till den nuvarande kväve- och fosforpolitiken. Dessa åtgärder kan bara motiveras om de är förknippade med tillräckligt stora positiva sidoeffekter på exempelvis luftkvalitet, biodiversitet och lokal vattenkvalitet. Vidare indikerar resultaten att relativt stora minskningar av fosforutsläppen kan åstadkommas utan ökade kostnader för miljöpolitiken genom att prioriteringen mellan kväve- och fosforåtgärder ändras. Effekten av detta är en förhållandevis begränsad minskning av kväveutsläppen. En fortsatt debatt kring det relativa värdet av att minska kväve- respektive fosforutsläpp förefaller därför angelägen.

Executive summary

In the so-called Baltic Sea Action Plan (BSAP), the governments around the Baltic Sea have recently agreed on a new set of targets for nutrient load reductions. The major motive for this is new and better knowledge about the link between nutrient loads and water transparency. Water transparency is considered to be a good indicator of eutrophication, i.e. over-enrichment of the sea.

There are two sets of BSAP targets:

1. *Basin targets* that define required reductions of nutrient loads to each of the seven basins in the Baltic Sea, and
2. Catchment targets which define required reductions in nutrient loads *from each country* to each of the seven basins.

The basin targets are chosen in order to meet a given improvement in terms of water transparency. Through the catchment targets, the necessary reductions are distributed amongst the countries' catchments. A fulfillment of these commitments on catchment basis thus means that the basin targets will be met. It should, however, be noted that the basin targets also can be met by other load abatement allocations. The BSAP targets have not yet been adopted by the Swedish Parliament. Instead, the official Swedish nutrient targets are based on older international targets agreed upon in the 1980's.

This report discusses:

a) how the new international targets could be met at least cost, e.g. through allowing emission permit trade – or rather load permit trade – amongst the BSAP-countries and
b) cost-effectiveness of the Swedish domestic nutrient policy.

An economic model over the Baltic Sea region is used to assess the costs of reducing nutrient loads to the Baltic Sea under different policies. The model comprises costs and effects of a large number of nutrient abatement measures in all countries. The analysis shows that:

- *The potential gains from basin-wide nutrient load trading are large. Well-functioning load permit trade is here estimated to reduce the total annual cost of meeting the BSAP's basing targets by 16 percent or by 724 millions €.*

The reason for these savings is that the catchment targets imply restrictions on the location of abatement and therefore, it will not be possible to take advantage of all low-cost abatement options. Nutrient load trading between countries at an intergovernmental level can solve this problem. Such trade among countries can reduce the cost for all countries

- *The development of a mechanism similar to the so-called Clean Development Mechanism under the Kyoto protocol could help to further reduce costs of meeting BSAP's nutrient load targets for the Baltic Sea.*

A drawback of the BSAP is that it does not cover all countries contributing to the eutrophication of the Baltic Sea. Belarus and the Ukraine contribute with significant amount of nutrients but do not have any commitment under the BSAP.

The Kyoto Protocol includes a flexible mechanism, the Clean Development Mechanism, which permits bilateral co-operation between countries with quantitative obligations and countries with no such commitments. The mechanism provides incentives for both types of countries to undertake low-cost abatement projects in countries with no commitment instead of more expensive ones in countries with a commitment. A similar system could be applied for Belarus and Ukraine.

- *Cost-effective fulfillment of BSAP's load targets can imply that water transparency is improved beyond the target levels*

The basin targets require substantial reductions of phosphorus loads. In many cases these targets are met by measures that simultaneously reduce both phosphorus and nitrogen, leading to an outcome where nitrogen loads are reduced beyond the targeted level. Hence, water transparency will be improved beyond the targets. In particular this is the case for the Baltic Proper. The extra improvement of water transparency that can result from current BSAP targets might be associated with small additional benefits but considerable costs. In these cases, costs could be saved through an adjustment of the load reduction targets, while still meeting the BSAP targets for water transparency. A revision of targets would have to be based on models, where costs as well as the final impact of measures on water transparency are taken into account.

- *Swedish nutrient targets could be met at lower cost*

An analysis of the Swedish national nutrient policy suggests that the BSAP catchment targets will be twice as expensive as current domestic targets, if both sets of targets are to be met at minimum cost. This is mainly explained by the further restrictions on the spatial allocation of abatement. For example, the concentration of Swedish phosphorus reduction efforts to the Baltic Proper catchment, required by the BSAP agreement, implies that expensive measures in the Baltic Proper catchment replace less expensive measures in other catchments. This raises the question of whether the same phosphorus reduction to Baltic Proper could be achieved in a less costly way by allowing for phosphorus measures in other catchments, while “discounting” their value to account for their lower impact on the Baltic Proper.

The analysis presented below shows that further abatement of nitrogen oxide emissions in the energy sector and, in most locations, further construction of buffer strips and reductions of emissions from households not connected to waste water plants are not cost-effective components in the current policy against nutrient emissions unless it can be shown that these measures are associated with large enough positive side-benefits e.g. for air quality, biodiversity and local water quality. For the same cost as actual policies, relatively large phosphorus reductions to the Baltic Proper can be undertaken through a reallocation of the nutrient reduction budget, and this reallocation will come at a comparatively modest cost in terms of smaller reductions of nitrogen loads. A further debate on the relative merits of reductions in nitrogen and phosphorus loads, respectively, therefore seems highly motivated.

1 Introduction

Eutrophication of the Baltic Sea is a major environmental problem. Although the conditions in the sea depend on both natural processes and human activities, human activities over the last century have dramatically increased nutrient inputs and are judged to be a major cause of the current over-enrichment (Wulff et al., 2007). The degradation of the Baltic Sea is explained by the fact that nutrient polluters in the surrounding countries can use the sea as a pollutant sink without having to consider or pay for the consequences. Thus, there is a so-called market failure whose solution requires international cooperation.

For more than 30 years attempts have been made to control the eutrophication of the Baltic Sea through international agreements and national policies. In 1974, the countries surrounding the Baltic Sea signed the convention on protection of the Baltic Sea marine environment, under which the so-called Helsinki Commission, HELCOM, was created. HELCOM administers co-operation, supports research, defines emission criteria and adopts recommendations on preventive measures against emissions (Ebbesson, 1996). While the 1974 Convention was relatively vague with regard to the requirements for pollution control, this was partly changed in the revision in 1992, where requirements for best available technology (BAT) and best environmental practice (BEP) were introduced for land-based pollution sources (Ebbesson, 2000).

No binding agreement was made in the Convention regarding the necessary load reductions to the Baltic Sea. Instead nutrient reduction targets for the Baltic Sea were originally defined in the Ministerial Declarations of 1988 and 1990. These declarations stipulated that emissions of nutrients to the Baltic Sea should be reduced by 50 % between 1987 and 1995, compared to emissions in 1985. This target was never met, however. The Baltic Sea Action

Program from 2007 defines a new set of provisional load reduction targets which are differentiated over the Baltic Sea's seven basins (HELCOM, 2007b). For these new targets, a new allocation of the abatement burden has been agreed upon.

The failure to reach earlier targets is sometimes explained by a lack of appropriate institutions, e.g. few enforcement powers (EC, 2005), sub-optimal national and sectoral policy instruments (EC, 2005) and lower level governments that are reluctant to enforce regulations when local actors have to bear the abatement costs (Eckerberg, 1997). Another reason may be the large costs associated with these targets (Gren et al., 1997; Gren, 2008) together with uncertainty about both the costs and benefits of emission reductions. In order to reduce the costs for meeting nutrient targets, NEFCO[1] (2008) has initiated a study of the feasibility of a nutrient trading scheme. NEFCO concludes that there are no fundamental judicial obstacles that hinder the introduction of a decentralized regional nutrient trading system with large similarities to the European carbon emissions trading system, ETS. However, they also emphasize that considerable efforts remain to develop such a decentralized system while accounting for existing environmental legislation.

The report presented here discusses how the costs for Baltic Sea nutrient policies can be reduced. This is done through:

1. Analysis of how and to what extent the costs of meeting the BSAP load targets, can be lowered by centralized nutrient trading amongst the countries surrounding the Baltic Sea. Also, it is investigated whether costs could be saved through an adjustment of load targets while still meeting the underlying objective of improved water quality. A numerical model is used for the calculations.
2. The use of a numerical model to identify and compare the cost-effective strategies for meeting Sweden's BSAP-targets with the corresponding strategy under the current Swedish national targets.
3. Evaluation of actual Swedish policy with regard to cost-effectiveness and environmental performance.

[1] NEFCO finances investments and projects in Russia, Ukraine, Estonia, Latvia, Lithuania and Belarus, in order to generate positive environmental effects of interests to the Nordic region.

The report includes a review of the literature on the optimal design of policy instruments for nutrient load reductions. This is done in order to take into account that transports of nutrient and transformation processes and the ecosystem responses are complex and that the multiple governments on international, national and local level take decisions on water management.

The study presented here is related to Gren (2008) that calculates cost-effective policies under the BSAP targets and Elofsson and Gren (2004) that evaluate Swedish policies against nitrogen loads between 1995 and 2000. This study adds to Gren (2008) through *(i)* an analysis of the environmental consequences of a cost-effective strategy and the associated implications for future target revisions and *(ii)* an investigation of the implications of different initial allocations of load permits for the distribution of net-gains from load permits trade. It adds to Elofsson and Gren (2004) also by extending the time span for the evaluation to 2005 and by including phosphorus in the analysis.

During the last 15 years or so, a number of studies of cost-effective nutrient load reductions on a Baltic-wide scale have been conducted. A couple of those investigate cost-effective solutions under the earlier HELCOM reduction targets (Gren, Elofsson and Jannke, 1997; Ollikainen and Honkatukia, 2001). Another paper analyses a somewhat more arbitrary target level for nitrogen loads (Schou et al., 2006). Similar to the present study, those[2] build on so-called engineering estimates of abatement costs[3] in combination with costs derived from partial equilibrium models. The circumstance that these papers produce estimates that differ can be explained by e.g. differences in measure coverage, choice of target formulation, differences in data and differences in assumptions about the capacity of different abatement measures (Elofsson, 2008). In contrast to the above mentioned studies, Johannesson and Randås (2000) use a computable general equilibrium (CGE) model – which takes into account the dispersion effects on other sectors – to analyse Baltic-wide nitrogen load trade. However, the need for a CGE-analysis for activities in primary production sectors has sometimes been questioned, since price changes in these sectors often have a small impact on other sectors

[2] Also Gren (2008) and Elofsson and Gren (2004) use this type of methodology.
[3] "Engineering costs" mean simply that costs for each specific measure are calculated based on information about investment, operation and maintenance costs and simple calculations of opportunity costs.

(Brännlund and Kriström, 1996). Since the agricultural sectors' share of total production is small in most of the Baltic Sea countries, policies directed towards this sector could hence be modeled within a partial equilibrium approach without resulting in large errors. However, also the waste water and the fossil fuel sectors are of concern for nutrient policies. In the former case, prices are determined by governments and not by markets in the region. Although this could be modeled in a general equilibrium framework, considerable assumptions have to be made that could limit the relevance of the approach. Policies directed towards the fossil fuels might have implications for other markets than the fossil fuel market in question and might imply that a CGE-framework could shed additional light on the costs of meeting nutrient targets for the Baltic Sea. However, given that there is no available CGE-model that captures the relevant sectors in the regions in a suitable manner, together with the considerable time required to build such a model, a partial equilibrium framework is chosen here.

The remainder of the report is organized as follows. Chapter 2 gives a short description of the rather complex links between emissions and loads of nutrients to the Baltic Sea and the environmental conditions of the sea. Chapter 3 presents the Baltic Sea Action Plan (BSAP) and the targets stated therein. It also contains an overview of international policy instruments currently affecting the nutrient loads to the Baltic Sea. Chapter 4 gives a corresponding description of Swedish national targets and the Swedish mix of policy instruments, followed by a calculation of the costs and effects of Swedish nutrient policies since 1995. Chapter 5 explains the concept of cost-effectiveness and its implications for the international as well as the domestic distributions of nutrient load abatement. This is followed by a review of the economic literature on optimal policy instrument. The review highlights some natural and institutional characteristics specific to the Baltic Sea pollution problem. Chapter 6 present the numerical model used to identify cost-effective policies. Chapter 7 presents the cost-effective solutions to different versions of the BSAP targets and the associated distribution of costs and abatement. The potential gains from basin-wise nutrient load trading are calculated. Chapter 8 presents and compares the cost-effective strategies under the BSAP targets (which have not yet been implemented nationally) and the cost-effective strategies under the current

Swedish nutrient target. Actual Swedish policy is compared to the cost-effective one under the BSAP targets, and potential improvements are analyzed with respect to costs and environmental effects. In chapter 9, the report is summarized and the results are discussed.

The reader who just wants to get the essence out of the report may just read the brief summaries that end most chapters and thereafter continue to Chapter 9 where the main results are summarized and discussed.

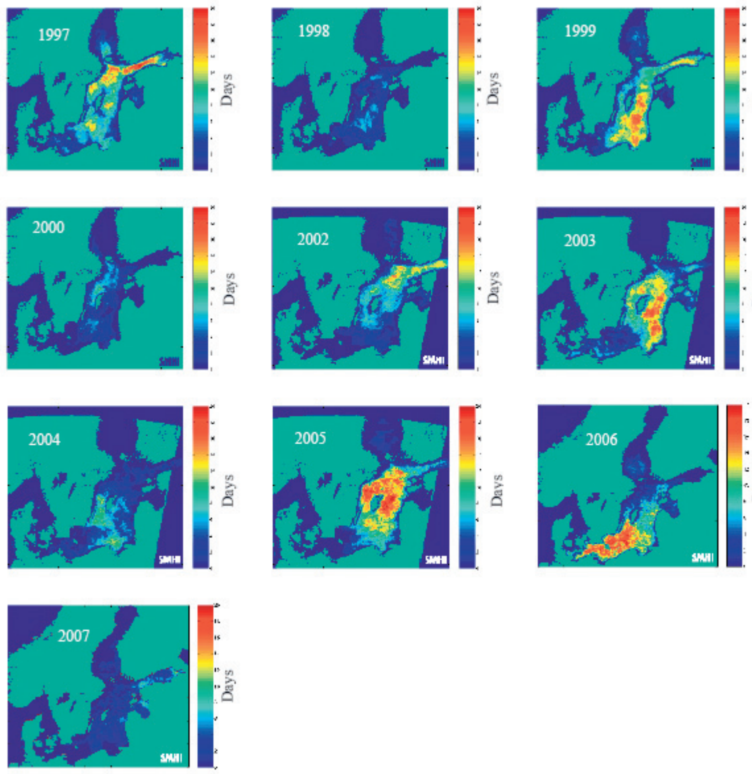
2 Emissions, loads and the ecosystem response

This chapter briefly describes the causes and effects of eutrophication, and the different natural processes that determine the impact of nutrient loads on the Baltic Sea. A concise survey of the scientific literature identifies increased nutrient loads as a major cause of eutrophication and shows the need to account for differences in pollutant pathways when evaluating the effect of different measures at various locations. Moreover, nutrient transports between different basins in the sea as well as nutrient exchange with the atmosphere and bottom sediments affect the environmental damage from pollution. It is noticed that recent research has highlighted the links between eutrophication and fishery. The chapter ends with a summary.

2.1 Emissions and their impact on the Baltic Sea

Eutrophication is caused by high nutrient concentrations which stimulate the growth of algae. This leads to impaired water quality, demonstrated by e.g. extensive blooms of potentially toxic blue-green algae (cyanobacteria), that are a nuisance to bathers and others searching recreation along the coasts of the Baltic Sea. The geographical distribution of blue-green algae blooms in the Baltic Sea varies between years. Blooms are, however, insignificant in the two northern basins, see figure 2.1. In addition, the decay of algae leads to oxygen deficit in the deepwater and thereby causes damage to biodiversity in the Baltic Sea. Locally, human-induced oxygen deficit was found in the Baltic Sea bottoms as early as in the 1930s, but it was not until the 1960s that the phenomenon became widespread, and currently, the Baltic Sea has the largest dead zone in the world (Diaz and Rosenberg, 2008).

Figure 2.1 Number of days with cyanobacterial blooms during 1997-2007, based on NOAA-AVHRR satellite imagery. (Year 2001 is missing due to antenna malfunction at the receiving station.)



Source: HELCOM (2008b).

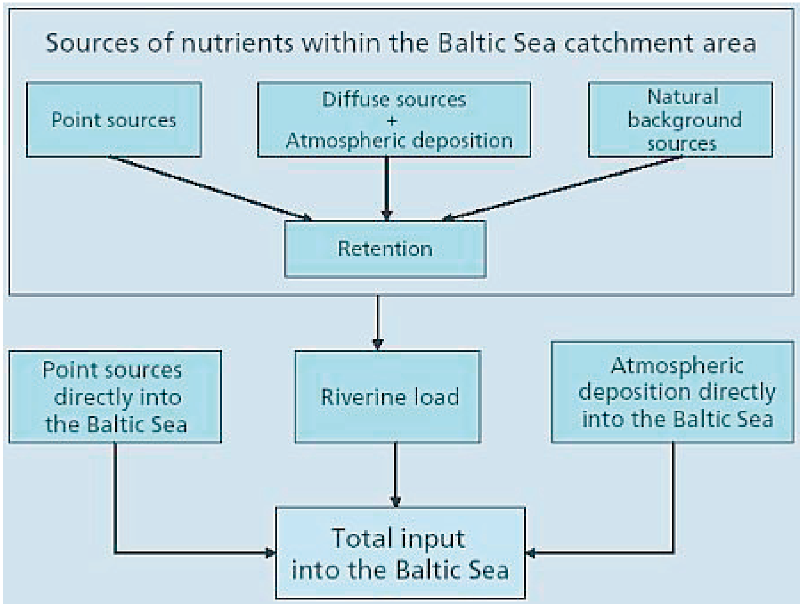
The cause of eutrophication is the excessive nitrogen and phosphorus emissions, coming mainly from land-based sources within the Baltic Sea catchment area. About 75 % of the nitrogen load and at least 95 % of the phosphorus load enter the Baltic Sea via rivers or as direct waterborne discharges. About 25 % of the nitrogen load comes as atmospheric deposition and thereof, 40 % originates from sources outside the Baltic Sea drainage basin (HELCOM, 2007b). From the 1970s to the mid 1990s, total riverine loads of nutrients to the Baltic Sea have been fairly constant (Stålnacke et al., 1999). As a result of the collapse of the Soviet Union and the associated decline in agricultural activity, riverine nitrogen loads from Estonia fell rapidly. The reduction in phosphorus loads, however, has been smaller from the 1980s to the beginning of this century (Iital et al., 2005). In Latvia, the response of loads to the fall in agricultural production has been slow and limited over the same time period (Stålnacke et al., 2003). Riverine nutrient loads have fallen in Finland over the same period, which is explained by abatement efforts in wastewater treatment plants. On the other hand, there is no clear evidence of decreasing loads from agricultural activities (Räike et al., 2003).

According to HELCOM (2008a) the main sources for inputs of nitrogen and phosphorus to the Baltic Sea are:

- Atmospheric emissions of airborne nitrogen compounds from combustion of fossil fuels (transportation, heat and power generation), and from animal manure and husbandry.
- Point sources including inputs from municipalities, industries and fish-farms discharging both into inland surface waters and directly into the Baltic Sea.
- Diffuse sources in agriculture, managed forestry and urban areas.
- Natural background sources, e.g. natural erosion and leaching from unmanaged areas and the nutrient losses from e.g. agricultural and managed forested land that would occur irrespective of human activities.

Nutrients from inland sources are subject to retention, i.e. some of the nutrients never reach the coastal waters but are captured in soils, vegetation or bottom sediments or lost to the air through denitrification. This implies that only a fraction of the emissions from the sources is discharged into the sea. The size of retention varies over the Baltic Sea drainage basin. Thus, the effect that a given emission reduction has on coastal load depends on where the emission source is located. In general, measures undertaken close to the coastal zone have a larger impact on coastal load than measures undertaken further upstream. There is large uncertainty about the size of retention and how it varies throughout the drainage basin. In particular, this holds for phosphorus retention, which is explained by phosphorus retention being very dependent on micro-level geographical characteristics and phosphorus transports being associated with long time lags. In figure 2.2, the major nutrient pathways are illustrated.

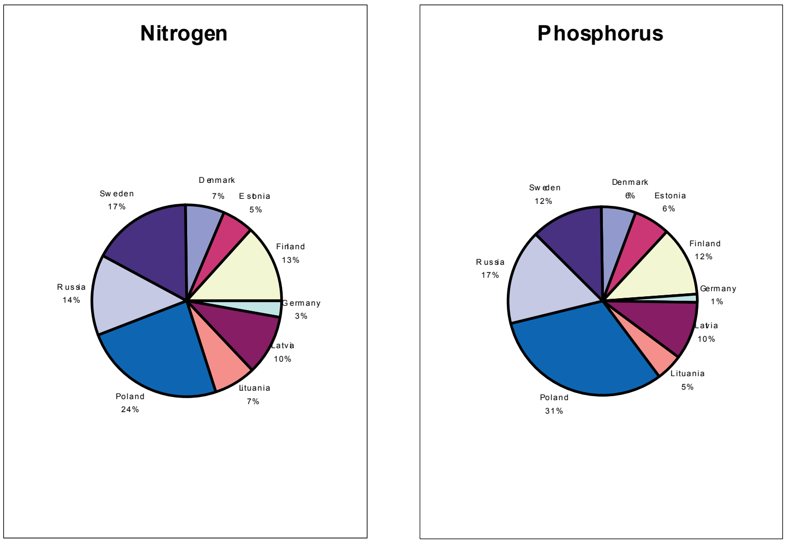
Figure 2.2 Sources of nutrients in the Baltic Sea catchment area.



Source: Helcom (2008a)

Throughout the Baltic Sea drainage basin, emissions of nutrients have been reduced at the sources since 1985 (Lääne et al., 2001), but nutrient concentrations in coastal waters have remained stable or only decreased slightly. This may be explained by the inherent complexity of inland and sea ecosystems and long response times. In figure 2.3, the contribution of different HELCOM countries to the total loads of nitrogen and phosphorus in 2005 is shown. Poland is the largest contributor to both nitrogen and phosphorus loads, followed by Sweden and Russia.

Figure 2.3 Proportion of waterborne inputs of nitrogen and phosphorus into the Baltic Sea by HELCOM countries in 2005. These include inputs from natural background sources as well as anthropogenic sources.



Source: HELCOM[4]

The Baltic Sea consists of several marine basins. Some of these basins are more sensitive to changes in nitrogen concentrations and others to changes in phosphorus concentrations. The concentrations of nutrients in one basin depend both on the loads of nutrients directly emitted into to the same basin and on nutrient imports from other basins (Gren and Wulff, 2004). Loads to one basin will therefore eventually affect nutrient concentrations in all basins (Gren and Wulff, 2004). The final reduction in concentrations can be larger than the reduction in coastal loads because of different biochemical processes in the sea, where e.g. there is an exchange of nutrients between the sea and the sea's bottoms on one hand and the sea and the atmosphere on the other (Vahtera et al., 2007). It takes several decades, however, before the full effect of a change in the loads to coastal waters materializes. Different symptoms of eutrophication, such as reduced water

[4] http://www.helcom.fi/environment2/ifs/ifs2007/en_GB/nutrient_load/, as available 2008-10-16.

clarity, increased primary production, blue-green algae blooms and increased bottom areas with oxygen deficit, depend to a varying degree on nitrogen and phosphorus concentrations in the sea. Through different mechanisms in the sea, reductions of phosphorus pollution will affect nitrogen concentrations, but nitrogen load reductions will not affect phosphorus concentrations (Wulff et al., 2007).

Water quality in the coastal zone depends both on nearby riverine nutrient loads and on nutrient imports from the open sea. Open sea water quality therefore affects coastal waters. This is important as most citizens living in the riparian countries may travel to coastal areas to enjoy bathing, boating and recreation and hence, attach particularly high value to the conditions of the coastal zone.

Research has recently shed new light on the links between eutrophication and fishery. By catching fish, commercial fishery has removed nutrients corresponding to 2.4 and 18 % of the total anthropogenic loads of nitrogen and phosphorus, respectively, to the open Baltic Sea between the end of the 1970s and the end of the 1990s (Hjerne and Hansson, 2002). Thus, in principle, fishery provides an option to control eutrophication. The current state of most fish stocks, however, does not permit increased fishery. Instead reduced fishing pressure is deemed necessary over a period of time for e.g. cod stocks to recover (Hansson et al., 2007). It has also been shown that the total fish biomass in the Baltic Sea has increased over the latter half of the 20s century due to increased eutrophication[5] (Thurow, 1997). This has made possible a 5- to 10-fold increase in Baltic Sea fish catches[6] over the past century (Hansson et al., 2007). Thus, a substantial reduction in nutrient loads to the sea will most likely reduce fish biomass in the Baltic Sea (Hansson et al., 2007), implying less fish for human and animal consumption.

[5] Increased areas of oxygen-free bottoms due to eutrophication have had a negative impact on the reproduction of cod. A good recovery of cod stocks, however, requires reduced fishing pressure - a change in the eutrophication level is not a necessary condition although it would improve the situation even more (Hansson et al., 2007).

[6] This refers to the total fish catches. The catches of sprat and herring have remained large in later years and therefore, total fish catch measured in tons is large, although the cod catch has fallen.

2.2 Summary

In this chapter, the links between emissions at the sources and the conditions of the Baltic Sea are described. It is noted that:

- Multiple sectors and sources contribute to the pollution of the Baltic Sea.
- Only a fraction of the emissions from the sources finally reach coastal waters.
- The Baltic Sea ecosystem reacts slowly and in complex ways to changes in nutrient loadings.
- Marine basins vary with regard to their sensitivity to nutrient loads.
- There is a considerable exchange of nutrients between different marine basins as well as between the open sea and the coastal zone.
- Algal blooms in the Baltic Sea vary from year to year depending on the weather conditions.
- In some parts of the Baltic Sea drainage basin, there has been a small reduction in nutrient loads due to e.g. reduced agricultural activity or efforts to reduce phosphorus emissions from point sources, but there is little evidence of a change in the environmental state of the sea.

3 International targets and policies

Multiple international regulations are in place which, intentionally or unintentionally, affect nutrient loads to the Baltic Sea. Here, the Baltic Sea Action Plan is first described in section 3.1, where the international targets for eutrophication are described. The link between the environmental target for water transparency and the agreed nitrogen and phosphorus load reductions to different marine basins from different countries is explained. In section 3.2 policy instruments currently applied at the international level, e.g. through HELCOM recommendations, EU legislation and EU's Common Agricultural Policy (CAP), are reviewed and the links between CAP, policies against eutrophication and climate policies are discussed. The chapter ends with a summary in section 3.3.

3.1 The Baltic Sea Action Plan (BSAP)

In 2007, HELCOM launched the Baltic Sea Action Plan (BSAP). In this plan, HELCOM presents revised targets for nutrient load reductions (HELCOM, 2007b). The revision is influenced by recent findings that show that policies for the Baltic Sea have to a too large extent focused on nitrogen reductions, while the importance of phosphorus reductions has not been recognized (cf. Boesch et al., 2006). According to the Ministerial Agreement in Krakow (HELCOM, 2007b) the overarching objective of the HELCOM countries is still to reach good environmental status in the entire Baltic Sea. To this end, HELCOM has adopted several ecological objectives that describe the characteristics of a Baltic Sea unaffected by eutrophication. These objectives are:

- Concentrations of nutrients close to natural levels,
- Clear water,

- Natural level of algal blooms,
- Natural distribution and occurrence of plants and animals and
- Natural oxygen levels.

The choice of measurable target conditions associated with these objectives indicates that HELCOM aims at restoring the Baltic Sea to a state similar to that which prevailed in the 1950s.

The BSAP targets replace the earlier target of a 50 % nutrient load reduction, mentioned in the introduction. The new targets are set in order to produce load reductions of nitrogen and phosphorus sufficiently large to meet the ecological objectives defined above.

Targets for water transparency were first defined by HELCOM. Transparency is measured as the so-called secchi depth, which is a measure of the sight depth. The target levels for the seven sea basins are shown in table 3.1. As can be seen in the table, the largest improvements in transparency are required for the Bothnian Sea, the Gulf of Finland and the Gulf of Riga.

Tabell 3.1 Current and target water transparecy, measured as secchi depths

	Observed secchi depth (m) average 1997 – 2003	Target secchi depth (m)	Required increase in secchi depth (m)	Required increase in secchi depth (%)
Bothnian Bay	6.2	6.6	0.4	6
Bothnian Sea	6.2	8	1.8	29
Baltic Proper	7.4	8.1	0.7	9
Gulf of Finland	4.6	5.9	1.3	28
Gulf of Riga	3.3	4.2	0.9	27
Danish Straits	7	7.7	0.7	10
Kattegat	8.5	9	0.5	6

Source: HELCOM (2007c), table 2.

The load reductions required to meet the water transparency targets are calculated by the Baltic NEST Institute (BNI) with the help of an internet based decision support system (http://www.nest.su.se). Data behind these calculations are primarily from official sources within HELCOM and EU.

In order to calculate the nitrogen and phosphorus reductions necessary for each basin, the loads to the Baltic Proper were first reduced in the BNI model until the transparency target for that basin was met. Then, loads to the Gulf of Finland were reduced until the target for the Gulf was met. Step by step, loads to the Gulf of Riga, the Danish Straits and Kattegat were reduced in the same way. For the Danish Straits and Kattegat, it was judged that phosphorus reductions are not necessary and for the Gulf of Riga, the same is assumed for nitrogen reductions. When targets for the above-mentioned basins were achieved, no reductions were needed to the Bothnian Bay or Bothnian Sea. The targets for these basins were automatically reached due to the water exchange between these basins and the Baltic Proper. Accordingly, targets for reductions in the loads of nutrients to each basin were obtained, see table 3.2. The targets imply a reduction of total nitrogen loads by nearly 20 % and total phosphorus loads by more than 40 %. For both nitrogen and phosphorus, the largest reductions required in absolute terms are those for the Baltic Proper.

It should be noted that the choice of reduction target levels is determined not only by the need to meet ecological objectives, but also by the calculation method used. If the researchers had changed the order of the marine basins in their iterative calculations, they would have ended up with other reduction levels. For example, they could have started with the Bothnian Sea, for which there is a need to reduce nutrients, and in that case, reductions to the Bothnian Sea would have been included among the basins with nutrient load reduction targets. In principle, an infinite number of combinations of nutrient load targets that meet the requirement for transparency improvements could be derived. The iterative process by which nutrient reduction targets have been calculated thus affects also the minimum costs of meeting a given environmental improvement. The importance of this for the costs of meeting the ecological objectives cannot be determined with available data.

Tabell 3.2 BSAP basin targets, ton and percent

	Load 1997-2003		Needed reduction		Percentage reduction	
	Nitrogen (ton)	Phosphorus (ton)	Nitrogen (ton)	Phosphorus (ton)	Nitrogen (%)	Phosphorus (%)
Bothnian Bay	51,436	2,585	0	0	0	0
Bothnian Sea	56,786	2,457	0	0	0	0
Baltic Proper	327,259	19,246	94,000	12,500	29	65
Gulf of Finland	112,680	6,860	6,000	2,000	5	29
Gulf of Riga	78,404	2,180	0	750	0	34
Danish Straits	45,893	1,409	15,000	0	33	0
Kattegat	64,257	1,573	20,000	0	31	0
Total	**736,714**	**36,310**	**135,000**	**15,250**	**18**	**42**

Source: HELCOM (2007a).

In order to quantify the reductions that each country should undertake, the load reductions that would be expected if all countries, including Russia and Belarus, complied with the HELCOM Recommendation for municipal wastewater treatment[7] and/or the EU Wastewater Directive, were first deducted from the necessary reductions in table 3.2 above. Thereafter, the remaining load reduction to each basin was distributed among the HELCOM Contracting States proportionally to their present load contributions to that basin, see HELCOM (2007c). This way, load targets for each catchment were obtained. In table 3.3, the reductions required for each catchment include the reduction through compliance with the wastewater recommendation/-directive. The reductions required by the countries through the Ministerial agreement in Krakow, see table 3.3 and 3.4, are about 5,000 tons lower for nitrogen and 2,000 tons lower for phosphorus compared to the total reduction requirements in table 3.2. This is explained by the "external" reductions expected from Belarus, which is not a HELCOM country and hence has not committed to any reductions.

[7] HELCOM Recommendation 28E/5: Municipal wastewater treatment.

Tabell 3.3 BSAP catchment targets, % of total reduction

	Nitrogen, ton (%)	Phosphorus, ton (%)
Denmark Kattegat	8,281 (6.4)	0 (0)
Denmark Danish Straits	8,486 (6.5)	0 (0)
Denmark Baltic Proper	542 (0.4)	16 (0.1)
Sum Denmark	*17,309 (13.3)*	*16 (0.1)*
Finland Bothnian Bay	0 (0)	0 (0)
Finland Bothnian Sea	0 (0)	0 (0)
Finland Gulf of Finland	1,199 (0.9)	146 (1.1)
Sum Finland	*1,199 (0.9)*	*146 (1.1)*
Germany Danish Straits	4,348 (3.3)	0 (0)
Germany Baltic Proper	1,701 (1.3)	242 (1.8)
Sum Germany	*6,049 (4.7)*	*242 (1.8)*
Poland Baltic Proper	62,395 (48.0)	8,755 (65.6)
Sum Poland	*62,395 (48.0)*	*8,755 (65.6)*
Sweden Bothnian Bay	0 (0)	0 (0)
Sweden Bothnian Sea	0 (0)	0 (0)
Sweden Baltic Proper	8,087 (6.2)	291 (2.2)
Sweden Danish Straits	1,733 (1.3)	0 (0)
Sweden Kattegat	11,128 (8.6)	0 (0)
Sum Sweden	*20,948 (16.1)*	*291 (2.2)*
Estonia Baltic Proper	257 (0.2)	10 (0.1)
Estonia Gulf of Riga	0 (0)	22 (0.2)
Estonia Gulf of Finland	639 (0.5)	190 (1.4)
Sum Estonia	*896 (0.7)*	*222 (1.7)*
Latvia Baltic Proper	2,562 (2.0)	144 (1.1)
Latvia Gulf of Riga	0 (0)	156 (1.2)
Sum Latvia	*2,562 (2.0)*	*300 (2.2)*
Lithuania Baltic Proper	11,746 (9.0)	881 (6.6)
Sum Lithuania	*11,746 (9.0)*	*881 (6.6)*
Russian Baltic Proper	2,821 (2.2)	724 (5.4)
Russian Gulf of Riga	0 (0)	114 (0.9)
Russian Gulf of Finland	4,145 (3.2)	1,661 (12.4)
Sum Russia	*6,966 (5.4)*	*2,499 (18.7)*
Total sum	**130,070 (100)**	**13,352 (100)**

Source: HELCOM (2007c).

Finally, the catchment-wise reduction targets in table 3.3 were summed up to national load reduction targets, see table 3.4. As can be seen from the table, Lithuania and Poland are required to undertake relatively large phosphorus reductions in percentage terms. This is primarily explained by the currently large emissions from wastewater facilities in these countries. Because of these large wastewater emissions, compliance to the Wastewater Directive would imply a substantial reduction of loads. In percentage terms, nitrogen reductions are more equally distributed among the countries, except for Finland, Russia and Estonia where smaller nitrogen reductions are required.

Tabell 3.4 Country loads and country targets

	Loads in sub basins with a reduction need (97-03)		Country reduction allocations		Percentage reductions	
	Nitrogen (ton)	Phosphorus (ton)	Nitrogen (ton)	Phosphorus (ton)	Nitrogen	Phosphorus
Germany	20,848	534	5,621	242	27	45
Denmark	57,501	51	17,207	16	30	31
Estonia	19,054	1,261	896	222	5	18
Finland	15,852	578	1,199	146	8	25
Lithuania	45,109	1,336	11,746	881	26	66
Latvia	10,447	1,613	2,561	300	25	19
Russia	89,386	6,683	6,967	2,500	8	37
Poland	215,350	13,717	62,395	8,755	29	64
Sweden	72,762	860	20,780	291	29	34
Total	**546,309**	**26,633**	**129,372**	**13,353**	**24**	**50**

Source: HELCOM (2007a)

It is not clear from the documents whether countries have committed to meet targets on catchment or country level. If the basin targets in table 3.2 are to be met, however, it is necessary that the catchment targets in table 3.3 are applied. If country targets stated in the BSAP agreement were interpreted as applying to the national level instead of the catchment level, the targeted reductions to different basins will not be met. Therefore, in this report, calculations are based on the basin and catchment targets, respectively.

For targets to be met, implementation into national policy and incentives for participation by the involved countries are important prerequisites. Although the new HELCOM targets described above are more closely linked to the ecological objectives than the earlier targets, the political and legal status of the targets and the corresponding country allocation of load reductions are unclear and it is not known when and if these targets will be implemented into national legislation in the contracting countries. In addition, the international targets for eutrophication have been developed without explicit consideration of benefits and costs. Thereby, policies for the Baltic Sea differ from policies against e.g. acidification, where cost-effectiveness has been a major determinant of the burden sharing (e.g. Atkinson, 1998) and for the CO_2 burden sharing within the EU, where both efficiency and equity have affected the outcome (Marklund and Samakovlis, 2007). In addition, the Kyoto Protocol allows for load trading, which facilitates cost-effective reductions. With such trading and assuming no or low transaction costs, the allocation of the abatement burden becomes a question of income distribution only, and will not matter for the costs of abatement.

3.2 Policy instruments at the international level

There are no internationally common policy instruments designed with the particular purpose to reduce eutrophication in the Baltic Sea. HELCOM has issued a large number of recommendations regarding measures that should be undertaken in order to reduce nutrient emissions. These recommendations, however, are not legally binding to the contracting countries.

All HELCOM countries except Russia are EU Members and thus subject to a number of EU directives that have implications for inland and coastal water quality. One is the Water Framework Directive, which requires that good inland water status is achieved through integrated river basin management (see e.g. Mostert, 2003). Another one is the Nitrates Directive, which promotes different nitrogen reducing management practices in the agricultural sector. A third is the Urban Waste Water Directive, which regulates collection and treatment of waste water in urban areas. Typically, measures under these directives will also have a positive environmental impact on the sea. In addition, the EU Commission has proposed a Marine Strategy Directive, which will lead to the establishment of European Marine Regions, based on geographical and environmental criteria. Each Member State, in cooperation with other relevant Member States and third countries within a Marine Region, will be required to develop strategies for its marine waters. The outcome of this directive in terms of policy instruments and load reductions still remains unclear.

Policies in different fields affect both the need for emission reductions and the relative costs of different abatement options. The most obvious case is the European Common Agricultural Policy (CAP), but also climate change policies have links to eutrophication e.g. through the EU-wide system for carbon trading.

The CAP has for many decades provided incentives for intensive agricultural production, which has led to larger emissions of nutrients than would have been the case without the CAP. In 2003, the so-called single payment scheme was launched with an aim to reduce the negative impacts of the earlier policy when agricultural support was linked to production. Although the reform was mainly driven by a need to reduce overproduction, the negative environmental effects from intensive production were also mentioned as a motive. The single payment scheme decoupled much of the agricultural support from production and is therefore expected to lead to lower nutrient emissions on the overall level (EC, 2007) although the impact may vary regionally with regard to both direction and magnitude (see e.g. Lehtonen et al., 2007).

In recent years, some agricultural support has been shifted over to the so-called rural development programs. For these programs, each country can decide on the extent and design of agri-environmental policy measures that address national environmental problems such as e.g. nutrient emissions. Even when such instruments are applied, they may barely compensate for the increase in nutrient emissions caused by support linked to production (Brady, 2003).

The CAP is not the only determinant of agricultural activity. The size and composition of agricultural production and hence nutrient emissions are also affected by the world market for agricultural products. The recently increased world market prices on agricultural products (see e.g. OECD-FAO, 2007) might if the trend continues lead to increased production and thereby counteract the effect of agricultural decoupling reforms. It may also change the relative costs of different abatement options.

Both climate change and climate policies may affect cost-effective abatement strategies for the Baltic Sea. Climate change affects agricultural production and eutrophication both directly and indirectly. The direct impact occurs through the effects of climate change on e.g. nutrient transports. Climate change is expected to lead to a larger riverine outflow of nutrients to the Baltic Sea (Arheimer et al. 2005) and might therefore increase the need for nutrient reductions. The indirect impact is determined by the responses of policy makers and farmers to climatic change (see e.g. Abler et al., 2002). Farmers in northern Europe may benefit from global warming as long as the temperature does not increase too much (EC, 2008a) and thereby increase their production which could increase the emissions and loads of nutrients. If this actually happens, it would make the eutrophication problem in the Baltic region worse. Policies developed in response to climate change might on the other hand also reduce nutrient emissions. Increased costs for fossil fuels, e.g. through carbon taxation and carbon trade systems, is likely to reduce the amounts of airborne nitrogen emissions (see e.g. Östblom, 2007; Östblom and Hammar, 2007). Climate policy may lead to larger areas with cultivation of perennial energy crops on arable land, which may again reduce nutrient emissions (Börjesson, 1999). Thus, the net effect of climate change and climate policy for the Baltic Sea is not easily judged and the relative costs of abatement in different parts of the sea's drainage basin may well be affected.

3.3 Summary

This chapter has briefly presented the development of eutrophication targets for the Baltic Sea. Policy instruments for reduced eutrophication at the international level were reviewed. Summarizing the chapter, it shows that:

- Since the establishment of HELCOM in the 1970s, considerable efforts have been made to reduce nutrient loads through international agreements.
- Targets agreed at the international level have not been met so far. A possible explanation for the failure to reach targets is that although agreements might be associated with positive net benefits in total, this may not hold for the individual countries

when the abatement burden is distributed without consideration of participation incentives.

- Old targets for nutrient load reductions have recently been replaced with new provisional targets, the BSAP's load targets for sea basins. These targets are derived so that they should lead to pre-specified improvements of the water transparency in the Baltic Sea. When developing these new targets, costs and benefits of the distribution of the abatement burden have not been taken into account, at least not explicitly.
- At the international level, there are currently no policy instruments that are designed with the purpose to reduce Baltic Sea nutrient loads in a cost-effective manner. Through several EU directives, there are policy instruments and regulations in place that affect sea water quality, but policies are scattered over different sectors and mainly constructed with a focus on emission reductions at the sources. Hence, the instrument mix does not ensure a cost-effective allocation of abatement across countries or policies. This is likely to increase the costs for achieving marine targets. However, the implementation of the Water Framework Directive and the Marine Directive might lead to a change in this regard.

4 Swedish domestic policy

The Swedish government started several decades ago to develop policies with the aim of reducing nutrient emissions. In this chapter, Swedish national targets for nutrient reductions are discussed in section 4.1. It is observed that the nutrient reduction targets laid down by the Swedish Parliament differ substantially from those agreed upon through the BSAP. This is followed by a brief presentation of national policy instruments in section 4.2, where it is shown that there are numerous of instruments, but no mechanism that ensures a cost-effective allocation of abatement efforts between or within sectors. In section 4.3, the costs and effects of Swedish nutrient policies since 1995 are compiled. The results suggest that the total annual nutrient abatement cost amounts to more than 300 Million EUR per year. The chapter ends with a summary in section 4.4.

4.1 Targets at the national level

Based on the earlier HELCOM target to reduce loads by 50 %, the Swedish Parliament decided to reduce halve nitrogen loads between 1987 and 1995 (RK, 1988). Towards the end of this period, it became clear that the target would not be reached on time and, accordingly, the parliament chose to revise the target and postpone the deadline. Currently, there is an official environmental quality objective called 'Zero Eutrophication', which stipulates that nutrient levels in soil and water should be such that they do not adversely affect human health, the conditions for biological diversity or the possibilities to use land and water in various ways. Several interim, operative targets should contribute to this end:

- By 2010 Swedish waterborne anthropogenic emissions of phosphorus compounds into lakes, streams and coastal waters should decrease by at least 20 % from 1995 levels. The largest reductions should be achieved in the most sensitive areas.
- By 2010 Swedish waterborne anthropogenic loads of nitrogen compounds into sea areas south of the Åland Sea should be reduced by at least 30 % compared with 1995 levels.
- By 2010 emissions of ammonia in Sweden should be reduced by at least 15 % compared with 1995 levels.
- By 2010 emissions of nitrogen oxide to air in Sweden should be reduced to 148,000 tons.

The motive for limiting the nitrogen target to sources draining to the Åland Sea and further south is that only marine basins located south of Norrtälje municipality are considered to be sensitive to nitrogen. The target for reductions in nitrogen oxide emissions is not only relevant for reducing eutrophication but also for other environmental quality objectives such as 'Natural Acidification Only' and 'Clean Air' while the ammonium target is not explicitly included under any other environmental quality objective.

In 2006, an international expert group appointed by the Swedish EPA delivered its conclusions on eutrophication of Swedish seas (Boesch et al., 2006). The expert group questioned the current Swedish policy. They emphasized the need for phosphorus reductions to reduce eutrophication in the open Baltic Sea and warned that large nitrogen load reductions might lead to increased blooms of blue-green algae. However, they said, nitrogen reductions could still be relevant to address eutrophication in certain, particularly sensitive coastal regions along the Swedish east coast. The expert group also judged that nitrogen reductions should still be the appropriate tool to reduce eutrophication along the Swedish west coast. The EPA, in turn, responded to these conclusions by recommending that ambitions with regard to nitrogen reductions to the Baltic Sea should not be reduced because nitrogen reductions may have an effect in the long run, but that efforts to reduce phosphorus emissions should be strengthened (EPA, 2006a). For the Swedish west coast, the EPA recommends that both nitrogen and phosphorus reductions should be carried out because the EPA believes that phosphorus reductions may positively affect the sea given that nitrogen reductions are also carried out. Thus, there are diverging views on the relative benefits of reducing the two nutrients.

Once the Ministerial Agreement on the new BSAP targets was set in 2007, the Swedish EPA suggested the first two operative targets above should be revised and adapted to the BSAP targets. For the time being however, the internationally agreed targets for Sweden and the Swedish national targets for nutrient load reductions differ from each other. Thus, even if policies in place were designed to achieve current national targets in a cost-effective manner, they may not be cost-effective with regard the BSAP targets.

4.2 Policy instruments at the national level

In Sweden, a large flora of policy instruments is in place to address nutrient pollution. Each sector is presumed to develop its own policy instrument combination, instruments that ensure cost-effectiveness across sectors, are not in place[8].

Some measures are voluntary, such as e.g. self-regulation, education and agri-environmental support schemes. Farmers apply self-regulation in cooperation with the Swedish farmer's organization (see e.g. LRF, 2008) and through the Rural Development Program, the Swedish Board of Agriculture informs and educates farmer on nutrient losses. Farmers can also voluntarily reduce nutrient losses through changes in agricultural practice and land use and achieve compensation for this through the Rural Development Program. Support to abatement is sometimes differentiated between regions with regard to the environmental effect.

Many mandatory policy instruments have been introduced to reduce nutrient emissions. Some of those imply direct regulation of emissions, technology or management methods at the individual plant or firm level. Wastewater treatment plants are regulated with regard to technology and emissions, nitrogen oxide emissions from stationary combustion plants are regulated with regard to emissions, and farms are subject to regulation of manure storage technology and manure spreading practices. Mandatory environmental taxes and fees are also applied in order to reduce

[8] In principle, the EPA is responsible for policy coordination across sectors.

nutrient emissions. A tax on fertilizer nitrogen was introduced in 1984 with the purpose to reduce fertilizer use, but has recently been revoked. A charge on nitrogen oxide emissions from stationary combustion plants over a certain size limit was introduced 1992.

In its response to the new BSAP targets, the Swedish EPA suggests additional national measures against waterborne nutrient loads that are, with few exceptions, directed towards the agricultural sector (EPA, 2008a). The national targets for ammonium and nitrogen oxide emissions are also suggested to be revised (EPA, 2008a), but it is judged that the revised ammonium target will be reached with the current policy. A more stringent target for nitrogen oxide emissions is, however, judged to require additional and/or more stringent instruments in the shipping, energy and transport sectors (EPA, 2008a).

4.3 Swedish policies for nitrogen and phosphorus since 1995: costs and effects

In this section, the costs and effects of Swedish environmental policy changes after 1995 are estimated. Measures judged to be undertaken with the aim to reduce nutrient load are collected in table 4.1. Reductions by industries and municipal wastewater treatment plants are judged to be driven by changes in legislation on general and plant level. Catch crops cultivation, spring plowing, wetland and buffer strips construction are assumed to be undertaken due to the support provided through the partly EU-financed Rural Development Program. Wetlands have also been created outside the agricultural sector with the help of support from e.g. the Local Investment Programmes (LIP). Reductions in ammonia emissions from the agricultural sector are assumed to be achieved through a combination of investment support and regulation. Reductions in nitrogen oxide emissions are assumed to be achieved through regulation of technology in the transport and energy sector. Investment in wastewater treatment in other countries has been financed by the Swedish government.

For some abatement measures, the reduction is calculated through a straightforward comparison of the emission levels in

1995 and 2005[9]. One difference between the calculations in this report and those made by the Swedish EPA (2007) is that the EPA figures for agriculture build on the total change in production patterns in the agricultural sector, much of which is due to Sweden joining the EU, while in this report only effects caused by environmental policy changes in the time period are included. Yet, for each single measure, the levels in 1995 and 2005 are used for the calculations here. Moreover, figures in this report for the waste water sector take into account population development in different regions when calculating the effect of environmental policy, which is not relevant in the EPA report. Also, for the industry, changes in the production volume since 1995 are taken into account in this report when calculating the impact of environmental policy, which is not the case in the EPA report.

The load reductions are shown in table 4.1 together with estimates of the associated cost. Cost estimates are mainly based on the model described and used in chapter 5-7. In a few cases[10], costs are obtained from reports produced by the Swedish Board of Agriculture[11]. The compilations show that annual Swedish nitrogen and phosphorus loads to coastal water have been reduced by approximately 15,500 and 530 tons, respectively. The costs for these reductions are estimated to exeed 330 Million EUR per year. The largest reductions are due to improvements of wastewater treatment. This measure is also associated with the largest cost. In spite of the often emphasized role of the agricultural sector for nutrient load reductions, this sector contributes with a relatively small share of the nitrogen reductions and an even smaller share for phosphorus. Measures against nitrogen oxide emissions from energy and transport account for approximately the same reductions of nitrogen as the agricultural sector but at a higher cost. Investment in wastewater treatment abroad has led to minor nitrogen reductions but substantial phosphorus reductions at a small cost. About 88 % of total cost can be attributed to measures that reduce nitrogen only, while around 10 % of the cost is for measures that only affect phosphorus. Less than 2 % of the costs are for measures that reduce loads of both nutrients.

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[9] This is of course a simplification, implicitly implying that all factors except environmental policy have been constant over the time period or that they have had only insignificant effects on the level of the activities discussed.

[10] For spring plowing and ammonium reductions.

[11] Data for calculation of Swedish reductions and costs can be found in Appendix B.

The results presented in table 4.1 can also be compared to calculations of reductions between 1995 and 2005 made by the Swedish EPA (2007). The EPA calculates the total difference in loads between 1995 and 2005, i.e. when all changes are included and not only those that are caused by environmental policy. The EPA estimates that the reduction of nitrogen loads to coastal waters was 12,900 tons for nitrogen. Phosphorus emissions were reduced by 350 tons at the sources[12]. A comparison with the results presented in table 4.1 indicates that factors outside the environmental policy field have counteracted environmental policies between 1995 and 2005[13].

Data in table 4.1 suggest that the nitrogen reductions since 1995 due to environmental policies are approximately 27 %. Hence, the domestic target of a 30 % reduction of the waterborne nitrogen loads to coastal waters south of Åland Sea (corresponding to 16,890 tons according to EPA (2007)) is nearly achieved. The target for phosphorus, a reduction of emissions to water by 20 %, corresponding to 500 tons, seems to have been achieved.

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[12] It should be noted that the EPA calculates changes in phosphorus emissions at the sources, because of large uncertainties regarding the relationship between emissions at the sources and final loads to the sea. In this report however, available estimates on this relationship has been used throughout all calculations.

[13] Note that there are some additional differences between the calculations presented in Table 4.1 and the ones carried out by the EPA. For instance, the EPA does not include the effect on the sea of reduced nitrogen emissions to air. Furthermore, in this report increases in emissions from some industrial subsectors are not taken into account.

Table 4.1 Annual Swedish load reductions and costs by measure after 1995

	N load red. to coastal water since 1995 (tons)	P load red. to coastal water since 1995 (tons)	Total N red. cost (MEUR)	N red. cost (average cost of reduction to coastal waters, EUR/kg)	Total P red. cost (MEUR)	P red. Cost (average cost of reduction to coastal waters, EUR/kg)	Joint costs of N and P reductions (MEUR)	Total cost of measures undertaken (MEUR)
Industry	1,899	188	42	22	13	71		55
Wastewater treatment	10,152	223	176	17	15	68		191
Wastewater treatment abroad	336	114	2	7	4	35		6
Transport sector (NOx-N)	1,290		30	23				30
Energy sector (NOx-N)	476		19	40				19
Agricultural sector								
Catch crops	300	1					4	4
Spring plowing	51		1	10				1
Wetlands	87	0.3					1	1
Buffer strips		1			2	2,244		2
Ammonium reductions	883		27	30				27
Sum agricultural sector	*1,321*	*2.3*	*27*		*2*		*6*	*35*
Total sum	**15,474**	**527**	**296**		**34**		**6**	**336**

Data for calculation of Swedish reductions and costs can be found in Appendix.

In table 4.2, the distribution of reductions and costs over different drainage basins are shown. Data for the regional distribution of measures are not available for all measures and therefore, in some cases, assumptions regarding this distribution have been made based on the regional differentiation of the policy measures[14] that are assumed to cause the changes.

The compilations presented in table 4.2 indicate that minor reductions are carried out in the Bothnian Bay. In the Bothnian Sea catchment, considerable nitrogen and phosphorus loads reductions are made to a non-negligible cost, in spite of the zero Swedish targets for nutrient reductions in this catchment. The most substantial reductions are made in the Baltic Proper catchment. Costs spent in the Baltic Proper catchment are approximately twice as large as those spent in the Kattegat catchment.

Table 4.2 Annual Swedish load reductions by catchment after 1995 and the cost for these reductions

	N red to-coastal waters since 1995 (tons)	P red to coastal waters since 1995 (tons)	Cost of N reductions, MEUR (average cost EUR/kg)	Cost of P reductions, MEUR (average cost EUR/kg)	Joint cost of N and P reductions (MEUR)	Total cost of measures undertaken (MEUR)
Bothnian Bay	188	19	4 (23)	1.3 (68)		6
Bothnian Sea	1,166	121	23 (20)	8 (63)	0.1	31
Baltic Proper	6,559	121	141 (21)	9 (115)	1.5	151
The Danish Straits	1,184	11	18 (15)	0.7 (68)	0.4	19
Kattegat	3,673	96	71 (19)	8 (89)	3.4	83
Skagerrak	2,462	90	38 (15)	6 (63)	0.3	44
Gulf of Riga	21	33	0.7 (36)	1.4 (42)		2
Gulf of Finland	220	36	0.3 (1)	0.5 (15)		1
Sum	**15,473**	**527**	**296 (19)**	**35 (72)**	**5.7**	**337**

Data for calculation of Swedish reductions and costs can be found in Appendix. Reductions required by HELCOM can be found in Outcomes from the Expert Meetings of the HELCOM Baltic Sea Action Plan, 12.9.2007. HELCOM HOD 22/2007. Helsinki. Skagerrak is not considered in the HELCOM targets.

[14] This is done when a measure is applied only in certain regions, then the effect is assumed to be zero in other regions.

4.4 Summary

This chapter has briefly presented the Swedish national targets for nutrient reductions and the policy instruments employed to reach these targets. Summarizing the chapter, it shows that:

- Swedish national targets for nutrient load reductions may soon become adapted to the BSAP. Even if Swedish policies were cost-effective with regard to the current targets, they may not be so with regard to the BSAP targets. Therefore, a policy change could be motivated.
- There is a broad set of policy instruments in Sweden aiming at reducing nutrient emissions to inland and loads to sea waters, including voluntary policies with and without compensation to the polluters and mandatory policies, such as regulation of emissions and technology as well environmental taxes.
- Calculations show that Swedish policy changes since 1995 have reduced annual nitrogen loads by 15,500 tons and annual phosphorus loads by 530 tons. The total annual cost for these reductions is estimated to exceed 330 Million EUR.

5 The cost-effective solution

This chapter briefly describes the fundamental policy problem from an economic viewpoint. The chapter is mainly intended for readers unacquainted with economic theory or who feel a need to refresh the memory with regard to economic theory underlying the concept of cost-effectiveness and the implementation of cost-effective policies.

If policy makers want to implement cost-effective policies against eutrophication of the Baltic Sea, many different factors need to be considered. Some of these factors relate to the functioning of the environment and the ecosystem, others to the organization of governments and the possibilities for governments to influence the actions of households, firms and lower level governments.

In this chapter, different issues of importance for the design of Baltic Sea eutrophication policies are discussed together with solutions that could lead to more efficient outcomes. The chapter starts with a short recapitulation of fundamental economic theory about cost effectiveness at the international and the national level and cost-effective policy instruments. In section 5.1, the reasons for and the intuition behind cost-effectiveness is presented, followed by a description of international and national cost-effectiveness in 5.2. In section 5.3, the fundamental differences between command-and-control and market based instruments are outlined. In section 5.4, the gains from emission trading are defined. This is followed by a discussion in section 5.5 and 5.6 on how the presence of differences in environmental impact between sources and uncertainty affect the optimal choice and/or design of policy instruments. It is observed that there are straight-forward ways to deal with these problems in policy design. Section 5.7 reviews the use of policy instruments when multiple countries contribute to the environmental problem and it is observed that depending on the distribution of costs and benefits of sea restoration, side-payments can be necessary if a cost-effective abatement strategy is to be realized. In section 5.8, the links between national and local governance are discussed and it is noted that a well-considered distribution of rights and duties between different governmental levels as well as an appropriate design of intergovernmental incentives can be necessary for efficient water management. In section 5.9, transactions costs are briefly discussed and it is observed that although there is limited knowledge on the magnitude of transaction costs, there is at least no empirical evidence that transaction costs would affect the ranking of different policy instruments for water management. Finally in section 5.10, the chapter is summarized.

5.1 Why is cost-effectiveness called for and what are its implications?

In the environmental debate cost-effectiveness is often called for. The European Union requires for example in its suggested Marine Strategy Directive that each member state puts together an action program with cost-effective measures (EC, 2008b). By cost-effectiveness is meant that a given set of environmental targets are reached at minimum cost to society.

A good reason for requiring cost-effectiveness with regard to policies against eutrophication of the Baltic Sea is that the achievement of the environmental targets is going to be costly even if a least-cost strategy is pursued (Turner et al., 1999; Gren, 2008). Cost-effectiveness implies that environmental targets are reached without wasting society's resources on unnecessarily expensive abatement programs. If costs become unnecessarily large, less resources will be available for other environmental and social purposes, such as e.g. biodiversity preservation, schools and health care.

Cost-effectiveness requires that measures are chosen in such a manner that targets are met at the lowest possible cost. For the Baltic Sea where nutrient targets are relatively demanding, this implies that measures with low costs for emission reduction at the sources and high impact on the environmental targets should definitely be included in the cost-effective strategy, but also a number of more expensive measures and measures with smaller effect.

When ‘costs’ are discussed in this context, the cost is determined by the resources that society has to give up in order reach the environmental target. When calculating this cost one should, ideally, take into account all direct and indirect costs. The direct cost is the cost of investment and operation associated with the implementation of measures. By indirect costs is meant costs associated with the policy instruments and its implementation and the policy’s impact on other environmental targets and on other sectors in the economy. In applied studies, cost estimates usually do not include all effects, but simplifications have to be made.

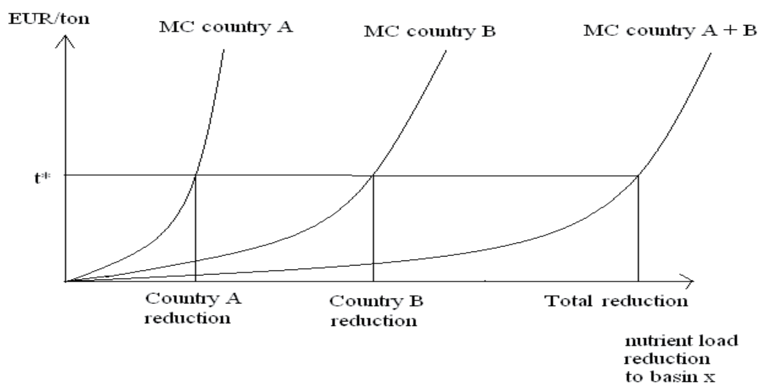
The marginal cost[15] for each measure is defined by the cost of the measure at the source and the impact it has on the chosen environmental target. The larger the impact on the environmental target, the lower the marginal cost and vice versa. Cost-effectiveness implies that the marginal cost for all measures must be equal. When marginal costs are not equal, then it is possible to reallocate resources from more expensive measures to cheaper ones such that the environmental target could be reached at a lower cost. Thus, to find the cost-effective allocation the costs and effects of all measures in the Baltic Sea region must be compared to each other. In addition, cost-effectiveness implies that the timing of a measure should be considered. Depending on the time lag between the implementation of a measure and its effect on the eutrophication of the Baltic Sea, different measures could be cost-effective at different moments in time. For example, there could be an advantage with early abatement in the coastal zone in order to have a rapid effect on the sea, followed later on by inland measures (cf. e.g. Hart, 2002; Laukkanen and Huhtala, 2007). A cost-effective allocation of abatement over time implies that the reduction in each year should be the one that ensures that the environmental target is reached on time at the lowest possible cost. For applied studies, the inclusion of time dynamics requires information about ecosystem response to changes in loads.

[15] The marginal cost is the cost for an additional unit of abatement of the load.

5.2 International and national cost-effectiveness

As mentioned above, the BSAP targets are expressed as load reductions required per basin in the Baltic Sea. Several countries contribute to the loads to each basin. These countries differ with regard to the cost of nutrient load reductions. How should then load reductions be allocated between countries in a cost-effective way? This is illustrated in a simple way in figure 5.1. In this figure, the marginal cost functions for two countries, country *A* and country *B* are included. The marginal costs for nutrient load abatement are assumed to rise with the level of nutrient load reductions. In country *A*, marginal cost rises relatively rapidly with abatement, while in country *B*, the marginal cost rises more slowly. To the right in the figure, the marginal cost functions of the two countries are added up, giving the aggregate marginal cost function. The total reduction required by international decision-makers is shown in the figure[16], and from the aggregated marginal cost curve, one finds that the marginal cost at this reduction level equals *t**. A cost-effective fulfilment of this target implies that only measures with a unit cost below or equal to *t** are implemented. If the reduction is to be carried out in a cost-effective way, country *A* and country *B* must both incur the same marginal abatement cost. Thus, country B abates more than country *A*, because of the larger low-cost abatement options in country *B*.

Figure 5.1 Internationally cost-effective reductions of nutrient loads to basin x.



[16] It is denoted "Total reduction".

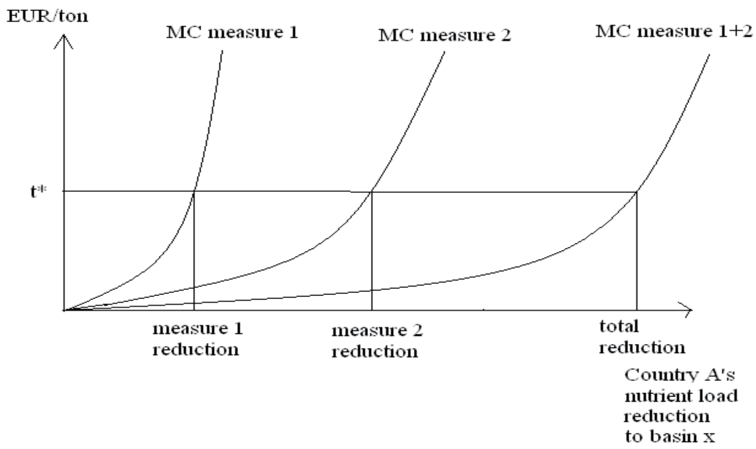
The second step is then for each country to identify what measures should be used within the country in order to reach the national reductions required. This problem can be illustrated in a similar way. The problem is now to choose a combination of different measures, such that the total national load reduction to basin x in figure 5.2 below, is reached at minimum cost. Suppose country A has only two different measures, measure 1 and measure 2. For both these measures the marginal cost function is increasing. The reason for this could e.g. be that for low levels of abatement, the measure can be located in places where the impact on the coastal load is high, but if more of the measure is used, less suitable locations have to be included. The marginal cost curve for measure 1 is increasing faster than that for measure 2, e.g. because more of the possible locations for measure 1 are located in the inland. When the marginal cost curves for measure 1 and 2 are added up, one obtains the country's marginal cost function[17].

The total reduction to basin x required by decision-makers in country A is shown in figure 5.2, as well as the associated marginal cost t^* of achieving this reduction. If the total reduction required by decision-makers in country A is determined with international cost-effectiveness in mind, it equals the total reduction for country A in figure 4[18]. Cost-effectiveness within country A requires that the marginal costs for measures 1 and 2 are equal. If they are not, costs could be saved by reducing the level of the measure with the higher marginal cost and increasing the level of the measure with the lower marginal cost. The cost-effective abatement by each of the two measures is shown in the figure. As can be seen from the figure, more of measure 2 is used, which is explained by fact that the marginal cost curve for this measure increases more slowly.

[17] This is the marginal cost curve of country A in figure 4 above.

[18] However, even if country A would ignore international cost-effectiveness and choose another national target, it is still possible that the *national* reductions could be achieve in a cost-effective way.

Figure 5.2 Nationally cost-effective reductions of nutrient loads to basin x.



5.3 Markets based instruments or command and control?

Economists have spent large efforts on analyzing different policy instruments and their pros and cons. A number of different properties are asked from policy instruments, policy makers are likely to wish that

- the instrument reaches the target in a least-cost way,
- targets are met with certainty,
- the instrument provides incentives for development of environmental technology,
- the instrument is flexible with regard to changes in the economic environment and
- the distributional effects are acceptable.

In the general case, market-based instruments such as environmental taxes and tradable emission permits will lead to a cost-effective achievement of environmental targets, while command-and-control instruments will not. The reason is that taxes and tradable emission permits will give market incentives for low-cost polluters to abate more and high-cost polluters to abate less. With command-and-control, i.e. when the emission level or technology is regulated for each polluter, the allocation of abatement will not be cost-effective unless the regulating agency knows the abatement costs for each single polluter, which is normally not the case.

Command-and-control directed towards emissions as well as tradable emission permits will in the general case lead to target achievement with high accuracy as the total level of emissions is directly controlled by the policymaker. With environmental taxes, policymakers might misjudge the aggregate cost function whereby emissions are reduced either too much or too little compared to the target. However, this problem could, in principle, be solved through a change in the tax level.

If it is made costly to emit, incentives are created for polluters and others to develop low-cost technologies to reduce emissions. With a command-and-control system, these incentives are smaller than with market-based instruments. Moreover, the incentives are larger with an environmental tax than with tradable emission permits where the permits are distributed at zero cost to the polluters because the cost to the polluting sector is larger when polluters have to pay for both abatement and the remaining emissions. It is often argued that market-based instruments are flexible with regard to changes in the economic environment as less administrative effort might be required to adjust these measures. Thus, there is to some extent a trade-off between flexibility and a reliable target achievement. Increases in environmental taxes can on the other hand meet considerable political resistance due to the higher costs to polluters, implying that it can be difficult to reach a political agreement about setting taxes on the economically optimal level. Both direct controls and tradable emission permits, where permits are distributed to the polluters at zero cost, are likely to meet less resistance.

The distributional impact of policy instruments is important to policy-makers. This impact is determined by the costs that different parties incur. Here, command-and-control is attractive to polluters because they only need to pay for actual abatement. Although total abatement costs become lower with taxes than with command-and-control, this is more than outweighed by the costs for tax payments, and this is a reason why command-and-control is usually preferred by polluters. The costs to the polluters are identical under environmental taxes and tradable emission permits, provided that emission permits are distributed through auctions. Often, however, emission permits are given away for free according to the so-called grandfathering principle, where each polluter gets emission permits proportional to his historical emissions. If so, tradable emission permits are less costly to the polluting sector than environmental taxes. One should note, however, that environmental taxes or permit auctions would lead to an extra income to the government, which can be redistributed to the citizens in a way that reduces income distribution problems.

For many types of environmental policy instruments, costs are distributed regressively, i.e. low-income groups pay a larger share of their income for these policies. This is shown to be the case e.g. for the Swedish carbon dioxide tax (Kriström et al., 2003). In addition, this tax affects households in rural regions more negatively than households in urban areas. For policies against eutrophication, no similar evaluations have been made and considering the large flora of subsidies to abatement, the distribution of costs cannot be judged without a thorough investigation. That issue is, however, outside the scope of this report.

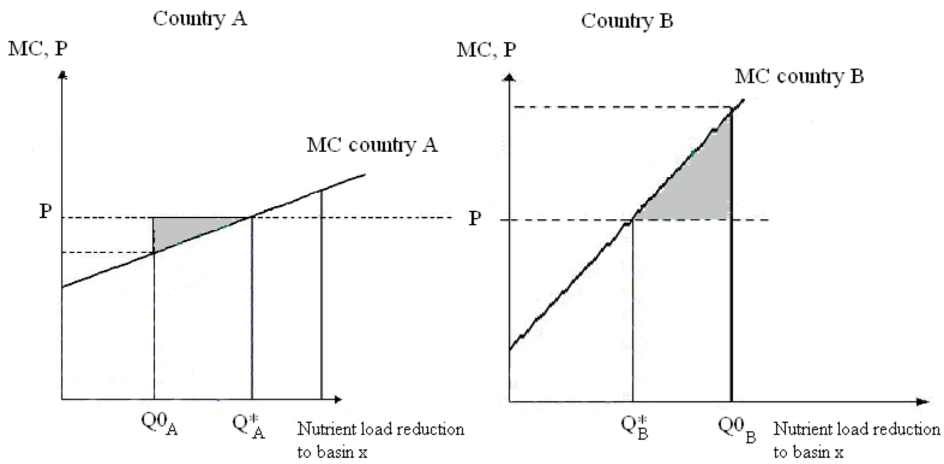
5.4 The gains from emission trading

Emission trading has been put forward by e.g. NEFCO (2008) as a potential way to achieve international cost-effectiveness with regard to nutrient reductions to the Baltic Sea. The positive experiences of emission trading for energy intensive industries in EU (see e.g. Ellerman and Buchner, 2007) have added fuel to this debate. In the brief exposition of the gains from emission trading made in sections 5.4 and 5.5, there is no distinction made between emission and load trading, implying that it is implicitly assumed that retention is the same in all countries and regions.

Emission trading can give rise to gains for both the buyer and the seller. Consider the two countries *A* and *B* discussed above. Suppose that they have been allocated abatement burdens in a way that is not cost-effective, e.g. as is the case with the allowed nutrient loads for the countries surrounding the Baltic Sea. The countries are given emission permits corresponding to the difference between initial emissions and the allocated abatement burden. They can choose either to meet the abatement burden in the home country or to buy or sell emission permits in the market.

Assume that the *MC* curve of country *B* is steeper than that of country *A*, and $Q0_A$ and $Q0_B$ are the initial abatement burden placed on country *A* and *B*, respectively. The situation is illustrated in figure 5.3.

Figure 5.3 Gains from emission trading between two countries.



On the left hand side of the graph is the *MC* curve for country *A*. $Q0_A$ is the amount of reductions required for country *A*, but at $Q0_A$ the MC curve has not intersected the market permit price *P*. Thus, given this permit price, country *A* would find it profitable to abate more and sell the excess permits to country *B*.

On the right hand side is the *MC* curve for country *B*. $Q0_B$ is the amount of reductions required for country *B*, but the *MC* curve already intersects the market price of CO_2 permits before $Q0_B$ has been reached. Thus, given the market permit price of CO_2, country *B* would profit from abating less than its abatement burden and instead buy emission permits.

In this example, country *B* would abate emissions until its *MC*-curve intersects with *P* (at $Q_B{}^*$) and, in order to comply with the international agreement, buy emissions permits from country *A* at the price *P*. Country *B*'s actual abatement plus the permits bought from country *A* sum up to the total required reductions for country B, $Q0_B$. Country B saves costs equal to the shaded area in the figure. This area represents the additional cost that country B would have had if it abated all of its required emissions by itself.

Country A makes a profit by abating more emissions than initially required and selling permits to country B. Its net benefits from selling emission permits equal the shaded area in the left figure. Thus both countries gain from emission trading. This is explained by the countries sharing the total cost reduction that arises when abatement is allocated in a cost-effective manner.

5.5 The location of the source matters

Under the BSAP targets, one kilo of a nutrient reaching a particular marine basin is presupposed to have the same impact on the environmental status of that marine basin. For example, the environmental damage is supposedly the same no matter whether a kilo of nitrogen reaching the Baltic Proper originates from Sweden or Poland. In such a situation, a uniform nutrient load tax applied in all countries discharging into a particular basin would generate the cost-effective allocation of abatement efforts. Similarly, if a load permit system would be applied, loads to one particular marine basin could be traded at a one-to-one ratio. However, the situation is different if one considers policy instruments applied at the sources. The reason is that the location of the sources matters for the impact on coastal load.

Everything else equal, measures located close to the coastal zone have a larger impact on the sea than measures undertaken in the inland. Moreover, the impact on the sea can differ depending on where in the Baltic Sea drainage basin the measure is undertaken because nutrient transports vary between regions due to differences in e.g. climate and soil types. A cost-effective policy instrument must take this variation into account. Thus, a uniform tax on emissions at the sources or one-to-one trading of emission permits will not lead to a cost-effective outcome. Instead, differentiated policies can reduce the costs of target achievement. A policy instrument that takes into account every difference in impact between different measures would, on the other hand, be extremely costly to administer. Hence, it is necessary to make a trade-off between precision and administrative costs.

In Sweden, some policy instruments are regionally differentiated, while others are not. Within the Rural Development Program only farmers in certain regions, mainly coastal regions in southern Sweden, are eligible for support to catch crops and spring plowing. Yet, those participating in the program all get the same level of support. For buffer strips similar rules apply; farmer in southern Sweden are all eligible for support, and compensation is equal for everyone[19]. Also, the support to wetland construction and maintenance is regionally differentiated. Swedish wastewater treatment plants along the coast from Norway up to Åland Sea have been regulated more stringently with regard to nitrogen emissions than have plants in northern Sweden. Larger plants in the inland of southern Sweden have been regulated according to the same principles as plants along the coast. Measures taken in Northern Sweden can, however, be cost-effective if the target is to improve water quality in the Baltic Proper, because the negative effect of the long distance between the source and the recipient is outweighed by the low cost and high impact on coastal load for some measures (Elofsson and Gren, 2004). Policies for regulation of wastewater treatment by households not connected to wastewater treatment plants are not regionally differentiated. Neither are fertilizer taxes and policy instruments directed towards airborne emissions. Notably, abstaining from differentiation does not necessarily imply large efficiency losses. Brännlund and Gren (1999) show e.g. that a uniform Swedish tax on nitrogen fertilizers does not imply large efficiency losses compared to a regionally differentiated tax because there is a negative correlation between the environmental impact in a region and the costs of fertilizer reductions.

At the international level, HELCOM often issues uniform policy recommendations for the whole Baltic Sea region. One example is that HELCOM (2007d) recommends a general replacement of phosphate detergents by phosphate-free ones in all Baltic Sea countries. Gren (2008) suggests that such a uniform policy is associated with positive net benefits everywhere in the drainage basin. However, HELCOM (2007e) also recommends measures to be taken towards all rural households not connected to wastewater treatment. Compilations by Gren (2008) suggest that this will not result in positive net benefits in any part of the

[19] Also inland farmers are eligible for this support, one motive is the effect of phosphorus on inland eutrophication.

drainage basin. If this holds then, from an economic point of view, the measure should not be undertaken anywhere in the region. For most types of measures, regionally differentiated policies are preferable (Gren, 2008). Thus, stringent uniform regulations can be cost-effective for low-cost measures with a high impact on the sea, but for measures with higher costs or lower effect, potential cost savings from differentiation throughout the Baltic Sea drainage basin need to be considered. High cost measures with low effect should only be used if the target is very stringent and no other abatement options are available.

5.6 The role of economic and ecologic uncertainty

With large scale environmental problems, policy decisions are typically taken under uncertainty as there is no perfect scientific knowledge about all the relevant natural processes that affect the environmental outcome. Neither is it possible to know the value that people attach to environmental improvements, now or in the future, with certainty. Therefore, the benefits of abatement are uncertain. In addition, abatement costs now and in the future are not known perfectly, e.g. because the potential for adaption of production processes and for technological development is unknown and different agents may find it in their interest not to reveal their abatement costs.

Already in the 1970s it was shown that uncertainty about the benefits of abatement alone does not matter to the choice of policy instrument[20] (Weitzman, 1974). Regardless of whether a price- or quantity-based instrument is used[21] there will be inefficiencies when benefits are misjudged, but the efficiency loss is the same independently of the instrument chosen. If abatement costs are uncertain, this affects the optimal choice of instrument. If marginal benefits of abatement decrease rapidly, while marginal abatement costs increase slowly, then a system with quantity control, such as emission permit trading, would be associated with a smaller expected efficiency loss. An environmental tax would be better if marginal abatement costs are increasing rapidly but marginal

[20] Weitzman's results are derived for linear marginal costs and marginal benefits of abatement. The implications of uncertainty when these functions are nonlinear are not fully known.

[21] I.e. an environmental tax or a system for emission permit trading.

benefits of abatement decrease slowly. Later research has shown that these results hold as long as uncertainty about abatement costs and environmental damage is uncorrelated (Stavins, 1996). Thus, uncertainty about the damages from anthropogenic nutrient emissions to the Baltic Sea[22] alone may not matter for the optimal choice between price- or quantity-based policy instruments unless it can be shown that this uncertainty is linked to the uncertainty about costs. Yet it can matter to the design of the chosen policy instrument, which is further discussed below.

Uncertainty regarding the future costs of abatement, explained e.g. by lack of knowledge regarding future development of the agricultural and the transport sectors, can matter for the choice of policy instrument. With the knowledge available at this date, it seems as if the marginal cost function is steeper than the marginal benefit function for nitrogen reductions to the Baltic Sea[23] and hence, a price instrument would be associated with smaller efficiency losses compared to a quantity control, if there is uncertainty about costs. Yet, in an international setting where there is no governmental body to collect the tax and ensure compliance, a tradable emission permit system can have advantages from an institutional perspective.

In some cases, the damage from pollution might not increase continuously with the level of emissions. Instead, there may be a threshold, where the damage suddenly jumps upward to a higher level. If policy makers know that the damage function has this property, but are uncertain about the costs of emission reductions, then economic theory advocates a combination of an environmental tax and a quantity standard (see e.g. Roberts and Spence, 1976). Then, if abatement costs turn out to be lower than expected, the environmental tax will lead to optimal emission reductions in a cost-effective manner, just as in the standard case described by Weitzman (1974). If costs are high, the quantity standard will ensure that emissions do not exceed the threshold level. It is often argued by some researchers that the Baltic Sea might have "flipped", i.e. passed a threshold level, where the sea becomes much

[22] E.g. due to limited knowledge about the marine response to changes in nutrient loadings and the storage of nutrients in soils, groundwater and sea bottoms, see EAC (2005).

[23] A nitrogen abatement cost function calculated by Gren (2008) suggests that the elasticity of the cost function with respect to nitrogen load reductions supply is just below 1.0 when total nitrogen load reductions are increased from 40 to 50 %. This implies that marginal costs increase at a moderate pace. Hökby and Söderqvist (2003) estimate that the price elasticity of demand for nitrogen reductions is -1.86, i.e. demand is relatively elastic, implying a slow decrease in marginal benefits.

more difficult to restore than if this threshold has not been passed (EAC, 2005, Österblom et al., 2007). If this holds, then some kind of combined tax-quantity approach might be motivated[24].

As was noted in the foregoing section, the impact of nutrient emissions on coastal loads depends on the location of the source. In addition, the impact on coastal loads is more uncertain for some types of sources than for others. In both cases economic theory suggests the use of "exchange rates" between sources when emission trading is applied, implying that reductions at one source are valued more highly than reductions at another (see e.g. Malik et al., 1993; Hoag and Hughes-Popp, 1997). For example, reductions at a source where the effect is uncertain might not be valued equally high as at a source with a certain effect[25]. The degree of differentiation has to be weighed against the transaction costs associated with instrument design and enforcement.

In general, polluters know the costs of abatement but policy makers may not. This situation is one with so-called asymmetric information. In such cases, economic theory often advices the use of contracts, where low and high cost polluters each are encouraged to choose a contract from a menu provided by the policy maker. A contract defines the actions required by the polluter and the associated compensation for these actions. With a well designed menu of contracts, the policy makers will obtain the maximum environmental effect from a given budget (see e.g. Salanié, 1997).

If there is both asymmetric information and uncertainty about the environmental damage from pollution, such that only the combined effect of from all polluting sources can be observed, then rewards for environmental quality above a given standard in combination with penalties for substandard quality might solve the problem for smaller recipients (Segerson, 1988). For large water bodies such as the Baltic Sea however, policy instruments that will solve this type of problem have not yet been envisaged.

[24] If the Baltic Sea has passed already passed such a threshold, i.e. if the ecosystem has "flipped", then it might be also be necessary to re-evaluate whether the benefits from nutrient load reduction outweigh the costs of reductions, as the benefits from reductions will be smaller for a "flipped" ecosystem. With a flipped ecosystem, the same policy instruments can be applied but the stringency will have to be increased.

[25] One drawback with the "exchange rate" approach, is that with fixed exchange rates, the final effect on nutrient loads will depend on the initial distribution of emission permits (Försund and Naevdal, 1998). This will no longer be a problem if exchange rates are successively adjusted with regard to the marginal impact of measures on the target.

5.7 Strategic decisions in the international arena

The organizational structure of governments is important for policy implementation. This section discusses the incentives for international cooperation, how policy instruments may affect these incentives and whether a single country should act as forerunner with regard to abatement.

Policy solutions at the international level are not easily achieved because countries with opposite interests must come to an agreement. In addition, there are usually no appropriate enforcement agencies at the international level that can ensure that all countries carry out what they have promised to do. The underlying problem at the international level is the risk of free-riding. Free-riding may occur when a single country is better off doing less at home than what is in the collective interest while still benefitting from other countries' abatement efforts. For the Baltic Sea, free-riding is a highly relevant problem, as loads from one country affect all other countries. Consequently, the Swedish Commission of the Marine Environment complains that there is a large implementation deficit with regard to many of the international agreements made for the Baltic Sea since 1972 (CME, 2003). The institutional structure of governance in the Baltic Sea region could change, now that all riparian countries except Russia are members of the European Union. The guidelines for the Marine Strategy Directive (EC, 2008b), however, suggest that the achievement of marine targets will be the responsibility of the countries in the region concerned.

A couple of studies (Gren, Elofsson and Jannke, 1997; Ollikainen and Honkatukia, 2001) compare the costs for different countries of participation in a cost-effective agreement with the costs of uniform, proportional reductions[26]. Gren (2008) compares the net benefits for the cost-effective solution with the BSAP basin targets to the "command-and control" solution through catchment targets. Gren (2001) investigates the countries' optimal choice of nutrient reductions when free-riding is possible and compares the net benefits to countries under free-riding and cooperation, respectively. Although different types of comparisons are made in these studies a common pattern emerges. The studies point out

[26] The comparison with uniform, proportional reduction is motivated, considering that the earlier HELCOM target was interpreted as a 50 % reduction in nitrogen and phosphorus loads from each country.

Poland, Latvia and Lithuania as countries that are not likely to prefer a cost-effective or cooperative solution unless other countries compensate them, see table 5.1. Sweden and Finland are the main winners from cooperation on Baltic-wide nutrient reductions, as long as no compensations are made between countries.

Table 5.1 Losers under different schemes for cost-effective or optimal international cooperation on nutrient reductions[1,2]

	Nitrogen			Phosphorus		Nitrogen and phosphorus
	Gren, Elofsson and Jannke (1997)	Ollikainen and Honkatukia (2001)	Gren (2001)	Gren, Elofsson and Jannke (1997)	Ollikainen and Honkatukia (2001)	Gren (2008)
Sweden						
Denmark		X		X	X	
Germany				X		
Poland	X	X	X	X	X	X
Estonia	X		X			
Latvia	X	X	X	X		X
Lithuania	X	X	X	X	X	X
Russia	X		X	X	X	
Finland						

[1] Losers are marked with X.
[2] Cost-effective or optimal schemes are compared to either uniform, proportional reductions or the outcome without international cooperation.
Source: Elofsson (2008)

There are two reasons why some countries lose under cost-effective or optimal international agreements compared to a scheme with proportional reductions or without international cooperation. The first is if a country has a large pool of low-cost measures and thus undertakes large reductions when abatement is allocated in a cost-effective way. This cost-effective reduction will be more expensive to the country in question than if abatement was proportional to loads. Second, the citizens of a country might have a low willingness to pay for environmental improvements, explained by e.g. low income and/or a low use of the Baltic Sea. In Poland for example, the Baltic Sea coastline is short and most of the population lives at a large distance from the coast, which might explain a lower concern by Polish citizens for the Baltic Sea compared to e.g. Swedish citizens. The explanation for some countries gaining from cost-effective or optimal international agreements is simply a reversal of the above arguments.

Thus, side payments seem to be a necessary condition for internationally cost-effective reductions to be carried out. Alternatively, another possibility to solve this redistribution problem is to introduce emission permit trade across the borders. With emission permit trade, low-cost polluters can sell their excess permits to high-cost polluters and hence make a profit. Thus, if a country is given a large share of the total permits, it could make a net gain.

International finance institutions, such as e.g. NEFCO and the European Investment Bank (EIB)[27], contribute to some extent to an international redistribution of abatement costs. These financing institutions, however, do not provide the financing countries with equally strong incentives as would a system with tradable emission permits. The motives for the financing countries are weaker e.g. because contributions to the financing institutions are not taken into account when evaluating the achievement of the financing country's own national target.

Nutrient reductions contribute to environmental improvements both for inland and marine waters. If phosphorus reductions are mainly of national and nitrogen of international concern, parallel international and local markets for tradable emissions permits might solve the environmental problems cost-effectively even if countries act in their self-interest (Caplan and Silva, 2005). This requires, however, an international financial institution that solves the income distribution problem between countries through so-called lump-sum transfers.

Belarus and Ukraine contribute to waterborne loads of nutrients and several European countries contribute to the airborne loads of nitrogen. These countries have not committed to reductions with the purpose to improve Baltic Sea water quality and have small incentives to do so, as the benefits to these countries are likely to be small. Under the Kyoto protocol, there is a corresponding problem as developing countries have not undertaken abatement commitments. To solve this issue, the Protocol prescribes the use

[27] The European Investment Bank is the European Union's long-term lending institution that finances e.g. projects that protect and improve the environment and promote social well-being in the interests of sustainable development.

of the Clean Development Mechanism (CDM). CDM is an arrangement allowing industrialized countries with a greenhouse gas reduction commitment to invest in projects that reduce emissions in developing countries as an alternative to more expensive emission reductions in their own countries. A crucial condition is that it can be established that the planned reductions would not occur without the additional support from developed countries. In this way developed countries can lower the costs of complying with their Kyoto targets by investing in greenhouse gas reductions in a developing country where reductions are cheaper, and then apply the credit for those reductions towards their own commitment goal.

In order to speed up international action, unilateral abatement is sometimes suggested as some believe that it might make other countries follow. In Sweden, unilateral action with regard to nutrient pollution of the Baltic Sea has been suggested by several parties (RD, 1998, 2000 a,b). There are several empirical examples of unilateral action that have not led to the intended effects (Barrett, 1994). In addition, the economic literature provides little or no support for unilateral action. If one country is altruistic, i.e. has a selfless concern for the welfare of others, unilateral behavior is not worth while because it actually stimulates the followers to pollute more, annihilating the whole environmental improvement (Hoel, 1991). Cost uncertainty could hypothetically lead to a win-win situation if one country acts before another and this action reduces uncertainty about abatement costs. However, there are small incentives for countries too choose such strategies in reality if each country acts in its own interest. One reason is that the country that abates first will abate less because of uncertainty about the follower's behavior and hence, a larger share of the abatement burden will fall on the follower country. Therefore, the follower country will be unwilling to wait for the first country to take on such a leader role (Elofsson, 2007). Instead, economic research has shown that so-called issue linkage is a common way to solve international distribution problems that arise from cost-effective environmental policies. Through such issue linkage, disadvantages to a country in one policy field can be compensated for by advantages obtained in another field (Carraro and Siniscalco, 1998).

5.8 The national versus the local dimension in water management

At the national level, policy implementation is carried out by multiple governments in different sectors and at different levels (see e.g. Lundqvist, 2004). The multitude of governments involved leads to difficulties to coordinate policies in order to achieve cost-effectiveness and it opens up for strategic considerations from decision-makers that represent different interests (Miceli and Segerson, 1999). For example, enforcement of nationally decided policies might be weakened when local governments are responsible for implementation, because local governments are elected to represent local interests, which may be in conflict with national interests (Eckerberg, 1997; Michaelowa, 1998). Lack of policy coordination between governments may in itself be a cause of socially inefficient outcomes e.g. if each government takes into account only the own costs or have a different view on the benefits of environmental improvements or if coordination is associated with large transaction costs.

National governments often impose environmental obligations on local governments for which the local government is or is not reimbursed. There are several motives for reimbursement through intergovernmental grants. One of the strongest is that the local government might not take into account interregional spillover effects (see e.g. Bradford and Oates, 1971). Then an intergovernmental grant could provide the local government with incentives to increase abatement to the socially optimal level. There is a trade-off between fully unfunded mandates on one hand, where the central government has incentives to decide on too stringent regulations, and fully funded mandates on the other hand, where the local government does not have any incentives to find the least cost measures (Miceli and Segerson, 1999). Threshold rules for central government funding of local government abatement mandates, which imply that the central government must pay a fraction of the cost if the total cost of a regulation exceeds a certain limit, might solve this problem and lead to efficient outcomes in some settings.

EU's new Water Framework Directive is currently being implemented and if its intentions are followed, it will lead to a strengthening of the regional governmental level for water management. However, at least in Sweden the decision rights on policy instruments have so far not been delegated to the so-called Competent Authorities, i.e. the regional authorities in charge of implementing the Directive. Consequently, the Swedish Competent Authorities complain that the rights and responsibilities of the Competent Authorities as well as those of local and national governments is unclear and that the policy instruments as well as the right to decide on those are in need of revision if water management programs are to be enforced (CA, 2007). Thus, lack of clarity regarding decision-rights may hinder the enforcement of nationally determined targets under the Water Framework Directive.

5.9 Transaction costs

Transaction costs can be a significant part of the costs for environmental policies, and in particular, this is likely to be the case for nonpoint source pollution. Therefore, consideration of transaction costs is important when evaluating policies for the Baltic Sea. Transaction costs are e.g.

- research, information and meeting costs,
- enactment and lobbying costs,
- design and implementation costs and
- administration, monitoring and prosecution costs.

Most of these costs are costs of labor time for researchers, court staff, legislators, government staff and stakeholders (McCann et al., 2005).

While all policy instruments may have transaction costs, they are relatively large for command-and-control policy instruments. A large information burden is here placed on the government that should identify costs and effects of measures for each individual polluter. Empirical evidence suggests that transaction costs can also be a considerable obstacle to permit trading when these costs are not been taken into account when designing the system (see e.g. Stavins, 1995). Given substantial transaction costs, the initial allocation of emission permits matters not only for political reasons but also for efficiency, as the initial allocation determines the amount of trade.

In a study applied to reductions of phosphorus in the Minnesota River, McCann and Easter (1999) show that the total transactions costs for the phosphate fertilizer tax is the lowest among the instruments compared. Transaction costs are successively higher for; educational programs on best management practices, a requirement for conservation tillage on all cropped land and an expansion of a permanent conservation easement program. They conclude that taking account of transaction costs does not change the efficiency ranking of command-and-control instruments versus environmental taxes and note that the size of transaction costs is likely to be determined by the type of environmental problem and the design of the policy instrument.

In an investigation of the transaction costs under the Kyoto Protocol, Michaelowa et al. (2003) conclude that the project-based Clean Development Mechanism (CDM) and Joint Implementation[28] (JI) are associated with considerable transaction costs, e.g. for the development of baselines to which the effect of the projects are to be compared and for control and certification of the projects. In order to reduce these transaction costs they suggest joint evaluation of bundles of projects instead of individual projects, less frequent control and certification, development of general guidelines for evaluation and a reduction of the number of parties involved in each project.

5.10 Summary

This chapter reviews the basics about cost-effective environmental policy. In addition it briefly summarizes the vast research on the implementation of international environmental policies and on the use of policy instruments when nonpoint sources contribute to pollution. The chapter shows that

- International cost-effectiveness requires that marginal costs for reducing loads of a nutrient to a particular marine basin are equal for all countries. Hence, the cost-effective

[28] Joint implementation (JI) allows industrialized countries to invest in emission reduction projects (referred to as "Joint Implementation Projects") in any other industrialized country as an alternative to reducing emissions domestically. In this way countries can lower the costs of complying with their Kyoto targets by investing in greenhouse gas reductions in an industrialized country where reductions are cheaper, and then applying the credit for those reductions towards their commitment goal.

allocation of abatement depends on the countries' relative costs of reducing coastal loads.

- At the national level, the cost-effective abatement strategy is determined by differences between measures with regard to abatement costs and impact on coastal loads.
- In the standard case, environmental taxes and emission permit trade will attain a load target at minimum cost for society as a whole, while command-and-control instruments will not.
- The impact of measures against nutrient pollution depends on the location of the source and for some measures, the impact is more uncertain than for others. This has implications for the design of policy instruments. A differentiation of environmental taxes between different parts of the Baltic Sea and between point and nonpoint sources could reduce the costs of nutrient load reductions. Similarly, if tradable emission permits are used, the introduction of "exchange rates" between regions and source types might reduce costs of meeting environmental targets if a decentralized emission trading scheme is applied. The efficiency gains from differentiation, however, must be weighed against the higher costs associated with the design and enforcement of a differentiated policy.
- The design and use of policy instruments is associated with transaction costs. Environmental taxes are associated with lower transaction costs than command-and-control measures and thus, this is another argument in favor of environmental taxes. Experience has shown that emission permit trade as well as the so-called Clean Development Mechanism and Joint Implementation can be associated with relatively high transaction costs. These transaction costs might, however, be reduced through the choice of instrument design.
- In spite of the international agreements for nutrient load reductions that were set up already in the 1980s, targets have not been met in the countries surrounding the Baltic Sea. Several countries in the region may not gain from an implementation of the agreements, which can explain some of the failure to meet targets hitherto. Although international financing institutions relieve some of this problem through a redistribution of capital for environmental investment to countries with low abatement

costs, economic theory suggest that international emission permit trade might solve this problem in a more efficient manner.

- Within the EU countries, inland water quality management is governed by the Water Framework Directive, which assigns the responsibility for water management to regional Competent Authorities. However, at least in Sweden much of the decision-right on policy instruments still lies with national and local authorities. The multiplicity of governmental levels and branches involved in decision-making for water management opens up for decisions based on local or sectoral self-interest, which can hamper the implementation of internationally agreed nutrient reductions.
- Belarus and Ukraine contribute to nutrient loads but have not committed to any nutrient abatement. In order to make use of low-cost abatement options in these countries, instruments akin to the so-called Clean Development Mechanism under the Kyoto Protocol might reduce the costs of meeting the targets for the sea. Hence, such instruments are called for.

6 Model and data

In order to assess the costs and effects of different nutrient policies, a cost-effectiveness model is used in the two following chapters, chapter 7 and 8. This model is described with regard to structure and data in the following. For the interested reader, a formal description of the model is provided in Appendix A.

The model includes abatement measures in the countries surrounding the Baltic Sea. The Baltic Sea drainage basin is divided into 24 different regions, while the sea itself is divided into seven different basins[29], see figure 6.1.

In each region, there are 14 measures to reduce nitrogen and 8 to reduce phosphorous, see table 6.1. For nitrogen, eight measures are in the agricultural sector, three are measures to reduce nitrogen oxide emissions and two are directed towards wastewater treatment plants. For phosphorus, five measures are in the agricultural sector while three are related to wastewater treatment. Data on measures' costs, impact on nutrient loads to the Baltic Sea and their potential are obtained from Gren et al. (2008) for all catchments draining to the Baltic Sea[30]. Costs and effects for measures in the Swedish Skagerrak catchment are assumed equal to those in the Kattegat basin, and capacities for the Skagerrak basin have been constructed in a similar manner as for the other basins.

[29] This division into basins coincides with the one used in the NEST (see e.g Savchuk, 2006).

[30] The interested reader can find all necessary data that have originally been used for the calculation of cost and effects in Gren et al. (2008). In contrast to Gren et al. (2008), the model employed here does not account for interdependencies with regard to the impact of different measures on coastal load. This means that in principle, costs are underestimated. However, a comparison with results in Gren (2008) suggests that this is of small importance for the results.

Figure 6.1 The Baltic Sea drainage basin. In the model, Estonia, Latvia and Germany are further divided into 3, 2 and 2 catchments, respectively, (Source: Elofsson (2006).

Cost functions for emission reductions are in most cases assumed to be linear, implying that the unit cost of reduction is constant. The costs are either so-called engineering costs, i.e. based on investment, management and operation costs, or estimated using a partial equilibrium framework. Costs for reductions in airborne emissions by selective catalytic reduction (SCR), a change of spreading time for manure, increased cleaning at sewage treatment plants, wetland construction and private sewers are calculated as annual costs, based on investment costs. Costs of reducing livestock holdings are the associated loss in short-run profits to farmers when abstaining from livestock production.

Table 6.1 Abatement measures in the model[1,2].

	Impact on nitrogen emissions	Impact on phosphorus emissions
Energy sector:		
Selective catalytic reduction (SCR) on power plants	X	
Transport sector:		
SCR on ships	X	
SCR on trucks	X	
Wastewater treatment:		
Increased cleaning at sewage treatment plants	X	X
Private sewers	X	X
P free detergents		X
Agricultural sector:		
Reductions in cattle, pigs, and poultry[2]	X	X
Fertilizer reduction	X	X
Catch crops	X	X
Energy forestry	X	
Grassland	X	
Creation of wetlands	X	X
Changed spreading time of manure	X	
Buffer strips		X

[1] It is sometimes argued that most measures in the agricultural sector affect both nutrients. However, the model does not include estimates of such double effects for all measures.

[2] Thus, there are three different ways to reduce livestock.

Costs for phosphorus-free detergents are the increased cost of production, compared to conventional detergents. For reductions in fertilizer use, the cost is the reduction in profit[31]. For fertilizer reductions, the unit cost of reductions is increasing. Costs are expressed in 2007 price level.

A measure's impact on coastal load is determined by a constant emission coefficient. This coefficient varies between measures and regions[32]. Thus, the scale of an abatement project does not affect

[31] Calculations are based on econometrically estimated demand functions, see Gren et al. (2008). For the cost functions, a constant elasticity of demand is assumed, and the loss in consumer surplus is calculated through integration of the demand curve minus the price.
[32] The emission coefficient has been obtained through division of the cost at the sources by the marginal cost of reductions in nutrient loads to coastal waters. These data are obtained from the Appendix in Gren (2008).

the marginal impact on coastal load. The data originally used by Gren et al. (2008) for calculation of these effects have been compiled from a large number of different sources, as a coherent database on the effects of different measures on the loads to the Baltic Sea is not available.

Data in the model have, originally, been collected from a large number of sources. Other estimates of costs, impact and capacities are also available, that might differ from those used in the model. Thus, there is no strong scientific consensus on the appropriate level of the data used. Likewise, there is no available compilation of the data on cost and effects of different measures used in different studies[33]. Yet, as shown in Elofsson (2008) the relative costs of different types of measures are rather robust across studies. Still, there is some uncertainty about costs, effects of measures on coastal loads and capacities of different measures. With another set of data, the results can, of course, be different from those presented here. However, given the large set of measures included and the spatial disaggregation, the results are relatively robust to changes in individual parameters. It should be recognized that no better data collection is currently available. Moreover, it deserves mentioning that implementation costs associated with the use of different policy instruments, in excess of the direct cost for the physical measures, are not included. As discussed in chapter 5, these aspects are of importance for policy decisions regarding eutrophication.

Summarizing, the model in this report has its main value in illustrating important mechanisms and relationships and in providing an indication of the outcome in terms of costs and measures and the distribution of those. When used as a foundation for policy decision it should, if possible, be compared to the results from other models.

[33] Several studies covering the Baltic Sea are discussed in the introductory section.

7 International cost-effective fulfillment of the BSAP targets

The cost-effective solutions to the BSAP nutrient targets are here analyzed from an international perspective. The BSAP targets for marine basins could, in principle, be met in numerous ways, for example

1. in a cost efficient manner such that reductions of nutrient loads *to each basin* is carried out at minimum cost or
2. through "command-and-control" such that reductions in nutrient loads *from each country to each basin* are made at minimum cost.

These targets are in the following called the basin and the catchment targets, respectively. The BSAP agreement is based on the latter concept in 1 above. In order to highlight the difference between the cost-effective solutions to 1 and 2 above, the targets are compared with regard to total costs and total abatement as well as the distribution of those.

To make this comparison possible it has been necessary to make some adjustments compared to the original BSAP targets included in table 3.2 and 3.3. The catchment targets for the Swedish Kattegat and the Russian, displayed in table 3.3, are first adjusted downwards by 10 and 45 %, respectively. The reason is that there is not sufficient capacity in the model to reach the catchment targets in these regions. Then, the basin targets are obtained by summation of the reductions required for the relevant catchments. These adjustments imply that the total target for each basin is identical under these modified basin and catchment targets.[34] If they were

[34] The adjustments made implies that the total nitrogen reduction target is lowered by less than 1% compared to the BSAP target and that the total phosphorus reduction target is lowered by less than 3%. It is assumed that these adjustments have only a negligible impact on the results. We do not, therefore, distinguish between these modified targets and the original BSAP targets.

not identical, the costs of meeting the targets could not be compared in a consistent manner.

The chapter is organized as follows: in section 7.1 the reductions of nutrient loads to coastal waters that result from a cost-effective achievement are presented. It is shown that the BSAP load targets imply water transparency improvements beyond the objective of the BSAP, even if reductions are carried out in a cost-effective manner. In section 7.2, the cost-effective distribution of abatement over the countries is presented and it is noted that the distribution of abatement differs considerably from the reductions implied by the catchment targets in the BSAP agreement. The minimum cost associated with load reductions are presented in section 7.3, and it is concluded that if only costs are taken into account some countries will prefer the BSAP agreement on reductions from different countries to a cost-effective allocation of reduction efforts which meets the target for different marine basins. In section 7.4, potential improvements in the distribution of the abatement burden are discussed, based on the marginal cost of nutrient reductions. In section 7.5 there is a discussion of the gains from load trading to different countries under various assumptions about the initial distribution of emission permits. This analysis shows that load trading where permits are initially distributed according to the BSAP target will reduce costs to all countries compared to if all countries comply with BSAP targets using domestic measures only. Poland, however, will make minor gains. The chapter is summarized in section 7.6.

7.1 Joint reductions of nitrogen and phosphorus

The BSAP agreement requires that certain reductions to each basin are achieved. One first question is then how the target formulation affects the achievement of these reductions. In table 7.1, it is shown that no matter whether basin or catchment targets are used, total nitrogen reductions will be larger than required by the international agreement. The reason is that in some cases, the phosphorus target is so demanding and many measures that reduce phosphorus also reduce nitrogen, the nitrogen target will automatically be achieved or even over-achieved[35]. This case is relevant for some basins under the basin target and for some

[35] Thus, in such cases only the phosphorus target is binding.

catchments under the catchment target. A further look at the results shows that one important reason for this outcome is the stringency of the phosphorus target for the Baltic Proper. With catchment targets, nitrogen loads will be lower than required by the agreement in both Baltic Proper and the Gulf of Finland. For phosphorus, there will not be any substantial over-achievement under a cost-effective attainment of the basin targets, except for the Danish Straits and Kattegat where phosphorus reductions will be made in “excess”, see table 7.2.

Table 7.1 Reductions in coastal load of nitrogen to different basins

	Targets		Cost-effective solutions			
	Basin target, ton N	Basin target, %	Basin target, ton N	Basin target, %	Catchment target, ton N	Catchment target, %
BB	0	0	0	0	0	0
BS	0	0	0	0	0	0
BP	89,569	69	103,988	72	124,147	76
GF	5,983	5	5,983	4	6,219	4
GR	0	0	150	0	1	0
DS	15,109	12	15,109	11	15,109	9
KA	18,296	14	18,296	13	18,296	11
Sum	**128,957**	**100**	**143,526**	**100**	**163,616**	**100**

Table 7.2 Reductions in coastal load of phosphorus to different basins

	Targets		Cost-effective solutions			
	Basin target, ton P	Basin target, %	Basin target, ton P	Basin target, %	Catchment target, ton P	Catchment target, %
BB	0	0	0	0	0	0
BS	0	0	0	0	0	0
BP	10,783	83	10,783	82	10,784	81
GF	1,997	15	1,997	15	1,997	15
GR	178	1	178	1	178	1
DS	0	0	88	1	136	1
KA	0	0	167	1	161	1
Sum	**12,958**	**100**	**13,213**	**100**	**13,256**	**100**

Thus, cost-effective policies might for many basins lead to over-achievement in the sense that nutrients will be further reduced than required to reach the internationally agreed reductions. The results suggest that in particular, there would be excess nitrogen reductions. The results, however, may be sensitive with regard to assumptions made about e.g. nutrient retention. Nevertheless, the occurrence of excess load reductions is a phenomenon that is likely to occur as many measures cause reductions in both nitrogen and phosphorus loads.

This implies that the cost-effective solution to the BSAP basin targets it is not likely to be the cost-effective solution for the targeted improvements in transparency[36], cf. table 3.1. Instead, the results suggest that unnecessarily large resources will be spent on nutrient reductions. Firstly, marine research has shown that emission reductions to one basin affect nutrient concentrations in all other basins. Thus, reductions in nutrient loads to a basin with no target for that nutrient will contribute to lower concentrations in other basins, for which there are targets. This suggests that targets for these other basins could be lowered while still having the same reduction in nutrient concentrations. Secondly, marine research also suggests that environmental damage from nutrient pollution depends on the availability of both nitrogen and phosphorus in the water (Wulff et al., 2007). Thus, even if transports between basins are ignored, over-achievement with regard to nutrient load reductions can imply that water transparency is reduced beyond the target level defined in the BSAP agreement. Although excess reductions can be associated with additional benefits, this may not be economically defendable, as transparency targets might be met with smaller reduction efforts while resources can be saved for other purposes. This issue can be solved through a down-ward adjustment of load targets for some marine basins.

[36] Neither can it be the cost-effective way to meet the reductions in the load to each basin, which are the ultimate consequence of the BSAP target, after nutrient transports between different marine basins have been taken into account. These final loads are different from the coastal loads.

7.2 The distribution of abatement

The distribution of abatement differs between basin and catchment targets. In table 7.3, the nitrogen reductions in different countries are compared for the two different targets. If a basin target is applied in a cost-effective way, regions with low abatement costs will abate more and hence their costs will increase compared to the situation under the BSAP catchment targets. Likewise, high cost regions will abate less and accordingly their costs will fall. Results show that Danish, Latvian, Russian and Swedish nitrogen reductions will be lower than suggested by the BSAP agreement (see the left column in table 7.3), while all others make larger reductions. This is explained by marginal nitrogen reduction costs being higher in these four countries than in other countries emitting to the same marine basins. With catchment targets, Finland, Russia and Poland undertake larger nitrogen reductions compared to those required for the catchments in these countries. This is explained by the catchment phosphorus target being the only binding target for some of the catchments in these countries.

Table 7.3 Nitrogen reduction to coastal waters according to BSAP catchment targets* and the cost-effective reductions under the different BSAP targets.

	Targets		Cost-effective solutions			
	ton N	%	Basin target, ton N	Basin target, %	Catchment target, ton N	Catchment target, %
Denmark	17,309	13	13,818	10	17,309	10
Estonia	896	1	2,231	2	897	1
Finland	1,199	1	3,215	2	1,435	1
Germany	6,049	5	6,941	5	6,049	4
Latvia	2,562	2	0	0	2,562	2
Lithuania	11,746	9	12,366	9	11,746	7
Poland	62,395	48	90,316	63	90,316	54
Russia	6,966	5	747	1	13,623	8
Sweden	19,835	15	13,893	10	19,835	12
Sum	**129,372**	**100**	**143,526**	**100**	**163,772**	**100**

*Required country reduction is calculated as the sum of the catchment targets in the country in question.

In Denmark, there are excess phosphorus reductions under both targets, see table 7.4. These excess reductions are explained by the stringency of the nitrogen reduction requirements for Denmark and the Kattegat basin. With a basin target, Latvia, Russia and Sweden would reduce their phosphorus loads less than required by the catchment targets, as reductions in these countries are relatively more expensive than in other countries which emit to the same basins.

Table 7.4 Phosphorus reduction to coastal waters according to BSAP catchment targets* and the cost-effective reductions under the different BSAP targets.

	Targets		Cost-effective solutions			
	BSAP target ton P	BSAP target, %	Basin target, ton P	Basin target, %	Catchment target, ton P	Catchment target, %
Denmark	0	0	171	1	222	2
Estonia	222	2	404	3	222	2
Finland	146	1	529	4	146	1
Germany	242	2	336	3	274	2
Latvia	300	2	150	1	300	2
Lithuania	881	7	1,699	13	881	7
Poland	8,755	68	8,443	64	8,755	66
Russia	2,122	16	1,269	10	1,268	16
Sweden	291	2	185	1	334	3
Sum	**12,959**	**100**	**13,186**	**100**	**13,601**	**100**

*Required country reduction is calculated as the sum of the catchment targets in the country in question.

7.3 The distribution of costs

The costs to the different countries under the different targets are compared in table 7.5. The catchment target is more expensive than the basin target. This is explained by the further restrictions that are set on the spatial distribution of abatement. From the table, one can find that countries' abatement costs differ between targets. Five countries have higher costs under basin targets than under catchment targets; Estonia, Finland, Germany, Lithuania and Poland. Thus, these countries are likely to prefer catchment targets to a cost-effective distribution of abatement under the basin targets. Conversely, Denmark, Latvia, Sweden and Russia will prefer the basin target to the catchment one as long as load trading is not allowed for.

Table 7.5 Annual cost to countries under different targets

	Basin target, MEUR	Basin target, % of total cost	Catchment target, MEUR	Catchment target, % of total cost
Denmark	360	9	451	10
Estonia	93	2	25	1
Finland	128	3	7	0
Germany	52	1	39	1
Latvia	16	0	96	2
Lithuania	421	11	161	4
Poland	2,123	56	2,204	49
Russia	334	9	962	21
Sweden	281	7	585	13
Sum	**3,809**	**100**	**4,533**	**100**

7.4 The marginal cost of meeting BSAP targets

A review of the marginal costs of meeting the BSAP targets further illustrates the potential gains to society from a reallocation of abatement between countries that emit to the same basin. It also gives an indication of the size of the (hypothetical) nutrient load permit price per basin, which would emerge in a well-functioning load permit markets.

In table 7.6, the marginal abatement costs for the two nutrients under the basin targets are presented. Results show that marginal phosphorus reductions costs vary between 341 EUR/kg reaching coastal waters for the Gulf of Riga to 545 EUR/kg for the Gulf of Finland. Optimality requires that marginal costs are equal to marginal benefits. This implies that a reduction in the phosphorus loads to the Gulf of Finland by 1 kilo should, on the margin, result in a more than 50 % larger reduction in environmental damage than a corresponding reduction to the Baltic Proper. If not, this distribution of abatement between basins is not economically optimal. Marginal nitrogen load reduction costs vary between 5 EUR/kg for the Gulf of Finland to 66 EUR/kg for Kattegat. Hence, a reduction of nitrogen loads by 1 kilo to Kattegat must then result in nearly 13 times as large a reduction in environmental damage as a corresponding reduction to the Gulf of Finland. Although benefits of reductions are not well known, these results might serve as a basis for further discussion of whether the relative stringency of targets in different basins is economically motivated.

Table 7.6 Marginal nutrient reduction cost under the basin targets

	Nitrogen (EUR/kg to coastal waters)	Phosphorus (EUR/kg to coastal waters)
Bothnian Bay	0	0
Bothnian Sea	0	0
Baltic Proper	0	352
Gulf of Finland	5	545
Gulf of Riga	0	341
Danish Straits	8	0
Kattegat	66	0

With a catchment target, marginal costs can be calculated for each catchment and nutrient, see table 7.7. Results show that there are large variations in the marginal costs for nutrient load reductions to the same basin. This further illustrates the fact that the burden distribution is not cost-effective, as cost-effectiveness requires that marginal costs are equal for all catchments emitting to the same basin. The BSAP agreement means that nitrogen reductions to the Baltic Proper will be much more costly on the margin in Sweden and Latvia than in other countries. Thus, there would be economic gains if some of the Swedish and Latvian nitrogen abatement burdens for Baltic Proper were re-allocated to other countries. For phosphorus, the results suggest that the Swedish and Russian abatement costs are high and that gains could be made by reallocating abatement to other countries. The results also suggest that Russia has been assigned an inefficiently large abatement burden for nitrogen to the Gulf of Finland. The Danish marginal cost for nitrogen reductions to the Danish Straits and the Swedish marginal cost for nitrogen reductions to Kattegat are higher than for other countries emitting to the same basins, and hence costs could be saved through a reallocation of abatement.

Thus, an adjustment of the distribution of the abatement burden may reduce cost. Policy makers, however, can never know the true costs and in addition, costs might change over time. Therefore, cost-effective policy instruments which operate on the international level, such as e.g. tradable emission permits, seem a more adequate solution to the problem of international cost inefficiency. With such policy instruments polluters, who are likely to know their own costs, will seek to adjust their abatement effort to their actual cost. The policy instruments, however, will of course not automatically solve the problem with uncertainty about the impact of different measures on coastal load. The implications of such uncertainty still have to be evaluated by decision-makers when designing policy instruments.

Tabell 7.7 Marginal nutrient reduction costs under the catchment targets[1]

	Nitrogen EUR/kg N	Phosphorus EUR/kg P
Baltic Proper:		
Germany Baltic Proper	1.5	174
Poland Baltic Proper	NB	347
Sweden Baltic Proper	32.3	650
Estonia Baltic Proper	6.2	336
Latvia Baltic Proper	15.0	358
Lithuania	0.6	307
Russia Baltic Proper	NB	313
Gulf of Finland:		
Finland Gulf of Finland	NB	80
Russia Gulf of Finland	18.1	70
Estonia Gulf of Finland	4.7	336
Gulf of Riga:		
Estonia Gulf of Riga	-	271
Latvia Gulf of Riga	-	358
Danish Straits:		
Denmark Danish Straits	60.5	-
Germany Danish Straits	2.5	-
Sweden Danish Straits	8.0	-
Kattegat:		
Denmark Kattegat	65.5	-
Sweden Kattegat	69.1	-

[1] Where there is no target, this is shown with a "-". For catchments where only one nutrient is binding, such that reductions in the other are "for free", there is a zero marginal cost which is denoted by "NB" (not binding).

7.5 Policy instruments and how they can be applied at the international level

Policy instruments were broadly discussed in chapter 5. Here it is briefly discussed how instruments can be applied at the international level in order to reduce the costs of meeting BSAP targets. One first observation is that although the existence of cost uncertainty makes an environmental tax on nutrient loads preferable to emission permit trading, the absence of a international government that can collect taxes and redistribute the resources to the citizens in the region suggest that emission permit trading might be a preferable option.

There is a difference between a centralized emission trading system and a fully decentralized system. In the centralized case, national governments exchange emission or, more exactly, load permits where one ton of a nutrient to a particular basin from one country can be exchanged for one ton of the same nutrient to the same basin from another country. With decentralized emission trading different market agents and local municipalities carry out the trade, and the subject to be traded is emissions from the sources. A centralized system could reduce the difference in marginal costs between countries and help solving some of the incentives problems associated with the BSAP targets through addressing the distributional difficulties, as it would increase the net benefits of the targeted environmental improvement. On the other hand, it would be a market with few agents and thus there might be a risk of large agents acquiring market power, which to some extent can reduce the advantages of trade. A decentralized system does not suffer from this problem as the number of agents contributing to nutrient pollution of the sea is large. However, a fully decentralized scheme should, at least ideally, take spatial differences into account and thus, trade barriers and/or exchange rates between sources are necessary. It might be difficult to come to an agreement about such exchange rates throughout the Baltic region and if this is the case, decentralized emission trading might be more easily applied at the national or regional level. This is likely to increase the direct abatement cost but might save on transaction costs associated with a Baltic-wide decentralized system.

Independently of whether the system is centralized or decentralized, it is necessary to measure the effects of actions. With either system, this could be carried out through modeling and calculation, as done in existing, small-scale systems of nutrient trading (EPA, 2008a). The alternative, physical measurements, seems to be an implausible way to proceed, considering the large natural variation in nutrient transports over the year and between years. Altogether, emission and/or load trading in the Baltic region can reduce costs and provide additional incentives for national governments and private agents to undertake low-cost abatement. The choice between a centralized or decentralized scheme should be made taking into account the potential cost savings of the respective schemes, the transaction costs and the potential effects of market power. The risk of so-called "hot spot"-creation needs to be considered in either case, which means that it must be analyzed whether a trading scheme leads to a concentration of nutrient emissions in certain places that can have effects on e.g. inland or coastal water quality.

7.6 Gains from load trading between national governments

In this section, the gains from nutrient load permit trading by national governments are calculated, based on the above results. The calculations give an indication of the potential gains of a centralized load trading scheme.

The total net gains from load permit trade are the total savings in abatement costs when basin targets are met in a cost-effective way instead of being regulated for each single catchment. This net gain equals the difference between minimum cost for the basin and catchments targets, respectively, i.e. 724 Million EUR per year.

With load trade, the net costs of compliance to each country will be determined by abatement costs minus income from permit trade. The income from permit trade depends on the initial distribution of load permits: with a large initial allocation of load permits it is more likely that a country can sell permits and hence receive an extra income, while if a country is given a small allocation of permits, it is more likely that it needs to buy permits on the market and hence has to pay for these permits.

Here, the net cost after trade to different countries is calculated for different possible initial distributions of load permits. If the net cost after trade is lower to a country than the abatement cost associated with the catchment target, the country is likely to be more willing to accept a nutrient trading regime relative the "command-and-control"-regime implied by the catchment targets.

It is often argued that the initial allocation of permits should be based on equity criteria. Following Van Regemorter (2005) the initial allocations are based on three different equity weights:

a) Equal rights to the use of the sea, translated into allocation the same level of load permits per head to each region.

b) Historic responsibility, translated into a greater effort now for big polluters in the past, implying load permits proportional to current loads, and

c) In function of the ability to pay, translated into an allocation of load permits inversely proportional to GDP per capita.

As can be seen from table 7.8, the choice of allocation rule has considerable effects on the net costs. If load permits are allocated according to historical loads, which is here interpreted as an allocation according to the BSAP catchment targets, all countries gain from trade compared to when countries are forced to fulfill the catchment targets within their borders. Finland and Estonia are the major winners from trade under this initial allocation, while Poland makes more or less negligible gains. If load permits are initially allocated according to the "equal rights"-rule, five out of nine countries have smaller costs than under the BSAP catchment targets without trade. If initial permits are allocated according to ability-to pay, eight out of nine countries have lower costs than under catchment targets without trade. With the ability-to-pay rule, Poland could easily be compensated by either Estonia or Finland that are the two major winners under the rule.

Table 7.8 Net cost after trade with different initial allocation of load permits, Million EUR per year[1]

	Historical loads[2]		Equal rights[3]		Ability to pay[4]	
	Net cost in Mill. EUR	% of cost under catchment target with no trade	Net cost in Mill. EUR	% of cost under catchment target with no trade	Net cost in Mill. EUR	% of cost under catchment target with no trade
Denmark	362	**80**	434	**96**	381	**84**
Estonia	-7	**-29**	178	712	-818	**-3,3**
Finland	-21	**-294**	-417	**-6**	-43	**-614**
Germany	23	**58**	76	195	0	**0**
Latvia	69	**71**	74	**77**	-192	**-200**
Lithuania	133	**83**	292	181	-86	**-53**
Poland	2152	**98**	1,935	**88**	3,300	150
Russia	755	**78**	1,014	105	809	**84**
Sweden	343	**59**	222	**38**	457	**78**

[1] Countries that have smaller net costs after trade compared to the cost under the catchment target with no trade have costs written in bold. A negative net cost in the table is a net gain to a country.

[2] The calculations in the first column assume the same allocation of load permits as that implied by the BSAP catchment targets.

[3] The calculations in the second column assume equal right to emit to each basin per head in the catchments that drains to the basin in question.

[4] The calculation in the third column builds on an initial allocation that is inversely proportional of GDP per capita 2006 according to World Bank data.

7.7 Summary

In this chapter, cost-effective solutions to the BSAP targets are discussed and it is shown that load permit trade can reduce the costs of meeting BSAP basin targets. The discussion is based on results from a numerical model covering the Baltic Sea drainage basin. The analysis shows that:

- The BSAP targets appear in two different versions, where one version, the “basin targets” defines reductions of nitrogen and phosphorus *to each basin* of the Baltic Sea, necessary to meet ecological objectives. The other version, which is the one intended to govern the implementation of the basin targets, takes the form of “catchment targets” which defines necessary nitrogen and phosphorus *to each* *basin from each country*[37]. Thus, the latter implies further restrictions on the spatial distribution of abatement. It is estimated that it will cost 3.8 Billion EUR per year to meet the basin target, while the catchment target will cost 4.5 Billion EUR per year. The two different targets will be associated different allocations of nutrient reductions between countries and basins.

- No matter whether load permit trade is allowed or not, total nitrogen reductions will exceed the reductions required by the BSAP targets by a substantial amount. The main reason is that for the Baltic Proper, large abatement efforts are necessary to meet the phosphorus target. Measures employed will at the same time reduce nitrogen loads. The excess nitrogen reductions are likely to contribute to water transparency in both Baltic Proper and in other basins beyond the improvements targeted by HELCOM. This suggests, however, that there is a possibility to save costs by adjusting basin targets such that water transparency targets are achieved, but not over-achieved. This could be attained through a reduction of some of the load targets.
- Results show that the marginal abatement costs for nitrogen and phosphorus reductions vary substantially between basins when basin targets are pursued. For this to be economically optimal, nitrogen reductions to Kattegat and phosphorus reductions to the Gulf of Finland must result in higher marginal benefits than reductions to other basins.
- With catchment targets, marginal abatement costs vary substantially between different countries emitting to the same basin. This shows that there are gains from nutrient load trading between countries that emit to the same basin.
- Several countries can have reasons to resist an abatement burden distribution where loads to each basin are reduced in a cost-effective way. The reason is that such a distribution is associated with higher costs for low-cost countries. This problem might be solved through the introduction of load trade, where some countries are allocated a relatively larger share of the total number of permits. Calculations show that if the initial permits are allocated according to the BSAP

[37] In the BSAP documents, the "catchment targets" are referred to as country targets, but the term "catchment targets" is used here as it more precisely defines how the targets are designed.

catchment targets all countries gain from the introduction of load trade at national level, gains to Poland, however, are minor.

8 National cost-effectiveness

The cost-effective solutions to the targets the BSAP allots to Sweden as well as the current Swedish national nutrient targets are in the following contrasted with actual Swedish nutrient policies after 1995. The strategy to meet nutrient targets for the Baltic Sea has been widely debated over several decades and in the last year it has become clear that the Swedish EPA and the Swedish Board of Agriculture differ strongly in the view they take on the appropriate strategy as well as the associated costs (EPA, 2008b). Here, it is investigated whether the actual policy pursued in Sweden between 1995 and 2005 is similar to the cost-effective policy under current and BSAP targets. Costs as well as the allocation of measures are compared and policy changes required to meet BSAP targets are discussed. Note that in this chapter, emission or load trading is not dealt with.

The chapter is organized as follows: in section 8.1, Swedish reductions after 1995 are compared to the BSAP targets. This comparison reveals that actual policy is far from meeting BSAP catchment targets for both nitrogen and phosphorus. In section 8.2 Swedish cost-effective policies are compared under current and BSAP targets, and it is shown that the restriction of abatement efforts by catchment under the BSAP will increase costs substantially compared to a cost-effective policy under current targets. In section 8.3, it is discussed how actual Swedish policy pursued between 1995 and 2005 could be improved upon, and it is observed that several measures currently applied might not be motivated unless they contribute substantially to other environmental targets. For some measures it is less clear that such side-benefits exist. The chapter ends with a summary in section 8.4.

8.1 Actual policy and the BSAP targets

When comparing to the BSAP target, it becomes clear that actual nitrogen reductions 1995-2005 are below those required in all catchments that have a BSAP target, see table 8.1. For phosphorus, reductions are made in all catchments although BSAP only requires reductions to the Baltic Proper, for which the BSAP target is not met. Table 8.1 shows considerable reductions have been made in catchments that do not have a BSAP target for that particular nutrient. As was noted in the foregoing chapter, these reductions are not in vain from an environmental perspective as nutrients are transported among the different basins and reductions in a nutrient in a catchment without a target will affect environmental quality in basins that have a target. Yet, from a policy perspective, these improvements will not be accounted for when evaluating the progress of different countries in meeting the BSAP targets. For example, nitrogen reductions made in the Bothnian Sea and Skagerrak catchments, and phosphorus reductions made in the Bothnian Sea, Kattegat and Skagerrak basins, are of no value with regard to the BSAP targets in spite of the effect that these reductions have on nutrient concentrations in the sea.

Table 8.1 Annual load reductions by catchment after 1995 and the BSAP targets[1]

	N red to coastal waters since 1995 (tons)	P red to coastal waters since 1995 (tons)	BSAP target for N red to coastal waters (tons)	Actual reductions since 1995/BSAP target for N red to coastal waters (%)	BSAP target for P red to coastal waters (tons)	Actual reductions since 1995/BSAP target for P red to coastal waters (%)
Bothnian Bay	188	19	0	+	0	+
Bothnian Sea	1,166	121	0	+	0	+
Baltic Proper	6,559	121	8,087	81	291	42
The Danish Straits	1,184	11	1,733	68	0	+
Kattegat	3,673	96	11,128	33	0	+
Skagerrak	2,462	90	0	+	0	+
Gulf of Riga	21	33	0	+	0	+
Gulf of Finland	220	36	0	+	0	+
Sum	**15,473**	**527**	**20,948**	**74**	**291**	**181**

[1] A "+" indicates that there are reductions made 1995-2005 but there is no BSAP target.

8.2 National cost-effective fulfillment of current and BSAP targets, respectively

It is well known from economic theory that the choice of environmental target formulation affects the cost-effective mix of abatement measures. In this section, it is investigated to what extent an adoption of the BSAP targets would call for an adjustment of Swedish nutrient policy. This is done through a comparison of the cost-effective solutions to BSAP targets and current Swedish targets. The cost-effective strategies under these two sets of targets are reported in tables 8.2 and 8.3. In the tables, data on costs for measures that solely give rise to reductions in one nutrient are separated from data on costs for measures that reduce both nutrients. For the latter, it is not possible to identify which nutrient gives rise to the costs. In addition, results are discussed in relation to actual policy, cf. table 4.1 and 4.2.

An investigation of the cost-effective measures under the BSAP target, see table 8.2, and current targets, see table 8.3, shows several interesting things. Firstly, independently of the target chosen, a considerably larger weight is placed on reductions in the agricultural sector compared to the actual policy during 1995-2005. The emphasis on agricultural reductions is even stronger under the cost-effective solution to the BSAP target than under the cost-effective solution to current targets. In addition, nitrogen oxide emission reductions are substantially larger under current as well as BSAP targets compared to actual policy. The significance of reductions in these sectors in the cost-effective solutions is explained by the restrictions on wastewater reductions in the model that reflect the fact that the potential for further reductions at wastewater treatment plants have been reduced since 1995.

The agricultural sector plays a relatively more important role under the cost-effective solution to the BSAP targets. This is explained by the further spatial restrictions on abatement under the BSAP targets. As a consequence of the further spatial restrictions, more measures in the agricultural sector must be used in order to meet the target in every single catchment. This implies that the cost of agricultural sector abatement is twice as high under the BSAP target as under the current set of targets.

Although the total phosphorus reductions are of similar magnitude under both targets, phosphorus abatement is restricted to the Baltic Proper region under the BSAP targets. This implies that the cost of phosphorus abatement is increased considerably, which is indicated in the tables. For example, private sewers are needed to meet the phosphorus target, implying a unit cost of 483 EUR/kg P to coastal waters. The inclusion of private sewers as a measure under the BSAP target explains one third of the difference in cost between the two targets while the larger costs in the agricultural sector explains two thirds. Thus, the geographical concentration of the phosphorus reduction target implies that less expensive measures in other parts of the country have to be replaced by expensive investment in private sewers in the Baltic Proper catchment. Also, more expensive phosphorus load reductions are made through measures at wastewater treatment plants and through reduced phosphorus fertilizer use in the Baltic Proper catchment. Considering that measures in other catchments also would affect phosphorus loads to the Baltic Proper through the nutrient exchange between basins (see e.g. Savchuk, 2003), it is likely that costs can be saved through a more general phosphorus policy, while the same reductions in phosphorus concentrations in the Baltic Proper still are achieved.

Under the BSAP targets, total phosphorus reductions will be larger than those required by the international agreement. Similarly as in chapter 7, this is explained by measures with an impact on both nitrogen and phosphorus, which are used in catchments without a phosphorus target. These measures can be cost-effective with regard to the nitrogen targets for the catchments, although as a side-effect they also reduce phosphorus loads.

Table 8.2 Cost-effective reductions and abatement costs under the BSAP catchment target

	N red. to coastal water (tons)	P red. to coastal water (tons)	Total N red. Cost (MEUR)	N red. Cost (EUR/kg to coastal waters)	Total P red. Cost (MEUR)	P red. Cost (EUR/kg to coastal waters)	Joint costs of N and P red. (MEUR)	Total cost* (MEUR)
Water sector:								
Wastewater treatment	105	97	2	20	13	128		15
Private sewers	452	120	31	68	58	483		89
P-free detergents		29				81		2
Sum water sector	*557*	*246*	*33*		*72*			*105*
NOx emissions:								
Shipping sector (NOx-N)	48		0	4				0.2
Transport sector (NOx-N)	3,083		73	24				73
Energy sector (NOx-N)	73		3	40				3
Sum NOx-emissions	*3,204*		*76*					*76*
Agricultural sector:								
Catch crops	1,537	6					18	18
Grassland	6,616						181	181
Wetlands	1,471	5					23	23
Buffer Strips								
Livestock reductions	5,787	58					171	171
Fertilizer reductions**	662	19	6	10	4	185	0	10
Sum agricultural sector	*16,074*	*89*	*6*		*4*		*394*	*404*
Total sum	**19,835**	**334**	**115**		**76**		**394**	**585**

*The total cost is the sum of the cost for the measures that reduce nitrogen, measures that reduce phosphorus and measures that reduce both nutrients

**The unit cost included in the table is the average cost, although in the model the cost function for fertilizer reductions is increasing.

Table 8.3 Cost-effective reductions and abatement costs under current nutrient targets

	N load red. to coastal water (tons)	P load red. to coastal water (tons)	Total N red. cost (MEUR)	N red. cost (EUR/kg to coastal waters)	Total P red. cost (MEUR)	P red. cost (EUR/kg to coastal waters)	Joint costs of N and P red. (MEUR)	Total cost* (MEUR)
Water sector:								
Wastewater treatment	107	144	2	17	13	88		14
Private sewers								
P-free detergents		54			4	76		4
Sum water sector	*107*	*198*	*2*		*17*			*18*
NOx emissions:								
Shipping sector (NOx-N)	48		0.1	4				0.1
Transport sector (NOx-N)	3,562		84	24				84
Energy sector (NOx-N)								
Sum NOx-emissions	*3,611*		*84*					*84*
Agricultural sector:								
Catch crops	1,588	6					19	19
Grassland	5,420		46	9				46
Wetlands	1,246	4					15	15
Buffer Strips								
Livestock reductions	4,270	47					104	104
Fertilizer reductions**	650	95	5	8	6	59	0	11
Sum agricultural sector	*13,173*	*152*	*52*		*6*		*138*	*196*
Total sum	**16,890**	**350**	**138**		**22**		**138**	**299**

*The total cost is the sum of the cost for the measures that reduce nitrogen, measures that reduce phosphorus and measures that reduce both nutrients

**The unit cost included in the table is the average cost.

In table 8.4, the distribution of reductions and costs of different regions is shown for the two targets. Under the BSAP target, the largest costs occur in the Baltic Proper catchment, followed by Kattegat. Under current targets, the major cost occurs in the Kattegat catchment, while costs in the Baltic Proper basin are lower. A look at the load reductions made in the two regions suggests that the larger costs in the Kattegat catchment are explained by the possibility to undertake relatively cheap phosphorus reduction measures in this region. Under the BSAP targets only small costs arise in the Danish Strait catchment, while considerable costs are allocated to this catchment under current targets. This is explained by the large potential to reduce nitrogen loads at low cost in this catchment under current targets. The possibility to allocate costs to the Skagerrak catchments seems to matter little to the overall cost under current targets.

Table 8.4 Cost-effective Swedish load reductions under the BSAP catchment targets and current Swedish targets

	BSAP targets			Current targets		
	Nitrogen red., to coastal waters tons	Posphorus red. to coastal waters, tons	Cost, MEUR	Nitrogen red. to coastal waters, tons	Phosphorus red. to coastal waters, tons	Cost, MEUR
Baltic Proper	8,087	291	314	2,748	115	70
Danish Straits	1,733	4	11	7,723	59	83
Kattegat	10,015	39	260	5,960	171	136
Skagerrak	-	-	-	459	6	10
Sum	**19,835**	**334**	**585**	**16,890**	**350**	**299**

Results suggest that overall, a cost-effective policy under the BSAP target implies higher total costs in both the Baltic Proper and Kattegat catchments, while costs in the Danish Straits catchment are lower.

8.3 Can existing Swedish policies be improved?

Looking at the cost-effectiveness of past reductions, it is here asked

1. what combination of measures, and location of those, could have reduced the costs for the Swedish nutrient policy 1995-2005 while maintaining the same environmental effect?
2. how large a reduction could, at the most, have been made for the same cost as those for reductions made 1995-2005?.

When comparing Swedish policies since 1995 with BSAP targets, one can note the BSAP catchment targets refer to average loads to coastal waters between 1997 and 2003. Loads to coastal waters in one particular year result from upstream emissions in an earlier year. The time it takes for emissions from the inland to reach coastal waters is not well known, and may be quite different for nitrogen and phosphorus. Here, however, Swedish policy achievements after 1995 are compared to the cost-effective reductions associated with the BSAP targets. This implies that it is implicitly assumed that emissions at the sources 1995 would result in average loads 1997-2003.

The model presented in chapter 6 has been used to calculate the minimum cost for the same amount of reductions as those achieved after 1995 to all basins[38]. These calculations suggest that the cost-savings associated with a change to a cost-effective policy are in the order of 5 MEUR per year, i.e. relatively small. It is interesting to note that there is a cost-saving in spite of the fact that the model assumes a much more limited capacity for improvements in wastewater treatment compared to the reductions undertaken in the period[39]. Next, it is worthwhile to investigate how the choice of cost-effective measures differs from the actually implemented ones. Results show that:

a) Reductions of nitrogen oxide emissions at power plants are made in actual policy during 1995 -2005 but are not included in the cost-effective solution. This is not surprising given the data on unit costs in table 4.1. Reductions in nitrogen oxide emissions contribute to other environmental targets and results suggest that the value of this contribution to other target must be at least 2.4 EUR/kg N for this measure to be economically optimal. A

[38] Including Bothnian Bay and Bothnian Sea, as well phosphorus reductions to Kattegat and the Danish Straits.
[39] This limitation is reasonable as the model is based on current data, and much of the capacity for e.g. wastewater reductions might be exhausted now, compared to 1995.

relatively recent study on willingness to pay for improved respiratory health in Sweden, suggests that the value of nitrogen oxide-reductions amounts to approximately 12 EUR/kg N (Huhtala and Samakovlis, 2007). Thus, measures against nitrogen oxide emissions from power plants might be defended due to their joint impact on eutrophication and respiratory health.

b) The cost-effective geographical distribution of some measures sometimes differs considerably from the actual one. In contrast to actual policy, the cost-effective strategy comprises only buffer strips in the Bothnian Sea region, as that is the only place where phosphorus reductions have been large enough to motivate such an expensive measure. Buffer strips could, however, provide additional environmental values, such as e.g. biodiversity. Buffer strips are currently used extensively in e.g. the Baltic Proper and Kattegat regions, and results suggest that for this measure to be economically motivated at the current level of abatement, the additional side-benefits would have to amount to at least 100 and 200 EUR/ha in the Baltic Proper and Kattegat regions, respectively. Although increased efforts to reduce emissions from rural households not connected to wastewater treatment plants is frequently demanded by the EPA, the results suggest that at the current level of abatement, this measure is only cost-effective in the Bothnian Sea. This is, again, explained by the large phosphorus reductions carried out in that region since 1995.

c) Some measures are included in the cost-effective solution but are not applied in reality, such as reductions in cattle holdings in Kattegat region and conversion to grassland in the northern catchments as well as in the Baltic Proper region. Those are measures that simultaneously reduce nitrogen and phosphorus emissions, implying that they have a cost advantage compared to measures that only reduce a single nutrient. In addition, chemical fertilizer use is reduced in the cost-effective solution. This measure accounts for 4 and 61 % of nitrogen and phosphorus reductions, respectively, in the model solution.

The model results show that marginal costs of phosphorus load reductions vary considerably between catchments, between 3 and 44 EUR/kg P. The highest marginal cost is in the Bothnian Sea catchment. Marginal costs of nitrogen load reductions are surprisingly similar for the catchments, approximately 2 EUR/kg N to coastal waters, with the highest cost in the Baltic Proper catchment. This might suggest that nitrogen policies in Sweden have to some extent been carried out with a view on cost-effectiveness with regard to the impact on coastal loads to the Baltic Sea, while this seems not to have been the case for phosphorus policies.

Next, it is discussed how much more the environment of the Baltic Sea could have been improved with the same money, had a cost-effective policy been employed since 1995. Thus, to what extent could BSAP targets have been met with the same budget? To investigate the maximum impact on BSAP targets, it is first calculated *how large a share of BSAP targets can be achieved*, given that this share is the same for all nutrients and basins, where a target is defined. Thus, the Swedish catchment targets are all considered equally important. The calculations show that all BSAP targets for Swedish catchments could be fulfilled to 65 %, given the budget for measures undertaken since 1995. This implies a reallocation of resources compared to actual policies. No resources are used in the two northern basins. No resources are used for phosphorus reductions in the southern catchments, except in the Baltic Proper catchment. The results also show that a relatively modest reduction in the nitrogen abatement efforts in the Baltic Proper catchment would imply that considerably higher nitrogen reductions were possible in the Kattegat region, where the nitrogen target is currently far from being met. In addition, it would be possible to achieve a much higher reduction in phosphorus loads to the Baltic Proper. One can also note that with equal weights on all targets, marginal costs for phosphorus reductions to the Baltic Proper amount to 48 EUR/kg P. Marginal nitrogen reduction costs for Baltic Proper and Kattegat are relatively similar, amounting to 30 and 28 EUR/kg N, while reduction costs to the Danish Straits are lower, 8 EUR/kg. Thus, equal weighting of the targets is economically optimal if, for example, the value of environmental damage avoided by reducing phosphorus loads to Baltic Proper by 1 kilo is 6 times as high as the corresponding effect of an additional reduction of nitrogen loads to the Danish Straits by 1 kilo.

However, it might be that not all BSAP targets should be equally weighted. There may be reasons to prioritize reductions of one of the nutrients, such as indicated e.g. by Boesch et al. (2006). *If nitrogen is prioritized*, and the aim is to achieve the maximum reduction of nitrogen loads for the budget available, then calculations show that nitrogen targets could be reached to 70 %. Thus, compared to the above scenario where all targets are weighted equally, only small improvements in nitrogen reductions could be achieved by ignoring the phosphorus target. In particular, nitrogen reductions in the Baltic Proper catchment are expensive on the margin in this scenario.

If, on the other hand, *the phosphorus target for the Baltic Proper is prioritized*, then results show that this target can be met with the available budget if policy makers are satisfied with having nitrogen targets fulfilled to 55 %. This level of nitrogen reduction would be an improvement for the Kattegat catchment, but imply smaller reductions for Baltic Proper and the Danish Straits compared to what has been done since 1995. In this scenario, marginal costs for phosphorus load reductions in Baltic Proper are nearly 1,390 EUR/kg at the target level, while the marginal nitrogen reduction cost in this scenario ranges from 8 EUR per kg for the Danish Straits to 24 and 28 EUR per kg for the Baltic Proper and Kattegat, respectively. Thus, for this type of weighting of the targets to be optimal, the impact on environmental damage would have to be more than 170 times larger for 1 kilo phosphorus emitted to Baltic Proper compared to 1 kilo of nitrogen emitted to the Danish Straits.

8.4 Summary

Above, Swedish national policies against nutrient emissions have been assessed. First, it is investigated whether a change from the current national nutrient reduction targets to the BSAP catchment targets will call for a significant change of the direction of Swedish nutrient policy, given an ambition to pursue cost-effective policies. Second, it is analyzed whether actual policies could have been improved upon, either by achieving a given reduction at lower cost or by attaining further nutrient reductions with the same budget. The main results of the assessment can be summarized as follows:

- The BSAP targets are more costly to attain as compared to the current targets, given that policies are carried out in a cost-effective manner. The calculations suggest that the costs are nearly twice as high as under the current policy. The explanation is the more restricted spatial distribution of abatement. Although there is uncertainty about underlying data, the conclusion that there is a substantial cost increase seems undisputable.
- Phosphorus reduction costs are higher with a BSAP target than under current targets. Although total phosphorus reductions required are of similar magnitude, the concentration of phosphorus reduction efforts to the Baltic Proper catchment under the BSAP targets implies expensive measures in the Baltic Proper catchment have to replace less expensive measures in other catchments. Considering that measures in other catchments also affect phosphorus concentrations in the Baltic Proper through the water exchange between basins, it is possible that costs can be saved through a more general phosphorus policy, while still meeting the intended reductions in phosphorus concentrations.
- Results suggest that with BSAP targets, relatively more efforts should be devoted to measures in the agricultural sector and relatively less efforts to reductions of air-borne nitrogen emissions.
- With targets for each catchment, such as the BSAP targets, much of the abatement actually carried out 1995-2005 will not count when evaluating target achievement. The reason is that for many basins there is only a target for a single nutrient. Yet measures that reduce both nutrients might be used to meet the single-nutrient target. Because of nutrient exchange between different parts of the sea, reductions in the other nutrient will also give rise to positive environmental effects.
- Policies for nitrogen oxide emissions in the energy sector are not cost-effective at the current level of abatement when only the effect on nutrient loads to the Baltic Sea is considered. For this measure to be economically motivated, the positive side-effects must exceed 2.4 EUR/kg N. Studies on the value of Swedish health effects from nitrogen oxides, suggest that those might be larger than that, implying that

reductions of nitrogen oxide emissions from the energy sector are cost-effective.

- Buffer strips are not cost-effective if BSAP targets or current targets should be met and only nutrient loads to the Baltic are considered. For this measure to be economically motivated the side-benefits with regard to biodiversity must amount to 100-200 EUR/ha.
- Further efforts to enforce regulations of emissions from households in rural areas, not connected to waste water plants, do not seem to be economically motivated.
- The annual budget of 330 MEUR that has been used for the nutrient policy improvements since 1995, is enough to meet 65 % of all Swedish BSAP targets. If nitrogen is prioritized, then 70 % of the nitrogen targets can be achieved and if phosphorus is prioritized, the BSAP phosphorus target can be met while simultaneously achieving 55 % of the nitrogen targets. Thus, the BSAP policy implies increased costs compared to the policy actually pursued in Sweden 1995-2005, even if the new policy is implemented cost-effective manner. As shown in the foregoing chapter, load permit trade could reduce the cost of meeting the BSAP basin targets substantially. This would benefit Sweden, although the savings made depend on the initial allocation of permits.

9 Summary and discussion

Eutrophication of the Baltic Sea is a major environmental problem with negative impacts on e.g. the value of recreational activities and biodiversity in the sea. Since the 1970s, the Baltic Sea countries have addressed the eutrophication problem through both international cooperation and national environmental policies.

In the 1980's the governments around the Baltic Sea agreed to reduce the total emissions of both nitrogen and phosphorus by half. This target was never met, however, and in 2007 new provisional targets were agreed upon in the Baltic Sea Action Plan (BSAP). The BSAP targets appear in two versions in the international agreement:

1. Targets for reductions of the nutrient loads *to each of* the Baltic Sea's seven *basins* (the *basin targets*), and
2. Targets for reductions of nutrient loads *from each country to each basin* (the *catchment targets*).

Given these environmental targets, the task is to develop cost-effective policy instruments to attain them. A restoration of the sea is likely an expensive project, even if load reductions are carried out in a cost-effective way. Thus, it is in the countries' joint interest to keep these costs as low as possible. Each government also needs to consider how to ensure cost-effectiveness domestically, something that requires coordination of policies across sectors, regions and measures. Designing cost-effective policies to reduce nutrient loads is a delicate task, considering how the environmental impact of a given amount of emissions crucially depends on where the source is located.

In this report, it has been investigated how the costs for meeting the water quality targets of the BSAP might be reduced by:

a) nutrient load trading amongst the Baltic Sea countries/-regions,
b) a different choice of load reductions to certain basins
c) reallocation of Swedish abatement efforts amongst measures and regions, given the Swedish nutrient load reduction targets.

The design of policy instruments for nutrient reductions is discussed in the light of the complex behavior of nutrient transport and transformation processes and the role that governments on international, national and local level play for water management decisions. The major conclusions are presented below:

- *The potential gains from basin-wide nutrient load trading and the introduction of a "clean development mechanism" are substantial.*

When comparing the two different versions of the BSAP targets, the catchment targets are a more expensive way to reach the intended reductions to the sea's basins compared to the basin targets. The reason is that the catchment targets imply further restrictions on the location of abatement and therefore, it is not possible to take advantage of all low-cost abatement options. Calculations in this report suggest that the minimum costs for meeting the catchment target exceed those of meeting the basin targets by approximately 20 percent or more than 700 MEUR per year. As a consequence of applying catchment targets, marginal abatement costs vary substantially between different countries emitting to the same basin. For example, the Swedish marginal cost for reducing phosphorus to Baltic Proper is approximately twice as high as the marginal costs for other countries emitting to the same basin. And perhaps even more surprising, the Estonian marginal cost for phosphorus reductions to the Gulf of Finland is approximately four times larger than the corresponding marginal cost for Finland and Russia. This clearly illustrates that is possible to reallocate abatement efforts, compared to the catchment targets, in ways that lowers the costs of attaining the BSAP basin targets.

However, identifying the cost and effect of each measure in each catchment with certainty is not an easy task. In particular, both future costs and effects can change as a consequence of e.g.

future changes in economic conditions or climate change. Therefore, a redistribution of the abatement burden between countries may only partially solve the problem of meeting targets at minimum cost. Still more important perhaps, is that even if a cost-effective allocation of abatement can be perfectly determined, several countries may resist a burden allocation where reductions to each basin are reduced in a cost-effective way. One obvious reason is that such a distribution is associated with higher costs to low-cost countries. However, there is a potential solution to this problem; emission or load trading. Well-functioning load permits trade among countries can reduce costs for all countries, provided that permits are initially distributed in an appropriate manner. If this is the case, then all countries will have incentives to agree on a load permit trading scheme. In the report, the implications of three different rules for allocation of load rights, based on equity criteria, are investigated. The results show that if initial load rights are distributed according to the BSAP burden distribution (i.e. according to historical loads) or according to ability to pay, all countries have incentives to choose load permits trade before the "command-and-control"-distribution of abatement implied by the BSAP catchment targets. Thus, load permit trade might solve some of the implementation difficulties at the international level. This can be important if there is a risk that, because of the high abatement costs, the BSAP agreement will suffer from the same implementation difficulties as the earlier international load reduction agreement. No doubt, a load permit trade system would provide high-cost countries with more adequate economic incentives than those provided by existing international financing institutions that today have the role of solving the income re-distribution problem associated with the allocation of the abatement burden.

Nutrient load trading can be implemented in numerous ways. For instance, it can be arranged as intergovernmental trading with load quota units, such as suggested in this report. If, instead, trade decisions where to be decentralized to individual companies and municipalities, the system will have to take into account that the effects of nutrient emissions varies between sources depending on location. Furthermore, for some measures the impact is more uncertain than for others. This can be solved through the introduction of "exchange rates" between different types of sources and regions, implying that reductions at one source are assigned a higher value than reductions at another source. Due to transaction costs associated with the identification of differences in impact, it would be costly to design a system that took the full variation in effect into account, but simplifications would have to be made. However, one should be aware that the need to estimate the size of these differences in effects is equally high with a decentralized "command-and-control" system, for "command-and-control" instruments to be cost-effective.

There are additional ways to reduce the costs of meeting environmental targets for the Baltic Sea. The countries Belarus and Ukraine contribute to nutrient loads to the Baltic Sea but have no obligations under the BSAP. They can, therefore, not be expected to undertake emission abatement to the extent that is optimal for the Baltic Sea region. A mechanism akin to the Clean Development Mechanism under the Kyoto Protocol would create incentives for emission abatement in these countries. A BSAP country committed to load targets can then invest in abatement activities in Belarus or Ukraine, and be credited the reductions (or a part thereof) attained there when it comes to complying to BSAP. Such a mechanism can reduce total abatement costs. However, one has to establish systems that prevent a BSAP country to be credited for project that would have been undertaken anyway.

- *Cost-effective fulfillment of the BSAP load targets can imply that water clarity is improved beyond the target levels*

This report shows that if the BSAP targets are applied by all countries in the region, nitrogen reductions may substantially exceed the agreed targets. The reason is that large abatement efforts are necessary to meet the phosphorus target. A large number of the measures needed to meet the phosphorus targets, in particular in the agricultural sector also reduce nitrogen loads, in some cases beyond the target levels. In particular, this is so in the Baltic Proper. When loads are reduced beyond target levels, water transparency improvements may well go beyond the objectives of the BSAP.

Although data used in the model are the best available, there is great uncertainty about both cost and effects of abatement. This implies that the above conclusions, regarding which nutrient will be abated beyond targets, can be sensitive to assumptions made about e.g. nutrient retention. Yet it is reasonable to believe that the results hold qualitatively, i.e. only one nutrient might be binding and therefore, excess nutrient reductions might be made even when policies are cost-effective with regard to the BSAP load targets. Thus, costs can be saved by lowering some of the targets while still meeting the intended water quality improvements.

- *Swedish nutrient targets can be met at lower cost*

The BSAP stipulates catchment-wise load targets for Sweden. Compared to the current Swedish domestic targets the BSAP catchment targets is here estimated to be twice as expensive to attain. This is due to the further restrictions on the spatial allocation of abatement that come with the BSAP. Cheap phosphorus abatement options in the Kattegat region and cheap nitrogen abatement options in the Danish Straits region will not be utilized. If basin-wise load trading is introduced, then at least the low-cost nitrogen abatement options in the Danish Strait region would be used which would lower costs for meeting the target for that particular basin.

The concentration of Swedish phosphorus reduction efforts to the Baltic Proper catchment, required by the BSAP agreement, implies that expensive measures in the Baltic Proper catchment replace less expensive measures in other catchments. Measures in other catchments also affect phosphorus concentrations in the Baltic Proper, and hence, a phosphorus policy with less spatial restrictions might achieve the same environmental improvement at lower cost. A policy adjustment in this direction would be straightforward to apply, at least technically, as information on the impact of a reduction to one basin on all other basins is available from existing marine models over the Baltic Sea.

Sweden has since 1995 reduced annual nitrogen loads by 15,500 tons for nitrogen and 530 tons for phosphorus. The annual total costs for these reductions exceed 300 Million EUR. The results presented here suggest that policies for nitrogen oxide emissions in the energy sector should not be used at all at the current ambition level if nutrient reductions to the sea are the only environmental target that matters. Also, construction of buffer strips, as well as more stringent enforcement of regulations to control emissions from households not connected to waste water plants, should be avoided in most parts of the country. Thus, these measures are not economically justified unless they are associated with large enough positive side-effects e.g. for air quality and biodiversity. Results from hypothetical valuation studies indicate that side-benefits might be large enough for nitrogen oxide emissions from the energy sector.

Analysis presented above suggests that for the 300 Million EUR that Sweden has spent since 1995, 65 % of all Swedish BSAP targets can be met if resources are used cost-effectively. If nitrogen is prioritized and phosphorus is ignored, then 70 % of the nitrogen targets could be achieved and if phosphorus is prioritized and nitrogen ignored, the BSAP phosphorus target could be met while simultaneously achieving 55 % of the nitrogen targets. Thus, relatively large phosphorus reductions to the Baltic Proper could be undertaken through a reallocation of the nutrient reduction budget, and this reallocation will come at a small cost in terms of the ambitions to reduce nitrogen loads.

The policy debate in Sweden is sometimes confusing due to the different views that different parties have towards the response of the ecosystem to nutrient reductions and the potential for these reductions. For instance, the Swedish Board of Agriculture claims that BSAP targets cannot be met even if large parts of Swedish agricultural land is laid fallow while the EPA claims that target achievement is not a problem (EPA, 2008b). The calculations in this report suggest that there can be difficulties to meet the nitrogen target in the Kattegat region, while the remainder of the targets can be met. An obvious solution for the Kattegat problem is basin-wise nutrition load trading, implying that Sweden can meet its nitrogen target for that basin by financing additional load abatement in Denmark. Moreover, Swedish costs, including those of the agricultural sector, can be reduced substantially through a centralized system of load permit trading between countries. Load permits can be bought from countries with lower costs, with both countries benefitting from this trade.

Recently, the Swedish EPA has come up with a proposal for a decentralized domestic trading system (EPA, 2008a). This is an interesting idea. Although the practical application of a decentralized system remains to be developed, one can note that there are positive experiences from similar systems in the US in terms of realized cost savings for the subjects of the regulation. This type of emission trading may also provide additional incentives for the development of abatement technologies in the agricultural sector. Also, if several sectors are involved in decentralized nutrient trading, this would facilitate a cost-effective allocation of measures between sectors, which would reduce the aggregate cost of Swedish nutrient policies. The US's experience shows clearly that a pre-condition for successful decentralized trading is that the government determines "exchange rates" for emission permits of different measures and regions, i.e. the relative value of different abatement options. These "exchange rates" need to be based on existing scientific knowledge in the field and it seems reasonable to believe that the knowledge on the appropriate value of "exchange rates" is of comparable quality in Sweden and the US.

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Appendix A. The cost-effectiveness model

In this appendix, the conditions for cost minimization for non-uniformly mixing pollutants are derived. Consider an aquifer that is negatively affected by nutrient emissions from human activities in the surrounding watershed. The watershed is divided into *i=1,…,n* regions. There are two different nutrients emitted to the aquatic environment, nitrogen and phosphorus. The nutrients are denoted by a subscript *r*, with *r=N,P*, where *N* denotes nitrogen and *P* phosphorus. Nutrient emissions can be reduced through different measures x_{ij}, with *j=1,…,m*, where denotes the type measure. The reduction of emissions of nutrient *r* from land-based sources to a marine basin are defined by $Q_r = \sum_i \sum_j \alpha_{rij} x_{ij}$, where a_{rij} is the impact of a measure on coastal loads. Because some of the nutrients are captured in vegetation, soils and inland waters on the way from the sources to the sea, $0 \le \alpha_{rij} \le 1$.

Also, assume that there is an international environmental agent who wants to reach nutrient reduction targets Q_r^* for each basin, and that it is required that $Q_r \ge Q_r^*$ is achieved at minimum cost. Furthermore, it is assumed that there are cost functions for nutrient reductions, denoted $c_{ij}\left(x_{ij}\right)$. The cost functions are assumed to be increasing and convex in the measures[40]. The decision problem of the environmental agent who wants to reduce nutrient loads to a particular marine basin can then be written as:

[40] For all of the measures except fertilizer reductions, the cost function in the programming model is linear, i.e $c_{ij}\left(x_{ij}\right) = k_{ij} x_{ij}$, where k_{ij} is the constant marginal cost.

$$
\begin{aligned}
&\underset{x_{ij}}{\text{Min}} \ \sum_i \sum_j c_{ij}(x_{ij}) \\
&\text{s.t.} \\
&Q_r = \sum_i \sum_j \alpha_{rij} x_{ij} \\
&Q_r \geq Q_r^* \\
&0 \leq x_{ij} \leq \bar{x}_{ij}
\end{aligned} \tag{1}
$$

Where $\bar{x}_{ij}$ is the maximum reduction that can be achieved with x_{ij}. The optimal allocation of emissions can be determined from the solution to the cost minimization problem. The Langrangean for the above problem is:

$$L = \sum_i \sum_j c_{ij}(x_{ij}) + \sum_r \lambda_r \left(Q_r^* - Q_r\right) \tag{2}$$

The objective function is convex according to assumptions made about cost functions. The load reduction function is differentiable and quasi-concave, and assuming that there exist a point x_{ij}^* that satisfies the conditions in (1), the following Kuhn-Tucker conditions are necessary and sufficient for a global solution to the problem stated in (1).

$$
\begin{aligned}
&\frac{\partial L}{\partial x_{ij}} = \frac{\partial c_{ij}}{\partial x_{ij}} - \sum_r \lambda_r \alpha_{rij} \geq 0 , \quad x_{ij} \geq 0, \quad x_{ij} \frac{\partial L}{\partial x_{ij}} = 0 \\
&\frac{\partial L}{\partial \lambda_r} = Q_r^* - Q_r \geq 0 , \quad \lambda_r \geq 0, \quad \lambda_r \frac{\partial L}{\partial \lambda_r} = 0
\end{aligned} \tag{3}
$$

With an interior solution, the optimal level of emissions is defined by

$$\frac{\partial c_{ij}}{\partial x_{ij}} = \sum_r \lambda_r \alpha_{rij} , \tag{4}$$

where λ_r is the Lagrange multiplier for reductions of nutrient *r*. On the left hand side of (4) we have the change in cost due to a marginal increase in nutrient reductions with a given measure in a given region. On the right hand side, we have the sum of the impacts of the measure on nutrient loads to coastal waters, weighted by the corresponding Lagrange multipliers.

The data used in the model are included in Appendix B.

Appendix B. Model parameters

Table B.1 Unit costs at the sources

	SCR heavy vehicle (SEK/ kg N)	SCR ships (SEK/ kg N)	SCR on power plants (SEK/ kg N	Fertilizer reductions (SEK/kg N)	(SEK/kg P)	Livestock reductions Cattle (SEK/head)	Pigs (SEK/head)	Poultry (SEK/head)	Manure spreading time (SEK/kg N)	Sewage, urban (SEK/ kg N)
Denka	34	5	54	0–2.93	0–25.61	3,100	618	39	9	141
Denso	34	5	54	0–2.93	0–25.61	3,100	618	39	9	141
Fibb	33	5	53	0–1.65	0–28.95	2,235	575	57	9	141
Fibs	33	5	53	0–3.42	0–250.82	2,217	534	54	9	141
Fifv	33	5	53	0–3.42	0–401.57	2,018	525	42	9	141
Gerso	30	5	51	0–5.22	0–98.37	2,053	555	39	14	141
Gerbp	30	5	51	0–5.22	0–98.37	2,053	555	39	14	141
Vist	31	5	52	0–0.83	0–2.06	1,162	398	31	9	113
Oder	31	5	52	0–0.83	0–2.06	1,162	398	31	9	113
Polcoast	31	5	52	0–0.83	0–2.06	1,162	398	31	9	113
Sebb	31	5	53	0–1.26	0–1.31	2,235	575	57	9	141
Sebs	31	5	53	0–1.59	0–4.36	2,217	534	54	9	141
Sebap	31	5	53	0–1.35	0–21.96	2,018	525	42	9	141
Sebano	31	5	53	0–1.35	0–21.96	1,899	486	50	9	141
Seso	31	5	53	0–1.35	0–21.96	2,074	501	39	9	141
Seka	31	5	53	0–1.35	0–21.96	1,788	525	42	9	141
Estob	32	5	52	0–1.35	0–21.96	1,843	712	55	15	113
Estog	32	5	52	0–0.70	0–0.91	1,843	712	55	15	113
Estof	32	5	52	0–0.70	0–0.91	1,843	712	55	15	113
Latvib	31	5	52	0–0.70	0–0.91	1,361	597	46	15	113
Latvig	31	5	52	0–0.41	0–0.91	1,361	597	46	15	113
Lith	31	5	51	0–0.41	0–0.91	492	230	18	15	113
Sukal	31	5	52	0–1.46	0–2.93	1,162	398	31	14	113
Supet	31	5	52	0–0.08	0–1.67	1,162	398	31	14	113

Source: Gren et al. (2008).

Table B.1 Unit costs at the sources, continued

	Sewage, rural (SEK/kg N)	Private sewers (SEK/kg N)	Catch crops (SEK/ha)	Grass land (SEK/ha)	Energy forest (SEK/ha)	Buffer strips (SEK/ha)	Wetlands (SEK/ha)	Sewage, urban (SEK/kg P)	Sewage, rural (SEK/kg P)	Private sewers (SEK/kg P)	P-free detergents (SEK/kg P)
Denka	301	509	829	8,286	6,473	8,286	11,681	573	1,260	2,397	103.4
Denso	301	509	829	8,286	6,473	8,286	11,708	573	1,260	2,397	103.4
Fibb	301	509	269	2,691	4,765	2,691	4,560	573	1,260	2,397	103.4
Fibs	301	509	269	2,691	4,824	2,691	4,560	573	1,260	2,397	103.4
Fifv	301	509	269	2,691	4,824	2,691	4,560	573	1,260	2,397	103.4
Gerso	301	509	507	5,068	6,227	5,068	8,490	573	1,260	2,397	103.4
Gerbp	301	509	507	5,068	6,227	5,068	8,490	573	1,260	2,397	103.4
Vist	254	429	76	758	747	758	2,041	385	837	2,013	103.4
Oder	254	429	76	758	747	758	2,041	385	837	2,013	103.4
Polcoast	254	429	76	758	747	758	2,041	385	837	2,013	103.4
Sebb	301	509	38	378	4,765	378	3,428	573	1,260	2,397	103.4
Sebs	301	509	55	545	4,824	545	3,595	573	1,260	2,397	103.4
Sebap	301	509	146	1,463	5,987	1,463	4,513	573	1,260	2,397	103.4
Sebano	301	509	164	1,638	5,987	1,638	4,688	573	1,260	2,397	103.4
Seso	301	509	320	3,199	8,169	3,199	6,249	573	1,260	2,397	103.4
Seka	301	509	164	1,638	6,473	1,638	4,688	573	1,260	2,397	103.4
Estob	254	429	76	758	747	758	1,735	385	837	2,013	103.4
Estog	254	429	76	758	747	758	1,735	385	837	2,013	103.4
Estof	254	429	76	758	747	758	1,735	385	837	2,013	103.4
Latvib	254	429	54	541	747	541	1,108	385	837	2,013	103.4
Latvig	254	429	54	541	747	541	1,108	385	837	2,013	103.4
Lith	254	429	21	210	747	210	559	385	837	2,013	103.4
Sukal	254	429	21	210	747	210	559	385	837	2,013	103.4
Supet	254	429	21	210	747	210	559	385	837	2,013	103.4

Source: Gren et al. (2008).

Table B.2 Impact on coastal load, kg N per unit[41] change at the source

	SCR heavy vehicle	SCR ships	Fertilizer reduction	SCR on power plants	Livestock reductions:			Spring man. spread
					Cattle	Pigs	Poultry	
Denka	0.14	0.14	0.002	0.14	5.06	1.08	0.07	0.12
Denso	0.14	0.14	0.002	0.14	5.06	1.08	0.07	0.12
Fibb	0.13	0.14	0.003	0.13	7.14	1.55	0.10	0.17
Fibs	0.13	0.14	0.009	0.13	7.15	1.55	0.10	0.17
Fifv	0.13	0.14	0.009	0.13	7.16	1.55	0.10	0.17
Gerso	0.07	0.14	0.013	0.07	3.86	0.88	0.06	0.10
Gerbp	0.07	0.14	0.013	0.07	3.86	0.88	0.06	0.10
Vist	0.10	0.14	0.008	0.10	3.70	0.96	0.06	0.13
Oder	0.10	0.14	0.008	0.10	3.70	0.96	0.06	0.13
Polcoast	0.10	0.14	0.008	0.10	3.70	0.96	0.06	0.13
Sebb	0.14	0.14	0.003	0.14	5.19	1.17	0.08	0.12
Sebs	0.14	0.14	0.004	0.14	5.77	1.30	0.08	0.13
Sebap	0.14	0.14	0.003	0.14	5.67	1.27	0.08	0.13
Sebano	0.14	0.14	0.003	0.14	5.67	1.27	0.08	0.13
Seso	0.14	0.14	0.015	0.14	9.10	2.04	0.13	0.20
Seka	0.14	0.14	0.004	0.14	8.36	1.88	0.12	0.19
Estob	0.14	0.14	0.011	0.14	8.65	2.19	0.14	0.26
Estog	0.14	0.14	0.011	0.14	8.65	2.19	0.14	0.26
Estof	0.14	0.14	0.011	0.14	8.65	2.19	0.14	0.26
Latvib	0.14	0.14	0.003	0.15	6.54	1.70	0.11	0.23
Latvig	0.14	0.14	0.003	0.15	6.54	1.70	0.11	0.23
Lith	0.12	0.14	0.061	0.12	7.69	2.02	0.13	0.27
Sukal	0.12	0.14	0.0002	0.12	5.59	1.49	0.10	0.25
Supet	0.09	0.14	0.0002	0.09	4.54	1.21	0.08	0.20

Source: Own calculation from Gren et al. (2008).

[41] Units are given in table B1.

Table B.2 Impact on coastal load, kg N per unit[42] change at the source, continued

	Catch crops	Grass land	Energy forest	Buffer strips	Wetlands	Sewage, urban	Sewage, rural	Private sewers
Denka	2.77	8.23	0.17	0.00	176.98	0.81	0.91	0.90
Denso	2.86	8.51	0.16	0.00	195.14	0.81	0.91	0.90
Fibb	1.83	8.49	0.17	0.00	32.81	0.64	0.72	0.71
Fibs	1.83	8.52	0.17	0.00	87.70	0.64	0.72	0.71
Fifv	1.84	8.60	0.21	0.00	414.57	0.64	0.72	0.71
Gerso	4.48	15.64	0.09	0.00	369.15	0.59	0.67	0.66
Gerbp	4.45	15.50	0.09	0.00	339.62	0.59	0.67	0.66
Vist	0.83	9.24	0.16	0.00	204.14	0.49	0.57	0.56
Oder	0.78	8.61	0.17	0.00	185.58	0.49	0.57	0.56
Polcoast	0.72	7.90	0.19	0.00	204.14	0.49	0.57	0.56
Sebb	0.10	2.01	0.73	0.00	1.24	0.69	0.78	0.77
Sebs	0.19	2.75	0.53	0.00	3.02	0.65	0.74	0.73
Sebap	0.72	4.84	0.30	0.00	12.79	0.36	0.40	0.40
Sebano	0.85	5.73	0.28	0.00	15.84	0.36	0.40	0.40
Seso	6.95	42.65	0.07	0.00	83.33	0.63	0.71	0.70
Seka	1.48	6.28	0.12	0.00	35.79	0.72	0.81	0.80
Estob	0.95	13.07	0.15	0.00	36.14	0.67	0.78	0.77
Estog	0.97	13.30	0.14	0.00	34.69	0.67	0.78	0.77
Estof	0.96	13.30	0.14	0.00	25.89	0.67	0.78	0.77
Latvib	0.27	3.86	0.38	0.00	17.32	0.48	0.56	0.55
Latvig	0.27	3.89	0.37	0.00	11.92	0.48	0.56	0.55
Lith	0.30	10.50	0.14	0.00	27.94	0.57	0.66	0.65
Sukal	0.12	1.35	0.31	0.00	3.91	0.35	0.41	0.40
Supet	0.11	1.24	0.34	0.00	5.94	0.35	0.41	0.40

Source: Own calculation from Gren et al. (2008).

[42] Units are given in table B1.

Table B.3 Impact on coastal load, kg P per unit[43] change at the source.

	Fertilizer reduction	Livestock reductions			Catch crops	Buffer strips	Wetlands
		Cattle	Pigs	Poultry			
Denka	0.0002	0.07	0.03	0.002	0.084	0.22	1.35
Denso	0.0002	0.07	0.03	0.002	0.111	0.27	1.67
Fibb	0.0026	0.15	0.05	0.003	0.044	0.33	1.93
Fibs	0.0224	0.15	0.05	0.003	0.053	0.36	2.74
Fifv	0.0373	0.15	0.05	0.003	0.055	0.36	6.07
Gerso	0.0011	0.04	0.01	0.001	0.351	0.75	2.83
Gerbp	0.0011	0.04	0.01	0.001	0.271	0.59	2.19
Vist	0.0004	0.23	0.09	0.006	0.024	0.30	3.01
Oder	0.0004	0.23	0.09	0.006	0.026	0.31	3.08
Polcoast	0.0004	0.23	0.09	0.006	0.023	0.29	4.45
Sebb	0.0002	0.09	0.03	0.002	0.005	0.15	0.11
Sebs	0.0005	0.11	0.04	0.003	0.008	0.15	0.11
Sebap	0.0012	0.05	0.02	0.001	0.002	0.08	0.07
Sebano	0.0022	0.05	0.02	0.001	0.006	0.11	0.11
Seso	0.0006	0.12	0.04	0.003	0.036	0.43	0.24
Seka	0.0084	0.07	0.03	0.002	0.003	0.05	0.10
Estob	0.0003	0.25	0.10	0.006	0.004	0.32	0.25
Estog	0.0003	0.25	0.10	0.006	0.003	0.31	0.28
Estof	0.0003	0.25	0.10	0.006	0.001	0.29	0.19
Latvib	0.0003	0.31	0.11	0.008	0.007	0.25	0.26
Latvig	0.0003	0.31	0.11	0.008	0.001	0.20	0.22
Lith	0.0020	0.44	0.16	0.011	0.005	0.24	0.23
Sukal	0.0002	0.12	0.04	0.003	0.015	0.12	0.06
Supet	0.0001	0.06	0.02	0.002	0.011	0.08	0.06

Source: Own calculation from Gren et al. (2008).

[43] Units are given in table B1.

Table B.3 Impact on coastal load, kg P per unit[44] change at the source, continued.

	Sewage treatment, upstream urban areas	Sewage treatment, upstream rural areas	Private sewers, upstream	P-free detergents, upstream connected	P-free detergents, upstream unconnected	P-free detergents, direct connected
Denka	0.86	0.99	0.98	0.24	0.99	0.24
Denso	0.86	0.99	0.98	0.24	0.99	0.24
Fibb	0.65	0.75	0.74	0.21	0.75	0.28
Fibs	0.65	0.75	0.74	0.21	0.75	0.28
Fifv	0.65	0.75	0.74	0.21	0.75	0.28
Gerso	0.35	0.40	0.40	0.08	0.41	0.21
Gerbp	0.35	0.40	0.40	0.08	0.41	0.21
Vist	0.56	0.63	0.62	0.38	0.63	0.62
Oder	0.56	0.63	0.62	0.38	0.63	0.62
Polcoast	0.56	0.63	0.62	0.38	0.63	0.62
Sebb	0.53	0.61	0.60	0.12	0.61	0.21
Sebs	0.53	0.61	0.60	0.12	0.61	0.21
Sebap	0.47	0.54	0.53	0.11	0.54	0.21
Sebano	0.47	0.54	0.53	0.11	0.54	0.21
Seso	0.88	1.01	1.00	0.21	1.01	0.21
Seka	0.53	0.61	0.60	0.12	0.61	0.21
Estob	0.58	0.65	0.64	0.37	0.65	0.58
Estog	0.58	0.65	0.64	0.37	0.65	0.58
Estof	0.58	0.65	0.64	0.37	0.65	0.58
Latvib	0.54	0.61	0.60	0.30	0.61	0.51
Latvig	0.54	0.61	0.60	0.30	0.61	0.51
Lith	0.63	0.71	0.70	0.54	0.71	0.78
Sukal	0.36	0.40	0.40	0.24	0.41	0.61
Supet	0.36	0.40	0.40	0.33	0.41	0.83

Source: own calculation from Gren et al. (2008).

[44] Units are given in table B1.

Appendix C. Costs for and effects of Swedish measures against eutrophication 1995-2006

Here the costs for and effects of load reductions caused by Swedish environmental policy changes after 1995 are calculated. Policies judged to aim at reduced nutrient load are collected in table C1. Reductions by industries and municipal wastewater treatment plants are assumed to be driven by changes in legislation through regulation of emissions and technology. Wastewater treatment outside the country-borders has been achieved through government subsidies. Reductions of emissions from energy and transport sectors are assumed to be caused by the introduction of catalytic converters required by legislation. Changes in waterborne loads from the agricultural sector through catch crop cultivation, wetland construction, spring plowing and construction of buffer strips are caused by changes in support for agri-environmental measures. Wetlands have been created also with support from the local investment programmes (LIP). Reductions in ammonia emissions from the agricultural sector have been achieved through a combination of investment support and regulation.

Table C.1 **Measures that have lead to load reductions due to policy change after 1995**

	Nitrogen	Phosphorus
Industry		
NOx in transport sector	X	
NOx from energy/industry sector	X	
Wastewater treatment	X	X
Wastewater treatment abroad	X	X
Catch crops	X	X
Wetlands	X	X
Spring plowing	X	
Buffer strips		X
Ammonia red. in agric. sector	X	

Industry sector emission reductions

Reductions in emissions are assumed to be casued by changes in legislation and in emission permits. The industries that are the most important for nitrogen and phosphorus emissions are: the pulp- and paper, the chemical industry and the metallurgic industry. Data on emissions from industry as well as industry production are available from SCB. Industry production has risen between 1995 and 2005, while emissions of nitrogen and phosphorus have declined. This suggests that emission reductions due to environmental policies are larger than simply the difference between total emissions in the different years.

Emissions reductions due to policies have been calculated through muliplication of emissions in 1995 with the industry production index and then, subtraction of emissions in 2005. The distribution of emissions on different drainage basins has been made using SCB data (SCB, 2007a; SCB, 2000). For the pulp- and paper industry, the regional distribution of inland emissions is not available for 1995, wherefore emissions are assumed to be proportionally higher in all catchments compared to 2004, except for the Bothnian Bay and Bothnian Sea catchments, where it is assumed that no nitrogen reductions have been made in the time period, as policies have focussed on southern Sweden (EPA 2007a, tables 3.2 and 3.3). The chemical and metallurgic industries are assumed to be located in the inland. All chemical industry on the Swedish west coast is assumed to be located in the Kattegat drainage basin. Emissions from the Danish Straits catchment cannot be distinguished in data, but are included in Baltic Sea catchment numbers. The metallurgic industry is assumed to be wholly located in the Bothnian Sea drainage basin. Costs and effects are assumed to be equal to costs for and effects of reductions in wastewater emissions in Gren et al. (2008). In table C2, the reductions in emissions between 1995 and 2005 are shown together with the costs for those emissions.

Table C.2 **Industrial N and P emission reductions between 1995 and 2005, effect on coastal load and annual cost**

	Nitrogen reduction at sources		Phosphorus reduction at sources		Reduction of coastal N load (tons)	Reduction of coastal P load (tons)	Nitrogen reduction cost (MSEK)	Phosphorus reduction cost (MSEK)
	Inland (tons)	Coastal (tons	Inland (tons	Coastal (tons)				
Bothnian Bay	0	90	0	11	90	11	13	7
Bothnian Sea	0	778	1	118	778	118	110	68
Baltic Proper	1,152	183	18	23	594	32	188	24
The Danish Straits	0	0	0	0	0	0	0	0
Kattegat	518	64	39	7	435	27	82	26
Skagerrak	0	2	0	0	2	0	0	0
Sum	**1,670**	**1,117**	**58**	**159**	**1,899**	**188**	**393**	**124**

Municipal waste water treatment

Nutrient emission reductions in municipal wastewater treatment plants, due to environmental policies, are calculated as emissions in 1995, multiplied by the population increase 1995-2005 minus emissions in 2005. To this end, data from SCB have been used (SCB, 2007a; SCB, 2001 and online population statistics). The regional distribution of inland emissions is not available for 1995, wherefore emissions are assumed to be proportionally higher than in 2004. An exception is made for Bothnian Bay and Bothnian Sea catchments, where it is assumed that no nitrogen reductions have been made in the time period, as policies have focussed on southern Sweden (cf. EPA 2007a, tables 3.3 and 3.4). The net reduction in total loads, given these assumptions are displayed in table C3. Costs and effects on coastal load are calculated using data from Gren et al. (2008), where data for wastewater treatment has been used.

Table C.3 WWTP emission reductions between 1995 and 2004, effect on coastal load and annual cost

	Nitrogen reduction		Phosphorus reduction		Reduction of coastal	Reduction of coastal	Nitrogen reduction	Phosphorus reduction
	Inland (tons)	Coastal (tons)	Inland (tons)	Coastal (tons)	N load (tons)	P load (tons)	cost (MSEK)	cost (MSEK)
Bothnian Bay	0	0	2	7	0	8	0	5
Bothnian Sea	0	0	6	0	0	3	0	3
Baltic Proper	1,756	4,061	22	33	4,688	43	820	31
The Danish Straits	127	814	2	9	893	10	133	6
Kattegat	1,159	1,416	17	59	2,246	68	363	44
Skagerrak[45]	9	2,319	0	90	2,325	90	328	52
Sum	**3,051**	**8,609**	**48**	**198**	**10,152**	**223**	**1,644**	**141**

Households

Measures aimed at households have not resulted in increased nutrient reductions since 1995. Legislation is in place but enforcement is weak (EPA, 2007).

Agriculture

The conditions for agricultural production changed radically when Sweden joined the European Union. A study by Johnsson och Mårtensson (2002) suggests that the net impact of increased production subsidies in combination with larger use of agri-

[45] Skagerrak N retention is 50 % according to http://www5.o.lst.se/projekt/skargard/-overgod.pdf.

environmental support implied that nitrogen emissions remained largely unaffected between 1995 and 1999. However, the 2003 agricultural reform led to a decoupling a agricultural support, which, according to the Swedish Board of Agriculture (2007) will lead to reductions of nitrogen leaching by approximately 10 % when the reform has reached its full effect (SJV, 2004). The effect of the reform on phosphorus leaching is unclear. In addition to the effects of decoupling, measures targeted towards nutrient leaching reduce emissions. Here, changes in nutrient emissions from agriculture that can be attributed to changes in pillar I-support, e.g. the decoupling reform in 2003, are not considered as a change caused by environmental policy, as the purpose of reforms has not been nutrient reductions. Here, only measures targeted at nutrient reductions are included.

Earlier, there was support for ley cultivation in whole Sweden. One of the purposes with this support was reductions of nitrogen leaching (SJV, 2003a). This support was however abolished in 1999. Currently, only farmers in the northern parts of Sweden can receive support for ley cultivation, but this support is not environmentally motivated. Here, this support is therefore not considered as a tool introduced to abate nutrient emissions.

Within the Swedish Rural Development program, which is partly financed by EU and partly by the Swedish government, there are several types of agri-environmental support aiming at reductions of nitrogen and phosphorus emissions. Although some similar measures were present in smaller extent already before 1995, it is here assumed that these measures were introduced after 1995, as data on the extent and impact of earlier measures is hard to obtain.

Catch crops and spring plowing

One of the available options to reduce nutrient leaching is through cultivation of catch crops. Data for counties have been recalculated to drainage basins through a comparison of the areas of different counties that fall within the borders of each catchment. Catch crops areas are for 2006 according to Environmental Objectives Portal (2008), effects and costs are according to Gren et al. (2008). The distribution of catch crops on different drainage basins is shown in table C4 together with the associated costs and effect on coastal load.

When spring plowing is used in combination with catch crops, the reduction of nitrogen leaching is increased by approximately 9 % (SJV, 2004, table 26). The total area of fields with support for spring plowing was 15,300 ha in 2006. Cost data for spring plowing are not available in Gren et al. (2008), wherefore the cost is assumed equal to the subsidy, which amounts to 300 SEK/ha. The geographical distribution of spring plowing is assumed proportional to that for catch crops. Data for spring plowing are collected in table C4. For the Skagerrak catchment, costs and effects are assumed the same as for the Kattegat catchment.

Table C.4 Catch crops and spring plowing: areas, effects and costs

	Catch crops				Spring plowing		
	Area of arable land w. catch crops, ha	N red to coastal waters (tons)	P red to costal waters (tons)	Costs (MSEK)	Area of arable land w. spring plowing (ha)	N red to coastal waters (tons)	Costs (MSEK)
Bothnian Bay	0	0	0.0	0	0	0	0.0
Bothnian Sea	0	0	0.0	0	0	0	0.0
Baltic Proper	42,320	30	0.1	6	3,910	6	1.2
The Danish Straits	16,052	112	0.6	3	1,483	19	0.4
Kattegat	89,505	132	0.3	29	8,269	22	2.5
Skagerrak	17,743	26	0.1	3	1,639	4	0.5
Summa	165,620	30	1.0	40	15,301	51	4.6

Wetlands

Most of the wetlands restored or constructed in Sweden are located on agricultural land and supported through the Rural Development Program. About 13 % of the constructed wetlands are supported from other sources. Data from Gren et al. (2008) is used for the effects and costs of wetland construction. Data on wetland construction areas in different counties have been obtained from Environmental Objectives Portal (2008). Data on costs and effects are collected in table C5.

Table C.5 Wetlands: areas, effects and costs

	Wetlands, constructed (ha)	N red to coastal waters (tons)	P red to coastal waters (tons)	Total cost (MSEK)
Bothnian Bay	0	0	0.0	0.0
Bothnian Sea	37	0	0.0	0.11
Baltic Proper	1,802	26	0.13	8.29
The Danish Straits	314	26	0.03	0.96
Kattegat	872	31	0.09	2.66
Skagerrak	10	4	0.01	0.31
Summa	**3,125**	**87**	**0.26**	**12.33**

Buffer strips

The main effect of buffer strips is to reduce phosphorus leaching. Support to buffer strips is provided through the Rural Development Program. Data on buffer strips in different counties have been obtained from Environmental Objectives Portal (2008). Data from Gren et al. (2008) is used for the effects and costs. Data on costs and effects are collected in table C6.

Table C.6 Buffer strips: areas, effects and costs

	Buffer strips (ha)	P red to coastal waters (tons)	Total cost (MSEK)
Bothnian Bay	0	0.0	0.0
Bothnian Sea	318	0.05	0.2
Baltic Proper	5,446	0.44	8.0
The Danish Straits	540	0.23	0.9
Kattegat	3,174	0.15	10.2
Skagerrak	411	0.02	1.3
Summa	**9,889**	**0.89**	**20.5**

Ammonium emissions

85% of ammonium emissions in Sweden originate from manure. Agricultural emissions fell from 54,575 ton 1995 to 46,250 ton 2005 (SCB, 2007b). Of this reduction, a reduction in livestock holding explains approximately 5,142 tons[46]. The remaining reduction can be explained by technological change. This technology change is assumed to be caused by changes in environmental legislation in combination with investment support (cf. Naturvårdsverket, 2003d). The total reduction in ”pure” nitrogen emission that results from this environmental policy-induced change (obtained by division by 1.2 to reflect the N content in ammonium) amounts to 2,650 tons. Of this, 19 % reaches the Baltic Sea directly and 59 % is deposited on land within the Baltic Sea drainage basin. In Sweden, total nitrogen deposition on land is 145 kton per year and of this, 10.5 kton reaches coastal waters[47]. Thus, approximately 7 % of total deposition on land reaches the sea. This is assumed to apply for the whole drainage basin. Then, in total, 23 % of ammonia emissions reach the sea. Data on ammonia emissions on regional level are available for 1995, and it is assumed that the regional distribution of ammonia emissions is the same in 1995 and 2006. The cost for measures that reduce ammonia emissions from manure are assumed to be 45 SEK per kilo ammonium, which is an average for the measures reviewed by the Board of Agriculture (1999). This corresponds to a cost of 54 SEK/kg N. Data on costs and effects are collected in table C7.

Nitrogen oxides from transport sector

The total Swedish nitrogen oxide emissions fell between 1995 och 2006 from 224 to 179 thousand tons (EPA, 2007), which corresponds to a reduction of pure nitrogen by 14 thousand tons[48]. A major share of the reduction is explained by more stringent emission regulation in the transport sector[49]. The reduction in emissions from the transport sector is 93 thousand tons NOx between 1995 and 2006 (equal to 56 %), corresponding to 28 thousand tons N (Miljömålsportalen, 2008). This reduction could, in principle, be caused by changes in the vehicle fleet, changes in fuel consumption and changes in emission reducing technologies. Here it is assumed that the cost of these reductions is equal to the cost of installing SCR-technique on heavy vehicles. Data on costs

[46] Own calculations from Elofsson and Gren (2003) and SCB (2007b).
[47] Own calculations from data per hectare deposition EPA (2007) and loads in EPA (2003).
[48] NOx is recalculated as N through division by a factor 3.3.
[49] Miljömålsportalen, as available 2007-12-11,http://miljomal.nu/Pub/Indikator.php?MmID=3&InkID=Kva-24-NV&LocType=CC&LocID=SE.

and effects are obtained from Gren et al. (2008). Resulting costs and effects are collected in table C7.

Table C.7 Ammonium emission reductions in the agricultural sector and nitrogen oxide emission reductions in the transport sector: effects and costs

	Ammonium			No_x		
	Ammonium emission red in agricultural sector (ton N)	N red to the Baltic Sea (ton)	Total cost (MSEK)	NOx reduction (ton N)	N red to the Baltic Sea (ton)	Total cost (MSEK)
Bothnian Bay	64	12	3	338	48	10
Bothnian Sea	500	96	27	1,423	202	44
Baltic Proper	2,001	384	108	3,831	545	119
The Danish Straits	310	59	17	422	60	13
Kattegat	1,548	297	84	2,713	386	84
Skagerrak	183	35	10	343	49	11
Sum	**4,606**	**883**	**249**	**9,071**	**1,290**	**281**

Nitrogen oxide emissions from the shipping sector

For shipping, fees that are differentiated with regard to Nitrogen oxide emissions were introduced in 2005, but the impact of these fees on emissions have not been analyzed so far (EPA, 2006b).

Nitrogen oxide emissions from energy sector

Nitrogen oxide emissions from the energy sector have fallen between 1995 and 2006. As the NOx-fee has not changed in this period, the reduction is here assumed to be caused by changes in emission permits. Data on emissions on regional level for 1995 and on reductions in total emissions are obtained from Miljömålsportalen (2008). The reduction is assumed to be proportional in all regions. Costs and effects have been obtained from Gren et al. (2008) and data for installation of SCR-technique are assumed to apply for all Nitrogen oxide emission reductions in the sector. Data on costs and effects are collected in table C8.

Table C.8 NOx emission reductions in the energy sector: effects and costs

	Reduction of energy sector NOx emissions (ton N)	Reduction in N load to the Baltic Sea (ton)	Total cost N reductions (MSEK)
Bothnian Bay	269	38	14
Bothnian Sea	636	90	34
Baltic Proper	1,342	191	71
The Danish Straits	103	15	5
Kattegat	876	124	46
Skagerrak	123	17	7
Sum	**3,348**	**476**	**177**

Measures abroad partly financed by Sweden

Through SIDA, Sweden has co-financed a number of wastewater treatment projects in Latvia, Lithuania, Estonia and Russia[50]. For the six projects undertaken in the Baltic States, the Swedish contributions to the projects amount to 8 % of total cost. The projects have been finalized between 1997 och 2001. These projects have led to a reduction of N and P emissions by 967 and 985 tons per year, respectively. Of this 8 % is assumed to be due to Swedish co-financing, i.e. 84 tons N and 79 tons P. Four of the six wastewater treatment plants are situated at the coast. For the two remaining, the impact on the sea is reduced through retention.

Sweden has also contributed to the financing of a new wastewater treatment plant in St. Petersburg. In, this case, the Swedish contribution to total cost is 10 %[51]. The loads to coastal waters will be reduced by 220 tons N and 360 tons P[52], whereof, then, 220 tons N and 36 tons P can be attributed to the Swedish contribution. Data on costs and effects are obtained from Gren et al. (2008) and figures for wastewater emission reductions for the

[50] For data on the Baltic States, see Specialrapporten. For data on Swedish investments in St. Petersburg´s, wastewater treatment plant see www.sida.se. and http://www.swedenabroad.com/Page____39394.aspx.
[51] Consulate General of Sweden, St. Petersburg. As available 2007-12-11. http://www.swedenabroad.com/Page____39394.aspx.
[52] Finland’s environmental administration. As available 2007-12-11. http://www.ymparisto.fi/print.asp?contentid=152024&lan=fi&clan=sv.

relevant countries are assumed to apply. Data on costs and effects are collected in table C9.

Table C.9 Nutrient reductions abroad, financed by Sweden

	"Swedish" N reduction (tons)	"Swedish" P reduction (tons)	Reduction in N load to coastal waters (ton)	Reduction in P load to coastal waters (ton)	Total cost N reductions (MSEK)	Total cost P reductions (MSEK)
Baltic Proper	98	50	95	45	11	19
Gulf of Riga	33	33	21	33	7	13
Gulf of Finland	220	36	220	36	3	5
Sum	**351**	**120**	**336**	**114**	**21**	**37**

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