A numerical model for dynamic cost effective mitigation of eutrophication with spatial heterogeneity in the Baltic Sea – technical report

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Abstract. The purpose of this paper is to present the dynamic model for the calculation of cost effective nutrient abatement for a marine sea with heterogeneous coupled marine basins which differ with respect to fast and slow responses to changes in external nutrient loads. A discrete dynamic model with heterogeneous sites – drainage and marine basins – is developed. The application to the Baltic Sea for achievement of the ministerial agreement among the riparian countries on water quality targets (Helsinki Commission Baltic Sea Action Plan) shows expected results; abatement is increasing over time and Poland faces the largest cost burdens. A specific finding is that, in spite of the complex and interlinked nutrient transports, policy design is facilitated by the stringent phosphorus load target on one specific marine basin, Baltic Proper, which is characterised by a relatively slow dynamic process and large interchange with other basins. The achievement of the phosphorus target in this basin then implies fulfilment of nutrient targets in the other basins. It is also shown that total costs under the dynamic cost effective solution can be considerably lower than the nutrient reduction program suggested by the international ministerial agreement, Helcom BSAP.

Key words; cost effectiveness, dynamics, nitrogen, phosphorus, eutrophication, Baltic Sea

1. Introduction

Damages from eutrophication in the Baltic Sea have been documented since early 1960s by a number of different studies (e.g. Wulff et al. 2001). The riparian countries also showed concern by, among other things, the manifestation of the administrative body Helcom in charge of policies for improving Baltic Sea in 1974, and ministerial agreements on nutrient load targets in 1988 and 2007 (Helcom 1993; Helcom 2007). However, in spite of the ambitious agreements of reducing nutrient loads by 50 per cent in 1988, long-term monitoring of nutrient transports, political concern, and improved scientific understanding of the functioning of the sea, degradation of the sea continues. Approximately 20 years after the meeting in 1988, the agreed level of nutrient reductions in 1988 is far from being reached. One important reason for the hesitation to reduce nutrient loads to the Baltic Sea is by all likelihood associated costs, which now start to increase at a higher rate than earlier since the low cost options, such as improvement in nutrient cleaning at sewage treatment plants located at the coastal waters of the Sea, have been implemented in several countries. Therefore, careful cost calculations are now likely to be more important than earlier. Furthermore, the timing of implementation of measures determines total costs of a cleaning program. The time frame of the recent ministerial agreement on nutrient reductions to the Baltic Sea is that these should be implemented by all riparian countries at the latest in 2021. Whether or not this is cost effective policy in a dynamic perspective depends on several factors such as the dynamics of nutrient and biological responses in different parts of the Baltic Sea and the discount rates.

The purpose of this study is to present a discrete dynamic model allowing for the minimisation of total costs under consideration of dynamic processes and heterogeneous marine basins. Examples of cost effective solutions - allocation of costs during time and in space - are presented for different assumptions of timing of nutrient load targets. Associated design of two types of policy instruments, charges and nutrient permit markets, is shown. The paper also evaluates whether or not the ministerial agreement in autumn 2007 on nutrient reductions to the Baltic Sea, the Helcom Baltic Sea Action Plan (BSAP), meets the conditions of dynamic cost effectiveness.

Similar to several other international waters, the Baltic Sea contains a number of interlinked and heterogeneous marine basins. The biological conditions in these basins differ, and the

BSAP therefore suggests different nutrient load targets for the basins. However, since the basins are coupled, nutrient load reduction to one basin diffuses into other basins. This means that both dynamic and spatial dispersions of abatement need to be taken into account when identifying cost effective timing and location of abatement. Starting in mid 1990s there is by know a relatively large economics literature on cost effective or efficient nutrient load reductions to the Baltic Sea (e.g. Gren et al., 1997; Elofsson, 1999, 2006, 2007; Gren 2001, 2008; Ollikainen and Honkatukla, 2001; Hart and Brady, 2003; Hart, 2003; Laukanen and Huhtala 2008; Laukanen et al., 2009). Several studies calculate cost effective or efficient allocation of abatement among the riparian countries in a static setting (Gren et al., 1997; Elofsson, 1999, 2006; Gren 2001, 2008; Ollikainen 2001).

All dynamic models on nutrient management except for Laukanen and Huhtala (2008) include one nutrient, either nitrogen or phosphorus (Hart and Brady, 2003; Hart, 2003; Mäler et al., 2003; Naevdal, 2003; Elofsson, 2006; Laukanen et al. 2009). Hart (2003), Brady and Hart (2003), Elofsson (2007) and Laukanen and Huhtala (2008) constitute empirical studies with thorough theoretical foundations. Hart (2003) evaluates the comparative advantages of mussel farming as a nitrogen abatement measure under consideration of dynamics on nutrient transports in the drainage basins and also in the coastal water. Brady and Hart (2003) calculate optimal allocation of different nitrogen abatement in the agricultural sector accounting for the dynamics in nitrogen transports in soil and groundwater.

Elofsson (2007) employs a two period model for analysing eventual first mover advantage when abatement costs are stochastic. She shows the existence of second mover advantages since abatement costs are revealed by the first mover. Laukanen and Huhtala (2008) examine optimal abatement of nutrient loading to the Gulf of Finland, a marine basin in the Baltic Sea, from agriculture and municipality waste treatment. A specific feature is the perspective on municipality investment as irreversible. They consider both nitrogen and phosphorus loads, but the latter is translated into a nitrogen equivalence. The results favour investment in municipality treatment, but are sensitive for ecological parameters, such as annual carry over rates of nutrients. The study is extended by Laukanen et al. (2009) by considering phosphorus release from the sediments. The model includes measures in the agricultural sector and investment in municipality waste treatment plants, where the latter is regarded as irreversible. It is assumed that phosphorus is the limiting nutrient and damage costs are then modelled as a function of phosphorus concentrations. The results reveal the important role of immediate investment in municipality treatment plants, but the results are highly sensitive, in particular to changes in the rate of phosphorus release from the sediments.

The Naevdal (2001) and Mäler et al. (2003) studies provide theoretical papers. Mäler et al. (2003) identify optimal policies for an eutrophied lake with a feed back mechanism to changes in external phosphorus loads. The nonlinear dynamic system is much driven by initial pollution accumulation, but it is shown that a constant tax on phosphorus loads may create a clean lake. Naevdal (2001) addresses the dynamic management of eutrophying waters in a theoretical framework. His main objective is to analyse policy implications of threshold effects where pollutant load turns from being beneficial to have deteriorating effects on the water ecosystem. The results point to the advantages of a policy which allows for some fluctuation of pollution concentration around the threshold level

However, none of these studies applied on eutrophication in the Baltic Sea consider both dynamic and spatial dimensions. As noticed by Smith et al. (2009) the dynamic and spatial dimensions are mainly applied on non-renewable resources and species management. One exception is Goetz and Zilberman (2000) who employ a two-stage optimal control problem to solve optimal spatial and temporal loads of fertilisers and manure phosphorus loads to a watershed. The spatial allocation of abatement is solved in the first stage, and the inter temporal problem in the second step. The authors carry out policy simulations and suggest a tax system that varies over space and time.

This paper extends on earlier dynamic specifications of management of eutrophication by adding both nutrients and spatial scales to the dynamic perspective. The choice of both nutrients is justified by the fact the nutrient limiting biological growth differs among basins (e.g. Savchuck and Wulff, 2009). The dispersion of nutrient among marine basins requires a spatial dimension. The long response time to changes in external nutrient load necessitates the dynamic scale. We develop a discrete dynamic model with heterogeneous marine basins which differ mainly with respect to the speed of dynamic processes and to the requirements of nutrient load targets. Cost effective solutions to predetermined targets with respect to nutrient concentration and time frame as expressed by the Helcom BSAP are calculated. The solutions are compared with the nutrient abatement program suggested by Helcom BSAP.

The paper is organised as follows. First, the model for calculating cost effective solutions is presented. Next, data retrieval is described in Chapter 3, and the results with respect to cost effective dynamic and spatial allocation of nutrient abatement are presented in Chapter 4. The paper ends with a brief summary and some tentative conclusions.

2. The model of dynamic and spatial nutrient management

Like many other international marine seas, the drainage basin of Baltic Sea consists of several drainage basins or countries *g* where *g=1,..,n*. For analytical convenience the areas of the drainage basins are assumed to coincide with the territory of the countries. Furthermore, the sea contains *i=1,..,k* different marine basins, which receives nutrient loads from its own drainage basin and also from other marine basins. In each time period *t*, every country discharges nutrients, M_t^{iEg}, where *E=N,P* nitrogen and phosphorus respectively, into different marine basins. The discharge is determined by the business as usual scenario (BAU) load minus abatement. Several types of abatement measures, in particular land use measures, reduce the transport of both nitrogen and phosphorus to the coastal waters of the Sea. Abatement of a nutrient is therefore described as $\beta^{iE} A_t^{ig}$ where β^{iE} is a coefficient relating the measure to reductions in load of nutrient *E*, which differs between countries. Discharges into a marine basin in each time period is then written as $M_t^{iEg} = I_t^{iEg} - \beta^{iE} A_t^{ig}$. The nutrient load to a marine basin *i*, R_t^{iE}, consists of the sum of loads from all riparian countries according to

$$R_t^i = \sum_g M_t^{iEg} \tag{1}$$

It is allowed for growth in BAU loads of nutrients, which can be caused by economic growth, according to

$$I_{t+1}^{gE} = (1+h^g) I_t^{gE} \tag{2}$$

$$I_0^{gE} = \bar{I}^{gE}$$

where h^g is the periodical growth rate in country *g*.

Following Gren and Wulff (2004) and Savchuck (2005) it is assumed that the connections between marine basins can be described by an input-output relation where the time independent coefficient $a^{ijE} = R^{ijE} / R^{iE}$ shows the nutrient transport from basin *i* to basin *j* in relation to total nutrient stock in basin *i*. Total nutrient load to a basin *i*, L_t^{iE}, is then written as

$$L_t^{iE} = \sum_j a^{jiE} R_t^j \qquad (3)$$

The response mechanisms and time required for adjustments to changes in external nutrient loads differ between the basins and nutrients. Nitrogen is mainly transformed into harmless nitrogen gas and assimilated by plants, but can also be supplied to the Baltic Sea by nitrogen fixating algal. Phosphorus is also assimilated by plants, but is also deposited on the sediments which can be released under suitable oxygen conditions. These adjustment mechanisms in the Sea to changes in nitrogen and phosphorus loads from the drainage basins may result in a non-linear system with associated difficulties of identifying optimal abatement paths (e.g. Mäler, 2000). Furthermore, the responses of nitrogen and phosphorus are connected. For example, reductions in phosphorus loads may increase the growth of nitrogen fixating algal (e.g. Savchuck and Wulff, 2009). However, simplifications are made by assuming linear and separate relations between stock of nutrient *E* in period *t+1* in basin *i*, S_{t+1}^{iE}, which is written as

$$S_{t+1}^{iE} = (1-\alpha^{iE})(S_t^{iE} + L_t^{iE}) \quad \text{for } i=1,..,k \text{ and } E=N,P \qquad (4)$$

$$S_0^{iE} = \overline{S}^{iE}$$

where $0 \le \alpha^{iE} \le 1$ is the self cleaning capacity in basin *i* of nutrient *E*.

Following the ministerial agreement on maximum nutrient loads from 2007, the nutrient targets are defined for different marine basins and nutrients as maximum nutrient concentrations (Helcom, 2007). The nutrient stock equations is then measured in terms of nutrient concentrations by multiplying the right hand side of (4) by a factor which includes water volume and atom weights of the nutrient, W^{iE}. The target in the terminal period *T* is then written as

$$((1-\alpha^{iE})(S_T^{iE} + L_T^{iE}))W^{iE} \le K^{iE} \qquad \text{for } i=1,..,k \text{ and } E=N,P \qquad (5)$$

Solving for S_T in (4), the restriction in (5) can be more explicitly written in terms of the dynamic and spatial connections among marine basins as

$$((1-\alpha^{iE})^T S_0^{iE} + \sum_{t=0}^{T} (1-\alpha^{iE})^{T-t} \sum_j a^{jiE} R_t^{jE}) W^{iE} \leq K^{iE} \qquad (5')$$

Equation (5') shows that the nutrient concentration in a basin *i* is determined initial conditions, carry over rates of nutrients in marine basins, and transports of nutrient loads among marine basins.

For each drainage basin there exists an abatement cost function $C^g(A_t^{gN}, A_t^{gP})$ which is positive and convex in A^{gN} and A^{gP}. The decision problem is now specified as the minimisation of total control cost for achieving the targets defined by (1)-(5), which is written as

$$\underset{S_t^{iE},\, A_t^{gE}}{Min} \quad \sum_{t=0}^{T} \sum_g \sum_E C^g(A_t^{gE}) \rho_t \qquad (6)$$

s.t. (1)-(5)

where $\rho_t = \dfrac{1}{(1+r)^t}$ is the discount factor with *r* as the discount rate. In order to solve the decision problem defined by (1)-(6), we formulate the Lagrangian expression as

$$L = \sum_t \sum_E \rho_t [\sum_g (C^g(A_t^{gE}) + \rho\mu_{1+1}^g (I_{t+1}^{iEg} - (1+h^g) I_t^{iEg})) \qquad (7)$$

$$+ \sum_i W^i \rho \nu_{t+1}^i (S_{t+1}^{iE} - (1-\alpha^{iE}) S_t^{iE} + L_t^{iE})]$$

where μ_{t+1}^g and ν_{t+1}^i are the Lagrange multipliers for equations (2) and (4). The necessary conditions for optimality deliver

$$\frac{\partial L}{\partial A_t^{gE}} = \rho_t \left[\frac{\partial C^g}{\partial A_t^{gE}} - \sum_E \sum_j \rho \nu_{t+1}^{jE} a^{ijE} \beta^{gE} \right] = 0 \qquad (8)$$

$$\frac{\partial L}{\partial I_t^{iEg}} = \rho_t (\mu_t^g - \rho(1+h^g)\mu_{t+1}^g + \sum_j \nu_{t+1}^{jE} a^{ijE}) = 0 \qquad (9)$$

$$\frac{\partial L}{\partial S_t^{iE}} = W^i(v_t^{iE} - \rho v_{t+1}^{iE}(1-\alpha^{iE})) = 0 \tag{10}$$

$$\frac{\partial L}{\partial(\rho\mu_{t+1}^g)} = \rho_t(I_{t+1}^{iEg} - (1+h^g)I_t^{iEg}) = 0 \tag{11}$$

$$\frac{\partial L}{\partial(\rho v_{t+1}^i)} = W^i(S_{t+1}^{iE} - (1-\alpha^{iE})S_t + L_t^{iE}) = 0 \tag{12}$$

Condition (8) simply states that, in optimum, marginal cost of nutrient reduction in a country *g* equals the sum of Lagrange multipliers, v_{t+1}^{iE}, times the coefficients describing transports from basin *i* to basins *j*. This condition ensures spatial cost effectiveness in each time period. This is most easily seen by assuming that the target is binding only for one basin *j*, and also for only one nutrient. Condition (8) is then reduced to

$$\frac{1}{a^{jjE}\beta^{jEg}}\frac{\partial C^g}{\partial A_t^g} = \rho v_{t+1}^{jE} \tag{8'}$$

which shows that the marginal abatement cost in each country *g* adjusted by the impact on basin *j*, the left hand side of (8'), equals the present value of the Lagrange multiplier. Expression (8') can also be used to illustrate cost effective design of economic instruments. It shows that the pollution charge in a drainage basin in period *t*, k^{gE}, is determined by

$$k_t^{gE} = \frac{\partial C^g}{\partial A_t^{g*}} = a^{jjE}\beta^{jEg}\rho v_{t+1}^{jE} \tag{8''}$$

From (8") it can be seen that the larger the impact on the marine basin, the higher is the cost effective charge. In a similar vein, optimal trading ratios of permits between any two countries, d_t^{hl}, under a market system can be derived from (8') as

$$d_t^{hl} = \beta^{jEh} / \beta^{jEl} \tag{8'''}$$

It is then assumed that the optimal nutrient load to the basin is distributed as permits to the countries in each time period without any banking or borrowing options between time periods (for a discussion of these options see e.g. Hagem and Westskog 2008). According to eq. (8''') the optimal trading ratio is determined by the relative impacts on basin *j* from abatement in country *h* and *l* respectively. When $d_t^{hl} > 1$ the impact is higher from country *h* and a pollution permit in this country accrues a higher value than that in country *l*.

Similarly eq. (10) generates dynamic efficiency, which can be seen by assuming binding constraint for only one basin, according to

$$\nu_t^{iE} = \nu_{t+1}^{iE} \frac{(1-\alpha^{iE})}{1+r} \tag{10'}$$

When (10') holds there is no room for net savings in costs by allocation abatement among periods since the marginal costs for achieving the marine basin target are equal as expressed in present value terms and impact on the target. The denominator translates future marginal cost into present terms and the numerator reflects the higher impact of future cleaning due to the existence of earlier period's self cleaning capacity. That is, expenses for early abatement are partly a waste of resources since part of that cleaning would have taken place by the sea itself.

The combined spatial and dynamic efficiency is illustrated under assumptions of binding constraints for two marine basins and one nutrient by inserting condition (8) into (10), which gives

$$\nu_t^{iE} = \frac{(a^{jj} C_{A_t^{iE}}^{iE} - a^{ij} C_{A_t^{jE}}^{jE})(1-\alpha^{iE})}{\rho(a^{ii}a^{jj} - a^{ij}a^{ji})} \qquad \text{for } i,j=1,2 \tag{13}$$

Expression (13) shows that the Lagrange multiplier, or the marginal cost for reaching nutrient load to basin *i*, is determined by the transport coefficients and the marginal abatement cost for the other basin *j*. The effect of increases in marginal costs in the other basin *j* on ν_t^{iE} is undoubtedly negative. This is also true for changes in ρ, a^{ii}, and α^{iE}. The effects of the other transport coefficients are indeterminate since these implies changes in loads to basin *j* and subsequently to basin *i*.

Equation (9) shows the effects on total minimum costs of changes in initial nutrient loads, which are determined by the marginal cost for achieving the targets, the growth in load and the discount rate.

3. Data retrieval

In principle, the empirical model builds on the static and spatial model developed in Gren et al. 1997 and Elofsson (1999). This means that different drainage basins are characterised by nutrient emission sources, leaching and transports to coastal waters (see Table A3 in the Appendix). Both cost functions and nutrient loads under business as usual are obtained from an existing static programming model of the Baltic Sea (Gren et al. 2008). For the purpose of matching data on costs of different abatement measures with nutrient transports in soil and water, the entire basin is divided into 24 drainage basins see Figure A1 in Appendix A. Nutrient transports from sources and costs of abatement measures are calculated for each of these drainage basins, which are briefly presented in this chapter. Unless otherwise stated, all data and calculations are found in Gren et al. (2008).

3.1 Cost functions

A pseudo data approach is used for estimating cost functions for nutrient reductions (see e.g. Griffin, 1978). Unlike traditional sources, such data sets are not constrained by historical variations in, for example, factor prices and yields from land affecting land prices. Observations on costs and nutrient reductions are then obtained by calculating minimum cost solutions for different levels of nutrient reductions to the coastal waters from each drainage basin. A cost function in nitrogen and phosphorus abatement is then obtained for each drainage basin and it is assumed that cost effective reductions are implemented in each drainage basin.

The cost minimization model includes 12 different measures for nitrogen reduction and 10 abatement measures for phosphorus reductions (Gren et al. 2008). Since the agricultural sector accounts for approximately 60 percent and 50 percent of nitrogen and phosphorus loads respectively, the majority of the measures affect this sector. Abatement measures reducing airborne emissions and sewage from household and industry are also included. The included measures are listed in Table A1 in the Appendix.

For each of these abatement measures, costs are calculated which do not include any side benefits, such as provision of biodiversity by wetlands. Furthermore, abatement measures located in the drainage basins may have a positive impact on water quality, not only in the Baltic Sea, but also in ground and surface waters. However, such data on side benefits are not available for the included abatement measures. This implies an overestimation of abatement costs of measures implemented in the drainage basins. On the other hand, the cost estimates do not account for eventual dispersion of impacts on the rest of the economy from implementation of the measure in a sector, such as eventual increase in prices of inputs of a simultaneous implementation of improved cleaning at sewage treatment plants.

The static model applies two methods for estimation of costs of the different abatement measures – partial equilibrium and engineering methods – which differ with respect to consideration of affected sectors' actual behaviour in the market. Partial equilibrium analysis is applied for calculations of costs of reductions in fertilisers, which rests on revealed behaviour on the fertiliser market. Data on prices and purchases of fertiliser are then used for deriving costs of fertiliser reductions, which correspond to associated losses in profits. Market prices are also used for assessing costs of conversion of arable land into less leaching land uses such as wetlands and buffer strips. However, there is not enough data to evaluate the effect of massive land conversion on the market price of arable land, and the engineering method is therefore applied for cost calculations. The engineering method assumes no changes in prices and constant unit abatement costs are then calculated, which result in linear cost curves as compared to the convex cost functions for fertilizer reductions. Due to lack of data, the engineering approach is used for calculating costs of, not only land use changes, but also of costs of all other abatement measures except for reductions in fertilisers.

Some of the measures included in the static programming model affect both nitrogen and phosphorus loads, which implies jointness in the abatement costs of nitrogen and phosphorus. Calculations are made for all even reduction levels between 2 and 60 per cent for each drainage basin, which gives 30 observations for each drainage basin. Unfortunately, it is not possible to regress the joint cost functions including both nutrients due to singularity. In order to estimate the impact on costs from joint abatement of nitrogen and phosphorus by, in particular, land use measures a three step approach is therefore applied:

1) separate regression of N and P reductions which gives the cost functions
   $C^{giN} = a^{gi}(N^{gi})^2 + \varepsilon^{giN}$
   $C^{giP} = b^{gi}(P^{gi})^2 + \varepsilon^{giP}$
   where N^{gi} and P^{gi} are measured in thousand tons and costs in millions of SEK,
2) generation of data for estimation of minimum costs for reductions of both nitrogen and phosphorus, estimation of the difference between the sum of costs of separate and simultaneous nutrient reductions according to $\Delta C^{gi} = C^{giN} + C^{giP} - C^{giNP}$ and carry out regression of the difference as $\Delta C^{gi} = c^{gi}(N^{gi}P^{gi})^2 + \varepsilon^{giNP}$,
3) combination of 1) and 2), which gives the joint cost function as
   $C^{giNP} = a^{gi}(N^{gi})^2 + b^{gi}(P^{gi})^2 - c^{gi}(N^{gi}P^{gi})^2$

Ordinary least square estimates are used for estimating cost functions for all three regression equations - nitrogen and phosphorus separately and simultaneously – and for each drainage basin, which are presented in Table A2 in the Appendix. As shown in Table A2 the estimated regression equations for ΔC^{gi} give the worst goodness of fit. The estimated c^{ig} coefficients are therefore calibrated in order to obtain costs at different reduction levels obtained from the optimisation model (see Table A2 in the Appendix).

There is a large literature on the appropriate level of the discount rate (see e.g. Weitzman, 2001). The level of the discount rate may also differ between the riparian countries due to different forecasts of economic growth, technological development etc. A strong simplification is made in this paper by assigning a uniform periodical discount rate for all countries. Calculations of cost effective solutions are made for two different levels; 0.02 and 0.03.

3.2 Nutrient loads and dynamics

Nutrient loads to the Baltic Sea from the drainage basins reported in Chapter 3.1 are, for all emission sources, calculated by means of data on emissions, which are sufficient for sources with direct discharges into the Baltic Sea, such as industry and sewage treatment plants located by the coast and air deposition on the sea. For all other sources further data are needed on the transformation and burial of nutrients from the emission source to the coastal waters. This requires data on transports of airborne emissions among drainage basins, leaching and retention for all sources with deposition on land within the drainage basins, and on nutrient retention for upstream sources with discharges into water streams. Such load calculations are obtained from Gren et al. (2008), which are reported for the different drainage basins in table A1 in the Appendix.

In principle, the biogeochemical process controlling large-scale eutrophication can be described by the nutrient dynamics within and between the major marine basins of the Baltic Sea (see Figure A1 in Appendix for a map of the seven drainage basins used in this study). Nutrient concentrations are determined by nutrient loads to the water, water volume, primary production, nitrogen fixation, denitrification, pelagic recycling, sediment release and burial (Savchuck and Wulff, 2009). Nutrient fluxes between and within the marine basins are estimated on budgets of nutrient transports during the period 1991-1999 which takes into account all these processes (Savchuck, 2005). Table 1 reports initial nutrient loads from the drainage basins to the marine basins and the initial stock of nutrients in each basin, i.e. $R^i = \sum_g \bar{I}^{iEg}$ and $\bar{S}^{iE}$ respectively from Chapter 2.

Table 1: Initial nutrient loads from drainage basins and stocks in marine basins, thousand tonnes

	Initial annual nutrient load[1]		*Initial nutrient stock* [2]	
	N	*P*	*N*	*P*
Bothnian Bay	23	0.63	445	10.6
Bothnian Sea	26	0.85	1236	53.5
Baltic Proper	578	25.5	3908	418
Gulf of Finland	137	5.37	394	25.2
Gulf of Riga	45	2.67	208	12.7
Danish Straits	61	1.03	114	10.4
Kattegat	43	0.70	132	11.6

1) Table A3 in Appendix, 2. Savchuk (2005), Figure 4

Table 1 reveals the dominant role of Baltic Proper with respect to initial loads and stocks of both nutrients. The large difference between nutrient loads and stock is also evident. This is a result from the dynamics of the Baltic Sea where some marine basins are more slow than others in processing nutrients. Another important factor is the interconnectedness of the marine basins, where the stock of nutrients in one marine basin depends on nutrient loads to and stocks in other marine basins.

The time required for full response to changes in the external nutrient loads differ among the marine basins. According to Savchuck and Wulff (2007), the response time for changes in phosphorus and nitrogen loads to the Baltic Proper approximates 60-70 years. However, there are no publications of systematic calculations of response times to changes in nutrient loads for all basins. Different sources are therefore used for assessing carry over rates in the seven marine basins. Furthermore, in order to obtain tractable solutions each period is divided into five years. The applied carry over rates in the reference case are reported in Table 2.

Table 2: Periodical carry over rates for nitrogen and phosphorus in the marine basins of the Baltic Sea

	Carry over rate, N	Carry over rate, P
Bothnian Bay[3]	0.70	0.85
Bothnian Sea[3]	0.70	0.85
Baltic Proper[1]	0.60	0.85
Gulf of Finland[2]	0.25	0.50
Gulf of Riga[3]	0.25	0.50
Danish Straits[3]	0.10	0.30
Kattegat[3]	0.10	0.30

1. Based on Savchuck and Wulff (2009);
2. Based on Laukanen and Huhtala (2008)
3. Calculated from half time response obtained from Mare Nest

The Bothnian Bay, Bothnian Sea and Baltic Proper are the slowest basins with respect to the processing of both nitrogen and phosphorus.

The interconnections among marine basins are described by an input-output relation for each of the nutrients, which is estimated at the steady state levels of nutrient dynamics in the Baltic Sea (Savchuck, 2005). Table 3 shows the input-output matrix for nitrogen, where the columns show the allocation of nitrogen into the row basins. For example, one unit reduction in the load from the drainage basin to the Bothnian Bay generates a final reduction by 1.106 in the own basin, 1.118 in Bothnian Sea, 0.919 in Baltic Proper, 0.074 in Gulf of Finland, and so on. Changes in any basin imply effects in nitrogen load to all other basins.

Table 3: Input-output coefficients for nitrogen transports among marine basins, from column basins into row basins.

	Bothnian Bay	Bothnian Sea	Baltic Proper	Gulf of Finland	Gulf of Riga	Danish Straits	Kattegat
Bothnian Bay	1.106	0.124	0.028	0.02	0.015	0.012	0.002
Bothnian Sea	1.118	1.306	0.294	0.206	0.163	0.124	0.025
Baltic Proper	0.919	1.074	1.454	1.016	0.804	0.614	0.126
Gulf of Finland	0.074	0.086	0.117	1.081	0.065	0.049	0.010
Gulf of Riga	0.023	0.026	0.036	0.025	1.02	0.015	0.003
Danish Straits	0.258	0.302	0.409	0.285	0.226	1.297	0.265
Katte gat	0.140	0.163	0.221	0.154	0.122	0.702	1.144

Source: Savchuk (2005), Table 3

Similar to dispersion of impacts of changes in nitrogen load to one basin, alterations in phosphorus inputs to one basin have effects on all other basins, see Table 4. It is interesting to note that the effects on the own basin is for all basins but Bothnian Bay larger for phosphorus than for nitrogen load changes.

Table 4: Input-output coefficients for phosphorus transports among marine basins

	Bothnian Bay	Bothnian Sea	Baltic Proper	Gulf of Finland	Gulf of Riga	Danish Straits	Kattegat
Bothnian Bay	1.034	0.096	0.069	0.046	0.053	0.029	0.006
Bothnian Sea	0.540	1.526	1.089	0.729	0.837	0.464	0.099
Baltic Proper	0.412	1.162	2.517	1.685	1.934	1.072	0.230
Gulf of Finland	0.075	0.212	0.459	1.307	0.353	0.196	0.042
Gulf of Riga	0.023	0.065	0.141	0.094	1.108	0.060	0.013
Danish Straits	0.265	0.747	1.619	1.084	1.244	1.821	0.390
Katte Gat	0.144	0.406	0.878	0.588	0.675	0.988	1.212

Source: Savchuk (2005), Table 4

However, the input output coefficients reported in Tables 3 and 4 reflect the final impacts after all adjustments have taken place. It is assumed that the final dispersions shown in Tables

3 and 4 are reached during the periods as described by the carry over rates shown in Table 2. For example, the carry over rate for phosphorus in Baltic Proper is 0.85. This means that 0.85 times the column coefficients for Baltic Proper in Table 4 are obtained in period 1, 0.85^2 in period 2, and so on

3.3 Determination of nutrient load targets

The basis for target setting in this paper is the most recent ministerial agreements on nutrient load restrictions for different marine basins of the Baltic Sea, the so-called Baltic Sea Action Plan (Helcom, 2007). BSAP suggest the following nutrient related ecological objectives

- concentration of nutrients close to natural levels,
- clear water,
- natural level of algal blooms,
- natural distribution and occurrence of plants and animals,
- natural oxygen level.

Conditions for the achievements of these targets differ among different parts of the Baltic Sea, but it is regarded that reductions in nutrient loads improve water quality in most parts of the Sea. According to BSAP phosphorus reductions are required to Baltic Proper, Gulf of Finland and Gulf of Riga, and nitrogen reductions to Baltic Proper, Danish straits and Kattegat. Phosphorus reductions, as measured in percent reductions from initial modelled loads, are largest for the Baltic Proper, and the largest nitrogen reductions are needed for Kattegat and the Danish Straits. It is predicted that these reductions will reduce the extension of hypoxic sea bottoms in the Baltic Proper by approximately 1/3, and nitrogen fixation, an indicator of the intensity of cyanobacterial blooms, is expected to decrease by 2/3. However, there is no clear correlation between these targets and nutrient loads. We have therefore used nutrient concentration as targets, since these are highly correlated with the ecological targets (Helcom 2007).

In principle, measurements of nutrient concentrations can be obtained by dividing the steady state nutrient loads in the marine basins by their content of water, which can be converted into μM/l by use of the atom weights of N and P. Division of the nutrient stocks reported in Table 1 with water volume in each basin generates nutrient concentration levels in the reference case. Corrections of these estimates with the atom weights for nitrogen and phosphorus express the concentrations in terms of μM which is common among marine scientists. The target concentrations are found in Helcom (2007). The water volumes, and nutrient concentrations in the reference and target cases are listed in Table 5.

Table 5: Water volume, and nutrient concentrations in the reference and target cases

	Volume[1], *Km*^{3}	*Reference case*[2]		*Target*[3]	
		TN, μM	*TP μM*	*TN, μM*	*TP, μM*
Bothnian Bay	1400	22.70	0.245	22.6	0.2
Bothnian Sea	4400	20.06	0.466	19.5	0.4
Baltic Proper	13000	21.47	1.039	18.5	0.5
Gulf of Finland	1000	28.14	0.815	24.7	0.6
Gulf of Riga	400	37.14	1.026	41.7	0.7
Danish Straits	300	27.14	1.121	19.9	0.6
Kattegat	530	17.79	0.707	17.0	0.6

1. Savchuck and Wulff (2009)
2 Calculated by dividing nutrient stock in Table 1 with water volume and correcting for the nitrogen
and phosphorus atom weights
3. Helcom (2007) Table 2 page 5

Table 5 reveals the needs of reductions in phosphorus concentration for all basins, and in nitrogen concentration for Baltic Proper, Gulf of Finland, Danish Straits and Kattegat. However, the Helcom BSAP excludes Bothnian Bay and Bothnian Sea as target basins, and suggests reductions in phosphorus loads only to three basins; Baltic Proper, Gulf of Finland, and Gulf of Riga (see Table A4 in the Appendix for suggested load reductions). Four marine basins are targeted for nitrogen reductions; Baltic Proper, Gulf of Finland, Danish Straits and Kattegat. In order to compare the dynamic cost effective solutions obtained in this paper with the BSAP suggestion, we define the same target basins as Helcom BSAP.

Finally there is a need for defining the targeted time period when the improvements are to be achieved. These are in turn determined by the timing of implementation of the measures and the response time in the marine basins. Helcom BSAP suggests 2021 to be the deadline for implementation of nutrient load reductions. A response time of 60 years, which is assumed for phosphorus reductions to the Baltic Proper, gives a target date at approximately 2080 in the reference case.

4. Dynamic cost effective achievement of the BSAP

As described in Chapter 2, it is assumed that cost effective allocations of abatement measures are implemented in each drainage basin. The cost effective solutions in this paper then generate optimal allocation among time periods and drainage basins. The GAMS Conopt2 code is used for solving the problem (Brooke et al. 1998). In order to obtain tractable solutions, the entire period is divided into 20 periods where each period corresponds to 5 years. Costs are calculated mainly for the reference case which is characterised by the initial nutrient load and stock as reported in Table 1, carry over rates shown in Table 2, discount rate of 0.03 and the target year of 2080. Costs are also calculated for other targets dates, both earlier and later. Furthermore, two different periodical discount rates – 0.02 and 0.03 – are used. Calculations are also made for assumption of 0.005 per cent periodical growth in BAU for all countries.

4.1 Dynamic and national allocation of costs

In Figure 1 we present results of minimum cost solutions for obtaining the BSAP targets on nutrient concentrations, i.e. phosphorus concentrations in the Baltic Proper, Gulf of Finland, and Gulf of Riga, and nitrogen concentrations in the Baltic Proper, Gulf of Finland, Danish Straits and Kattegat to be achieved at the latest in 2080, i.e. 70 years from first implementation period in 2010, which constitutes the reference case. Results are also presented for earlier and later target dates.

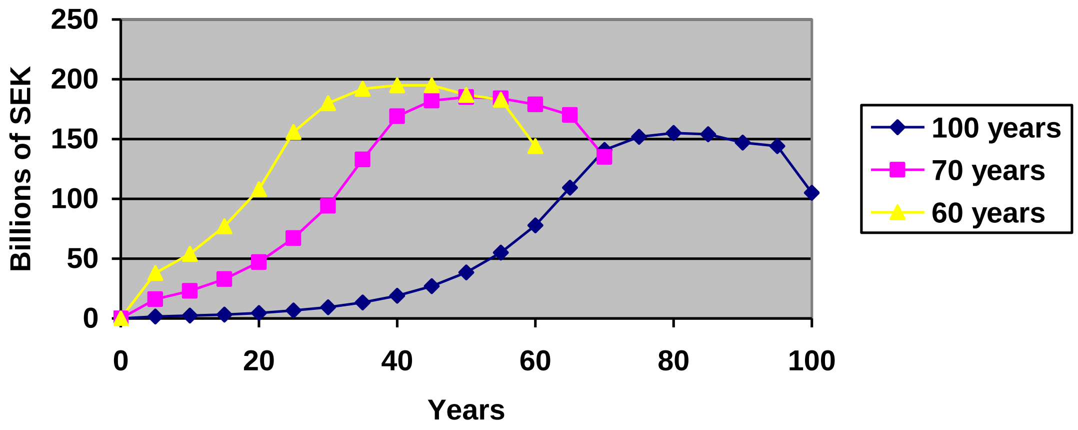

Figure 1: Discounted costs for achievements of nutrient targets for marine basins in three different time periods in the reference case

As expected from the theoretical results in Chapter 2, abatement expenses are delayed as much as possible. For all time frames, there is a peak in abatement costs approximately 20 years before the target time. In spite of these differences in abatement costs during periods, there is a steady decrease in phosphorus concentration, µM, during years. This is shown for concentrations in the Baltic Proper, for the reference scenario in Figure 2.

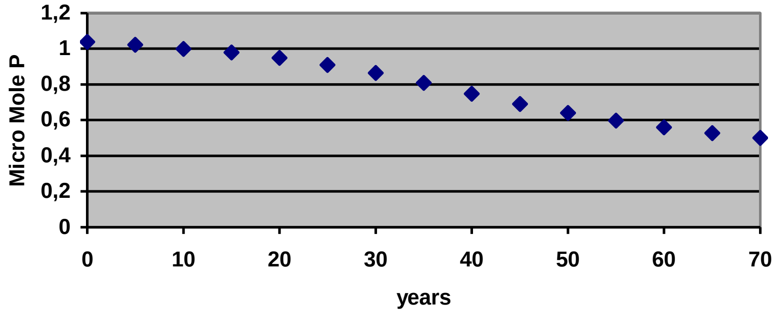

Figure 2: Optimal path of phosphorus concentration in the Baltic Proper

Along the cost effective path, there is a large difference in financial burdens among countries, where Poland carries the largest burden and Germany the lowest costs, Figure 3.

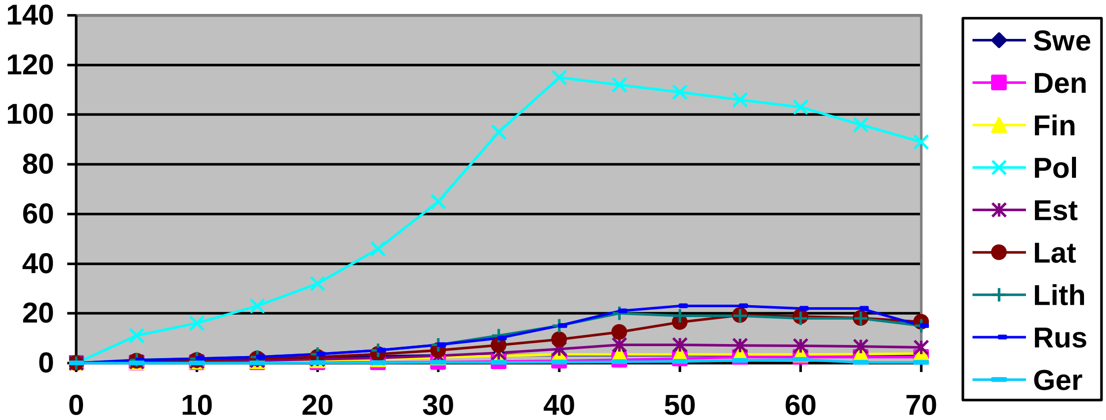

Figure 3: Allocation of discounted abatement costs among countries under the reference case. (Swe, Sweden; Den, Denmark; Fin, Finland; Pol, Poland; Est, Estonia; Lat, Latvia; Lit, Lithuania; Rus, Russia; Ger, Germany).

The control cost for Poland reaches its peak 40 years after implementation and then declines. The pattern is similar for all countries, but the peak is reached at later periods. The reason is the difference in response times for the two nutrients and between basins. Poland discharges effluents only into Baltic Proper which has a relatively slow adjustment process of both phosphorus and nitrogen. The shorter response time for nitrogen, the later is the abatement implemented.

4.2 Examples of policy design

There is a large literature on the efficient and appropriate design of policy instruments for international environmental problem (see e.g. Kolstad and Toman, 2005). Most of this is applied to climatic change, and very few on international waters (Elofsson et al., 2003). However, a common finding of all applied literature is the advantages of international economic instruments for the achievement of overall cost effectiveness. In principle, there are two types of economic instruments; charge/subsidy systems and pollution permit markets. Both these systems need periodical adjustments in order to obtain dynamic cost effectiveness; the instrument becomes more stringent during time due to the need of larger abatement. This means higher charges in current terms or decreasing size of the permits issuance under a market scheme. If the market functions perfectly, the equilibrium permit price is the same as the charge.

Recall from the theoretical Chapter 2, that charges and size of permit markets are determined in a multi-basin target setting, which generally creates a quite complex policy design problem. However, in the particular case of the Baltic Sea, policy design is much simplified due to the BSAP's relatively large requirement of decreases in phosphorus concentration in the Baltic Proper (see Table 5), and the linear spread of impacts among basins in each time period. The restriction on maximum phosphorus concentrations in the Baltic Proper then becomes the only binding constraint. Achievement of this target implies the fulfilment also of other targets due to the impact of both nitrogen and phosphorus loads of abatement, to the slow phosphorus dynamics in Baltic Proper, and to the linear spatial dispersal of nutrients among basins. As shown by eq. (8") in Chapter 2, the optimal charges on phosphorus load along the cost effective path for all other basins than Baltic Proper are then determined by their impact on phosphorus concentrations in the Baltic Proper. Such effluent charges are presented in Figure 4.

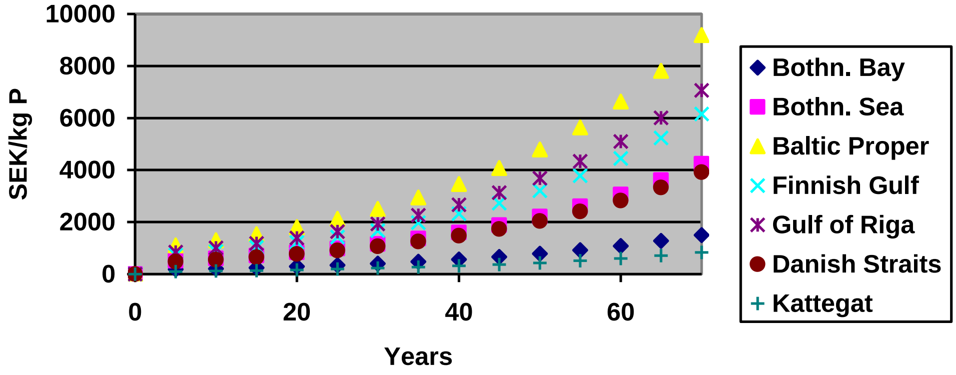

Figure 4: Charges on phosphorus loads in current values to different marine basins in current along the cost effective path

As shown in Figure 4, charges are introduced in all periods in all basins, but at different levels. The relations between charges among basins are determined by the input-output coefficients presented in Table 4. The smaller the impact on the phosphorus loads to the Baltic Proper the lower is the charge. Figure 4 shows that next to Baltic Proper, the highest charges are implemented for loads to the Gulf of Riga, and the lowest charge levels for phosphorus loads to Kattegat.

Under a permit market system, the trading ratios between loads to the Baltic Proper and other basins are determined by the input-output coefficients presented in Table 4. This means, for example, that the trading ratios for loads to the Finnish Gulf and Gulf or Riga are 0.67 and 0.77 respectively. Under perfect functioning of the phosphorus market in Baltic Proper, the equilibrium prices in current values are the same as the charge levels presented for Baltic Proper in Figure 4. In order to achieve these prices, the phosphorus load bubble for the Baltic Proper declines over time as shown in Figure 5.

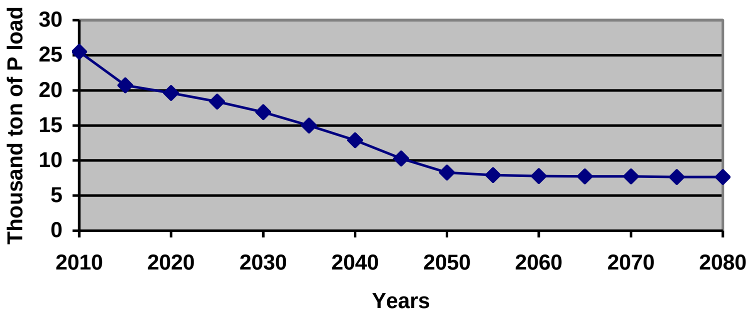

Figure 5: Size of the market for phosphorus loads to the Baltic Proper during years for achievement of the BSAP in year 2080.

The bubble size path can be divided into two main periods; a relatively rapidly shrinking bubble during the first 40 years, and then small adjustments during the next 30 periods until the achievement of the final target of 7.65 Kton of phosphorus loads. The relatively slow dynamic process of the Baltic Proper calls for this early rapid decline in the size of the phosphorus bubble.

4.3 Sensitivity analysis

The calculated costs presented in Chapter 4.1 and the policy design shown in Chapter 4.2 are, however, sensitive to assumptions made, in particular, on the carry over rate of nutrient from one period to another. This is illustrated in Figure 6, where costs are calculated for achievement of the BSAP targets within different target years, discount rates, and growth in BAU nutrient loads.

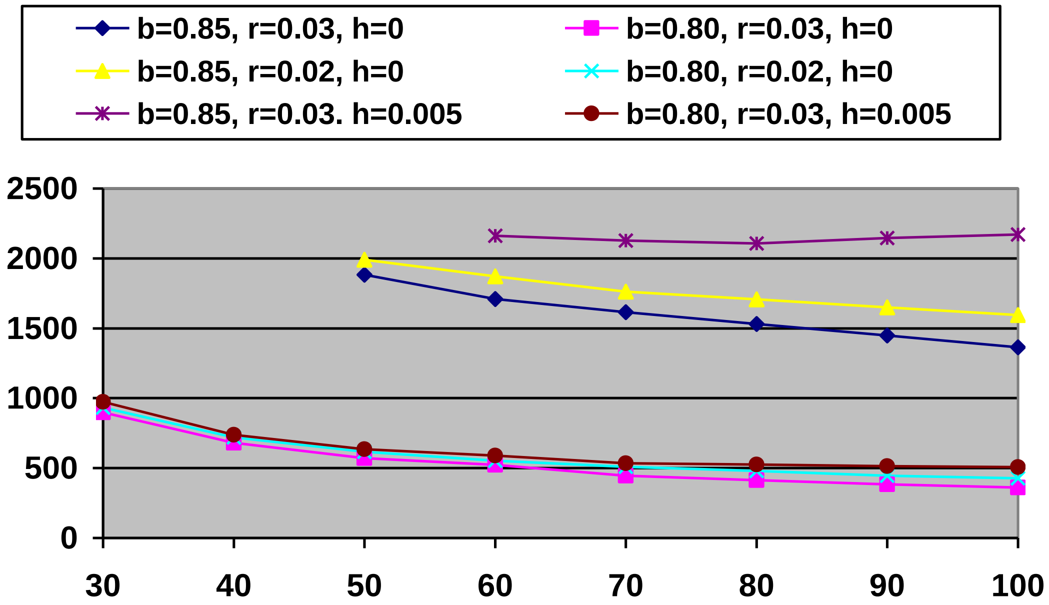

Figure 6: Total costs for different target years under different assumptions on carry over rate of phosphorus in the Baltic Proper, b, discount rate, r, and growth rate in nutrient loads, h.

When the carry over rate is 0.85, i.e. b=0.85, the BSAP target can not be achieved prior to 50 years after the implementation of the program. There is a slight increase in costs for all target years when the discount rate decreases from r=0.03 to r=0.02. The impact on costs is higher from a change in growth rate of BAU loads from h=0 to h=0.05. It is interesting to note that the declining cost for future target years then are counteracted by the growth in BAU load, which results in a small increase in total cost for the last target years. Another implication of the growth in BAU loads is that the BSAP target requires 60 instead of 50 years to be reached.

The results presented in Figure 6 show that the assumption of the carry over rate has the largest influence on costs irrespective of target year. The decline in carry over rate from b=0.85 to b=0.8 reduces total costs by approximately one half. Another noteworthy result is that the increase in BAU loads has considerable impacts when b=0.85 but only a slight cost increasing effect when b=0.80. The reason is that the large inherited nutrient stock (see Table 1 in Chapter 3.2), which implies a relatively large decline in the stock of phosphorus during a period also for a slight decrease in the carry over rate.

5. Comparison with the BSAP suggestion

If and how does the Helcom BSAP differ from the dynamic and spatial cost effective solutions presented in Figures 1-6? In order to answer this question there is a need for a more precise interpretation of the policy suggestions in the BSAP. It is stated that the abatement measures leading to reductions should be implemented at the latest at 2021, which is then interpreted as the time target. Since no change in total load during time is discussed at all, it is here understood that the suggested reduction, or maximum load, should be obtained every year. BSAP also suggests allocation of cleaning among countries in each period. According to SEPA (2009), estimated annual cost of this country allocation amounts to 49.6 millions of SEK, and is divided among countries as shown in SEPA (2009) Table 16 page 48. Assuming that this scheme is introduced in 2021, i.e. the third period, the estimated costs and allocation among countries are as reported in Table 6, where the corresponding allocations are shown for cost effective solutions.

Table 6: Total cost in present terms, % of total cost, and annual cost for the riparian countries under the BSAP suggestion and the cost effective program for achieving the targets in 2080.

	BSAP: Total, bill sek	% of total cost	Annual cost, bill sek[1]	*Cost effective solution:* Total, Bill sek	% of total cost	Annual cost, bill sek[2]
Sweden	51	2	0.9	36	2	0.5
Denmark	102	5	1.8	18	1	0.3
Finland	18	1	0.3	35	2	0.5
Poland	1511	70	25.2	1015	63	14.5
Estonia	28	1	0.5	60	4	0.9
Latvia	139	6	2.3	133	8	1.9
Lithuania	125	6	2.1	142	9	2.0
Russia	116	5	1.9	173	11	2.5
Germany	74	3	1.2	5	0[3]	0.1
Total	2163	100	36.1	1617	100	23.1

1. The entire period is approximately 60 years due to BSAP implementation in 2021. 2. Implementation 2010 gives a period of 70 years. 3. The cost corresponds to 0.3 per cent of total cost, which becomes 0 when rounded.

Total costs under the BSAP suggestion is approximately 30 per cent higher than the cost effective solution. There are two sources of inefficiencies for this differences; inefficient allocation of cleaning among countries and among time periods. In Gren (2008) and SEPA (2009) it is demonstrated that total abatement cost of the BSAP is considerably larger than that of cost effective solution in a static perspective. The implementation of uniform cleaning over years is also inefficient since cost can be reduced by adjust implementation according to discount rate and nutrient carry over rates for different marine basins as illustrated in Figures 1-3.

However, some countries gain from the implementation of BSAP as compared with the cost effective solution; Finland, Estonia, Lithuania and Russia. The other countries face lower cost burdens under the cost effective solution. Although Poland has considerably larger total cost under BSAP, the percentage share of total cost is similar to that under the cost effective solution. Russia faces the largest change in relative costs.

6. Conclusions

The purpose of this paper has been to present a numeric dynamic model of cost effective nutrient management in the Baltic Sea. The model was constructed on the basis of four different stylized facts:

- both nitrogen and phosphorus loads cause damages from eutrophication
- the Baltic Sea consists of a different coupled marine basins
- the response times to changes in external nutrient loads differ in the marine basins
- abatement costs vary among the littoral countries due to differences in geographical and climate conditions, technologies, and economic conditions

Owing to these facts, the numerical discrete dynamic model consists of three main components; cost functions for nitrogen and phosphorus reductions for each of the 24 drainage basins in the Baltic Sea drainage basins, coupled seven marine basins, and nutrient dynamics of each marine basin. Admittedly, in order to keep the model tractable, simple linear relations were assigned for each marine basin's carry over of nutrient from one period to the next. These carry over rates were allowed to vary among the marine basins and for phosphorus and nitrogen. Phosphorus dynamics are more slow than that of nitrogen as expressed by higher levels of carry over rates among periods.

The model was applied for the calculation of costs, in total, for different countries and during time, for cost effective achievement of the nutrient concentration targets expressed by Helcom BSAP. The BSAP suggests nutrient concentrations targets to be achieved in different marine basins. However, the time target for achievement of the ecological goals is not stated explicitly. Costs for different time perspective were therefore calculated, and the estimated costs differed considerably depending on time for achievement of the targets. However, a common result for all simulations was the need to focus on policy design for only one target on one marine basin, phosphorus concentrations in the Baltic Proper. The reasons were the relatively stringent target for this basin, the slow dynamics, and the spread of impact to other basins from Baltic Proper. Thus, a seemingly complex policy design problem turns out to be relatively simple when considering only cost effectiveness. However, accounting for other criteria for successful implementation of international policies, such as perceived fairness, may contradict this simplicity in practice.

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Table A1: Abatement measures in the drainage basins of the Baltic Sea

N reduction (12 measures)	*P reduction (10 measures)*
Selective catalytic reduction (SCR) on power plants	
SCR on ships	
SCR on trucks	
Reductions in cattle, pigs, and poultry	Reductions in cattle, pigs, and poultry
Fertilizer reduction	Fertilizer reduction
Increased cleaning at sewage treatment plants	Increased cleaning at sewage treatment plants
Private sewers	Private sewers
	P free detergents
Catch crops	Catch crops
Energy forestry	Energy forestry
Grassland	Grassland
Creation of wetlands	Creation of wetlands
Changed spreading time of manure	
	Buffer strips

Source: Gren et al. (2008)

Table A2: OLS estimates of coefficients in the quadratic cost functions (n=31), and calibration parameters of the nutrient cost functions

Region	*Nitrogen abatement, a^{ig} (t values and adj. R^2)*	*Phosphorus abatement; b^{ig} (t values and adj. R^2)*	*N and P abatement; c^{ig} (t values and adj. R^2)*	*Calibration parameters of c^{ig}*
Denmark;				
Kattegat	14.2 (t=38.9, adj. R^2=0.97)	4971 (t=125, adj. R^2=0.99)	109.2 (t=17.4, adj. R^2=0.91)	1.544
Danish straits	4.71 (t=30.2, adj. R^2=0.96)	2766 (t=51.7, adj. R^2=0.98)	2.54 (t=5.06, adj. R^2=0.44)	1.249
Finland;				
Bothnian Bay	8.79 (t=95.3 adj. R^2=0.99)	4347 (t=124, adj. R^2=0.99)	24.5 (t=10.5, adj. R^2=0.78)	1.350
Bothnian Sea	8.21 (t=66.5, adj. R^2=0.99)	2290 (t=150, adj. R^2=0.99)	35.7 (t=8.27, adj. R^2=0.69)	-1.558
Gulf of Finland	7.78 (t=81.4, adj. R^2=0.99)	2993 (t=80.1, adj. R^2=0.99)	7.16 (t=5.81, adj. R^2=0.51)	1.204
Germany;				
Danish Straits	8.0 (t=27.3, adj. R^2=0.96)	61982 (t=20.0, adj. R^2=0.93)	590.8 (t=41.0, adj. R^2=0.98)	1.165
Baltic Proper	8.04 (t=28.6, adj. R^2=0.96)	65525 (t=21.3, adj. R^2=0.94)	623.3 (t=55.2, adj. R^2=0.98)	1.182
Poland;				
Vistula	0.54 (t=28.6, adj. R^2=0.96)	255.3 (t=136, adj. R^2=0.99)	0.007 (t=13.2, adj. R^2=0.85)	1.317
Oder	0.99 (t=26.4, adj. R^2=0.96)	419.5 (t=133, adj. R^2=0.99)	0.04 (t=13.8, adj. R^2=0.86)	1.331
Polish coast	4.75 (t=20.8, adj. R^2=0.99)	1483 (t=105, adj. R^2=0.99)	2.78 (t=21.6, adj. R^2=0.94)	1.229
Sweden;				
Bothnian Bay	64.9 (t=29.8, adj. R^2=0.96)	10426 (t=29.1, adj. R^2=0.97)	680.9 (t=9.00, adj. R^2=0.72)	2.085
Bothnian Sea	25 (t=34.9, adj. R^2=0.98)	2468 (t=27.3, adj. R^2=0.96)	16.3 (t=9.24, adj. R^2=0.74)	5.777
Baltic Proper	6.49 (t=85.7, adj. R^2=0.99)	3230 (t=40.4, adj. R^2=0.98)	7.94 (t=35.4, adj. R^2=0.98)	1.058
Danish Straits	6.38 (t=24.9, adj. R^2=0.74)	13118 (t=110, adj. R^2=0.99)	558.7 (t=33.4, adj. R^2=0.97)	1.268
Kattegat	2.95 (t=111, adj. R^2=0.95)	6712 (t=45.3, adj. R^2=0.98)	32.3 (t=28.0, adj. R^2=0.96)	1.112
Estonia;				
Baltic Proper	18.8 (t=21.8, adj. R^2=0.93)	20727 (t=91.5, adj. R^2=0.99)	11.5 (t=19.9, adj. R^2=0.93)	359
Gulf of Riga	10.0 (t=22.6, adj. R^2=0.94)	9432 (t=18.3, adj. R^2=0.99)	6.21 (t=23.7, adj. R^2=0.95)	67
Gulf of Finland	1.33 (t=40.0, adj. R^2=0.98)	2160 (t=68.4, adj. R^2=0.99)	2.23 (t=44.7, adj. R^2=0.98)	1.261
Latvia:				
Baltic Proper	22.3 (t=27.9, adj. R^2=0.96)	5522 (t=95.8, adj. R^2=0.99)	230.9 (t=46.2, adj. R^2=0.98)	1.028
Gulf of Riga	4.93 (t=36.6, adj. R^2=0.97)	1635 (t=67.5, adj. R^2=0.99)	2.52 (t=25.0, adj. R^2=0.95)	1.190
Lithuania	39.6 (t=7.38, adj. R^2=0.63)	1268 (t=23.8, adj. R^2=0.94)	0.70 (t=43.8, adj. R^2=0.98)	27
Russia;				
Kaliningrad	43.6 (t=7.15, adj. R^2=0.62)	5846 (t=63.9, adj. R^2=0.99)	163.6 (t=11.3, adj. R^2=0.80)	1.313
S:t Petersburg	4.68 (t=15.8, adj. R^2=0.89)	733.5 (t=47.0, adj. R^2=0.98)	0.19 (t=27.1, adj. R^2=0.96)	3.214

Table A3: Area of land use, nitrogen and phosphorus loads from different drainage basins of the Baltic Sea

Region	*Area[1] thous. Km²*	*Arable land[1] thous. km²*	*Nitrogen[2] load, kton N*	*Phosphorus[2] load, kton*
Denmark:				
Kattegat	9.60	8.03	15	0.36
The Sound	16.16	12.93	30	0.69
Finland:				
Bothnian Bay	134.3	9.08	20	0.52
Bothnian Sea	46.66	5.37	16	0.42
Gulf of Finland	52.56	3.57	18	0.67
Germany:				
The Sound	9.77	7.26	21	0.21
Baltic Proper	11.95	8.49	22	0.20
Poland:				
Vistula	192.90	124.10	199	11.4
Oder	117.59	75.51	111	7.0
Polish coast	25.58	15.38	27	1.9
Sweden:				
Bothnian Bay	128.86	1.55	3	0.11
Bothnian Sea	180.19	5.67	10	0.43
Baltic Proper	92.38	17.34	25	0.56
The Sound	2.90	2.47	10	0.13
Kattegat	71.65	10.60	27	0.34
Estonia:				
Baltic Proper	6.07	2.15	5	0.11
Gulf of Riga	11.34	4.69	9	0.26
Gulf of Finland	65.49	32.84	44	1.20
Latvia:				
Baltic Proper	96.69	10.00	10	0.50
Gulf of Riga	122.45	62.25	36	2.41
Lithuania	96.69	59.01	93	2.90
Russia:				
Kaliningrad	20.00	15.08	16	0.83
S:t Petersburg	310.10	49.04	75	3.50

1) Shou et al. (2006) table A3.5. page 63. 2) Updated from Gren et al. (2008) tables B1 and B2.

Table A4: Helcom BSAP suggested basin reduction targets, in %

	P	N
Baltic Proper	66	29
Gulf of Finland	29	5
Gulf of Riga	34	
Danish Straits		32
Kattegat		31

Source: Helcom (2007) page 2

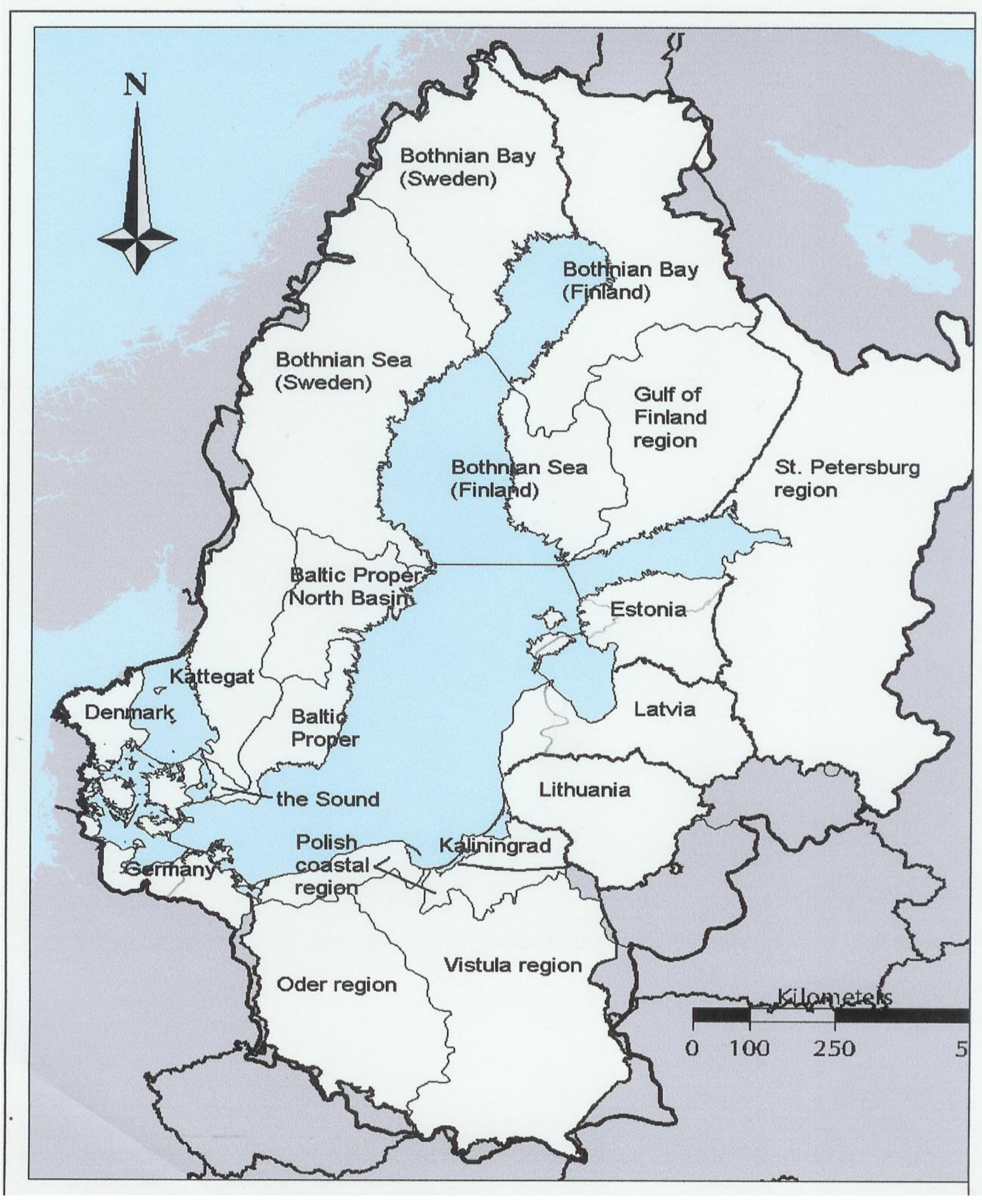

Figure A1: Drainage basins of the Baltic Sea (originally from Elofsson, 2003). (Drainage basins in Denmark (2), Germany (2), Latvia (2), and Estonia (3) are not provided with names, but are delineated only by fine lines)

Pris: 100:- (exkl moms)

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