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Data sheet

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Abstract: This report documents the revised cost-minimisation model for reducing nutrients loads to the Baltic Sea. The work is part of the MARE-project, and the key issue for the development of the cost-effectiveness model is to enable consistent modelling of the costs and the effects on nutrient loads of various measures implemented in the regions surrounding the Baltic Sea.

Keywords: MARE, Baltic Sea, Cost Minimisation Model, Cost Functions

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Preface

This report and the model development described herein are parts of the research programme MARE (MArine Research of Eutrophication) funded by the Swedish Foundation for Strategic Environmental Research, MISTRA. The objective is to develop a user-friendly decision support system termed NEST, enabling users to evaluate possible measures for reductions in nitrate and phosphate loads with respect to costs, and effects on the Baltic Sea (see: www.mare.su.se).

The main users of NEST are expected to be decision-makers within the Helsinki Commission (HELCOM), as well as organisations, authorities and researchers in the Baltic Sea States. Within the NEST model, the user can assess the most cost-effective measures to achieve targets for improved quality for separate sea basins within the Baltic Sea. The user can do this by changing various parameters in the system and creating different scenarios.

The first phase of the MARE project began with the development of NEST 1 in 1999, and the next phase of development of NEST 2 is taking place from 2002 to 2006. In this second phase the Department of Policy Analysis at the National Environmental Research Institute in Denmark (NERI) has become a partner in the MARE-project, being responsible for the further development of the cost-minimisation model in the NEST 2-model. The work carried out in this second phase has partly consisted of reformulation of the model to make it more transparent and enable interactive changes of the cost- and load modelling. Moreover, the data are updated, and the cost-functions are re-estimated. The data are based on official statistics from FAO and EUROSTAT, and further reviews of national data and studies have been included in formulating the cost-functions as far as they could provide reasonably general and well documented information.

In this report the model and the actual GAMS programme are documented. The development of the load-functions has been done in collaboration with the drainage basin modelling group. The primary readers of this report are expected to be scientists, expert users and others with specific interest in the development and assumptions behind the cost-minimisation model in the NEST system. Further development of the model should therefore include interactive use of the model with national experts, testing effects on cost-efficient solutions of changing both the cost- and load-functions.

We would like to express our gratitude to the Swedish research programme MISTRA and the steering board of MARE for financing this work, and for giving NERI the opportunity to contribute to this part of the project.

Definition of central terms

Due to the multidisciplinary approach in the project a number of central terms used are explained in the box below in order to facilitate the reading of the report.

Term	Explanation
Total abatement costs (TAC)	Total costs of abating emissions (and/or loads) by a certain amount
Marginal abatement costs (MAC)	The cost of the last abated unit of emissions (and/or loads)
Unit costs	The average costs per unit (ha, etc.)
Opportunity costs	Costs resulting form loosing the opportunity to utilise a resource optimally, e.g. as a consequence of restrictions on agricultural land use
Economic rent	The residual when all costs except remuneration of *land* are subtracted from the production value
Secondary benefits	Benefits arising for other environmental goods than that subject to the policy
Emission	Outlets of a pollutant, e.g. nitrate, from the source
Load	The amount of a pollutant, e.g. nitrate, reaching the recipient
Impact coefficient	Coefficient describing the relationship between emissions and loads
Effects (environmental)	Changes in the state of a recipient resulting from changes in loads
Measure	Term describing an initiative aimed at reducing (environmental) effects
Function	Functional form describing the quantitative relationship between e.g. the scale of implementation of a measure and total abatement costs (*in casu* TAC-function)
Cost efficient	The policy or mix of measures that fulfil a pre-specified target at the least costs

Summary

This report documents the revised cost-minimisation model for reducing nutrients loads to the Baltic Sea. The work is part of the MARE-project (see www.mare.su.se), and the key issue for the development of the cost-effectiveness model is to enable consistent modelling of the costs and the effects on nutrient loads of various measures implemented in the regions contributing to nutrient loads.

The purpose of the cost-minimisation model is to establish a framework for prescribing cost-efficient scenarios of reduced nutrient loads to the Baltic Sea. Nutrient loads derive from emissions both airborne and waterborne but in this work we focus on emissions only from countries with coastlines adjacent to the Baltic Sea, or with water drainage and transport to this sea region. Further, it demonstrated how secondary environmental effects, e.g. climate gas emissions, can be included in the welfare economic abatement costs.

The geographical boundaries of the model consist of the 9 countries surrounding the Baltic Sea divided into 24 drainage basins. The Baltic Sea is divided into 8 sea regions enabling setting up regional environmental standards for each sea regions. The division into drainage basins and sea regions reflects that emissions from each country contribute differently to the loads of the different sea regions because of retention, i.e. dilution and de-nitrification during transport to the sea regions, but also between the water bodies in the sea regions. Thus, the model solutions reflect regional differences in costs and loads. Cost functions and load reduction functions are developed for six different measures (establishment of wetlands, reduced livestock production, catch crops, reduced nitrogen fertilisation, sewage treatment, and NOx abatement), and for each measure a maximum level for implementation is defined at the national level.

An essential feature of the model is how the costs are modelled. For some of the measures with limited impact implemented at small scale it may be reasonable to assume marginal costs to be constant, i.e. the total costs are a linear function of the abatement level. However this model operates at national scales with measures which should result in environmentally significant changes in the nutrient loads to the Baltic Sea. Therefore, total costs are likely to be marginally increasing with the scale of implementation of each measure. This means that the extra cost of reducing loads by one extra unit are increasing, and non-linear costs functions are therefore implemented for those measures where the necessary data are available. This approach is in line with economic theory and general recommendations.

The result is presented as an aggregate cost estimate for the countries in the Baltic Sea region. However, this should not be interpreted as an indication of which countries should eventually bear the costs. This is important to note when passing the results to policy makers, as the model prescribe how the effort should be mixed in order to reach the least cost solution but not how this solution is reached in a political economic context.

Sammenfatning

Denne rapport præsenterer en revideret model til beregning af omkostningsminimerende strategier til reduktion af næringsstofbelastninger i Østersøen. Arbejdet er udført som en del af MARE-projektet (se www.mare.su.se) og det centrale i udviklingen af omkostningsminimeringsmodellen er, at gøre det muligt at generere konsistente omkostninger og effekter af næringsstofbelastningen som følge af reduktionstiltag implementeret i de lande, der grænser op til Østersøen.

Formålet med omkostningsminimeringsmodellen er, at give mulighed for at analysere omkostningseffektive scenarier for reduktion af næringsstofbelastningen i Østersøen. Næringsstofbelastninger skyldes luft- såvel som vandbårne emissioner, men denne rapport fokuserer kun på emissioner fra lande med kystlinier tilgrænsende Østersøen eller med eller transport af spildevand til disse kystområder. Endvidere demonstreres det, hvorledes sekundære miljøeffekter, fx klimagasemissioner, kan inddrages i de velfærdsøkonomiske reduktionsomkostninger.

Modellen omfatter alle de ni lande der omgiver Østersøen, med en yderligere opdeling af disse til 24 oplande. Østersøen er opdelt i otte havregioner, hvilket muliggør formulering af regionale miljøstandarder for hver enkelt havregion. Opdelingen i de 24 oplande og de otte havregioner afspejler det faktum at emissionerne fra de enkelte lande bidrager forskelligt til belastningerne af de enkelte havregioner, bl.a. på grund af retention af næringsstoffer under transport, men også på grund af transport af næringsstoffer mellem havregionerne. Modelløsningerne afspejler således regionale forskelle i omkostninger og belastninger. Omkostningsfunktioner og belastningsreduktionsfunktioner er opstillet for seks forskellige tiltag (etablering af vådområder, reduceret husdyrproduktion, efterafgrøder, reduceret kvælstofgødning, spildevandsrensning, og reduktion af NO_X-emissioner), og for hvert tiltag defineres en et maksimalt omfang for deres implementering.

Et vigtigt element i modellen er, hvordan omkostningerne modelleres. For tiltag implementeret i lille skala kan det være rimeligt at antage, at de marginale omkostninger er konstante, dvs. at de totale omkostninger er en lineær funktion af implementeringsniveauet. Men denne model er baseret på nationalt niveau med tiltag, der kan resultere i markante miljømæssige effekter i Østersøen. De samlede omkostninger vil derfor sandsynligvis være marginalt stigende. Dette betyder, at de ekstra omkostninger, der er forbundet ved reduktion af belastningen med én ekstra enhed, er stigende, og ikke-lineære omkostningsfunktioner implementeres derfor for de tiltag, hvor den nødvendige information er tilgængelig. Denne fremgangsmåde er i overensstemmelse med økonomisk teori og generelle anbefalinger.

Resultatet af en modelkørsel præsenteres som et aggregeret omkostningsestimat for landene i Østersøregionen. Dette skal imidlertid ikke tolkes som en indikation af, hvilke lande der i sidste ende skal afholde omkostningerne. Dette er vigtigt at holde sig for øje, når resultaterne overgives til administratorer og politikerne, da modellen belyser hvordan indsatsen skal sammensættes for at opnå den mest omkostningseffektive løsning, men ikke hvordan denne løsning opnås i en politisk sammenhæng, herunder hvordan den finansieres.

1 Introduction

1.1 Background

Inputs of nutrients, such as nitrogen and phosphorus, to the sea are natural prerequisites for life, and not environmental problems per see. Nutrient emissions only become problematic when the inputs increase to such extents that the original properties or functions of the ecosystem are affected, and the sea becomes too eutrophic (see www.mare.su.se). Intense algae blooms, turbid water with reduced transparency, oxygen deficiency and reductions in the amount and presence of sediment-living animals may be the severe results of eutrophication. The economic values of eutrophication of coastal ecosystems, and reductions in the eutrophication level, are not investigated very intensively, but studies performed in the Baltic region indicate that the values can be significant (Elofsson, 2003; Söderquist, 1996; Markovska & Zylicz, 1999). The enclosed brackish-water Baltic Sea, with its slow water exchange and natural barriers, is particularly sensitive to eutrophication. However, within this sea region different effects will dominate in separate parts of the sea basins due to large differences in natural conditions between the basins, as well as different loads of nutrients to the basins.

Much is already known about the effects of eutrophication in the Baltic Sea ecosystems and about the costs of measures to reduce the nutrient loads. The background for this project, and for the development of a cost-effectiveness assessment tool, is that there is not one single efficient measure and solution valid for the entire Baltic Sea. The reason is that it is not likely that the same measures for reduction of eutrophication can be applied cost-effectively both in a local coastal area and in the open sea perspective - specific conditions in different parts of the sea with respect to both production and natural load conditions must also be considered. For example, the same set of measures cannot automatically be implemented in the Gulf of Bothnia and in the Southern Baltic, since these basins have different natural properties. Furthermore, the measures are connected with different costs in different areas.

In all there are nine countries surrounding the Baltic Sea, and in the cost-minimisation model 24 drainage basins within these countries form basic units for the estimation of costs and loads for the different policy measures relevant for reducing nutrient loads to the Baltic Sea.[1] The drainage basins can be seen from the map (Figure 1)

[1] The former version of the model consisted of 82 *drainage basins* in 9 *countries.* The drainage basins were formerly termed *emitting regions.*

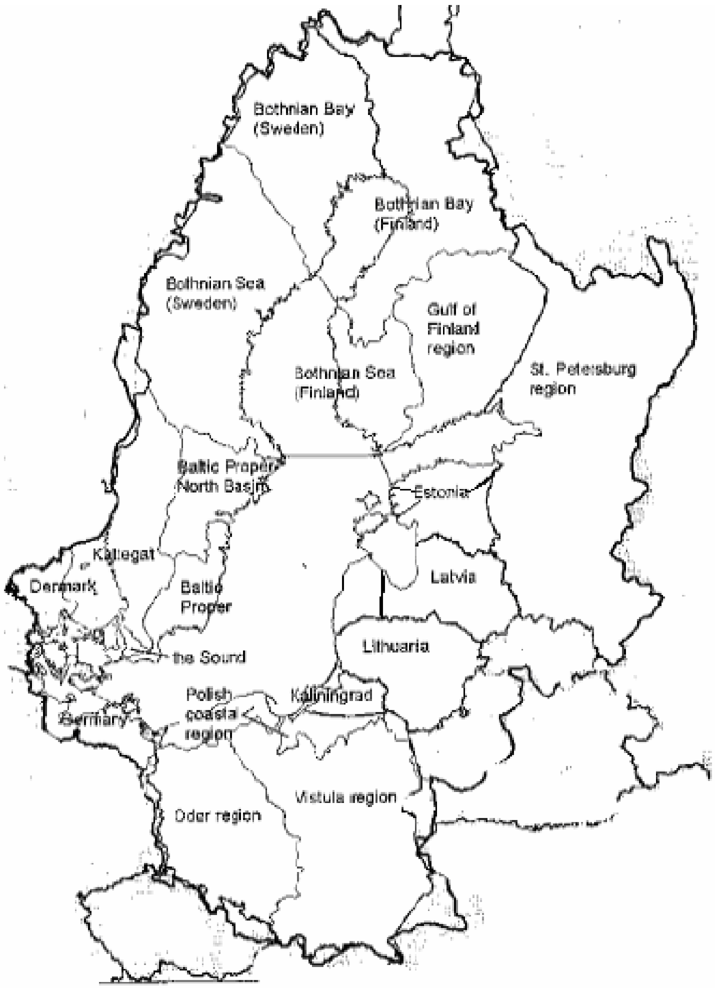

Figure 1 The Baltic Sea, sea regions and drainage basins.

Because of the large scale of the Baltic Sea, MARE divides the entire Baltic into *seven sea regions (basins)* enabling to set up regional environmental standards for each of these sea regions. The division into sea regions reflects the fact that the emissions from each country contribute differently to the loads of the different sea regions because of retention, dilution and de-nitrification during transport to the sea regions, but also between the water bodies in the sea regions. For example, even though the unit costs of measures reducing nitrogen emissions may be minimised when implemented in the Polish drainage basins this is not per see the most cost effective solution for reducing loads to the Danish straits. This is because the cost-effectiveness of the measure should be related to the loads and not to the emissions.

1.2 Purpose of the work

The purpose of the cost minimisation model is to establish a framework for prescribing cost efficient scenarios of reduced nutrient loads to the Baltic Sea. Nutrient loads derive from both airborne and waterborne emissions but in this report we only focus on emissions from countries with coastlines adjacent to the Baltic Sea, or with water drainage and transport to this sea region.

The purpose of this report is to describe the principles of the revised cost minimisation model, the cost functions, and the data available and data used in the cost-minimisation model. The cost minimisation model is basically built on similar principles as the past version developed in the first phase of MARE (cf. Elofsson, 2003), as both model versions are set to minimise the costs in the Baltic Sea area to fulfil different environmental targets. But the model itself has changed significantly. All data input of the model are updated, and the structure of the model has been changed in order to make the modelling more transparent and to provide facilities for easy updating. This also enables a transparent implementation of the estimated load functions based on the results from the drainage basin model. This specific task has been worked out in close collaboration with the natural scientists at the universities of Linköping and Stockholm who are responsible for the drainage basin model. However, it should be noted that because of delays in the development of the drainage basin model the documentation in this paper focuses alone on the economic content of the model and the environmental economic modelling principles. The load responses implemented at this time are based on educated guesses from on Danish and Swedish experiences and a literature review. The assumptions and parameters behind the current load response modelling are documented in Appendix 1. Once the final drainage basin specific load functions can be supplied the load response parameters can be replaced, which in turn may require repeated calibration of the model.

The paper initially describes the principles of the cost-minimisation model and the data needed for solving the cost minimisation problem. Hereafter the policy options or measures are described, and the abatement cost functions for each of the measures in the model are presented. These functions form the core elements of the model and different specifications of the abatement cost functions are demonstrated and discussed. The costs are estimated as welfare economic costs. The inclusions of secondary environmental effects (e.g. changes in climate gas emissions) in the cost estimates are demonstrated for two of the measures in the model. But due to scarcity of data the present model-version is set default to estimate the cost-minimisation without these effects.

2 The model

2.1 The cost-minimisation: objectives and limitations

The starting point of the cost minimisation problem is to decide on the goals for the reduction of loads in one or more sea regions. The result from the cost minimisation is presented as an aggregate cost estimate for all of the countries in the Baltic Sea region, as well as effects distributed between measures and countries. The model also provides effects in terms of reduced N and P emissions from sources and drainage basins, and reduced N and P loads to the sea regions.

However, this should not be interpreted as an indication of which countries should eventually bear the costs. This is important to note when passing the results to policy makers, as the model prescribes how the effort should be mixed in order to reach the least cost solution but not how this solution is reached in a political economic context. As an example a scenario may indicate that a huge effort should be put in Poland, but how this effort is financed, i.e. which countries actually bears the costs, is a political economic question, which cannot be addressed by a model.

The present version of the cost minimisation model is static comparative presenting the net effects of changing from one environmental load to another. Thus, the model does not include time aspects and this is important when interpreting the results. This is because some measures may result in a more or less instant reduction in loads whereas others may have a significant time-lag between the time of implementation and the time of the resulting changes in loads. Therefore, information on the timing of the changes in loads needs to be supplied additionally if this parameter is of importance for the decision making. The influence of time-lag effects on the cost minimising abatement strategies will be analysed as part of the economic sub-project and the results are reported separately.

2.2 The principles of the cost-minimisation model

The cost minimisation problem is formulated as a choice of the cost minimising mix of policy measures within drainage basins. The cost minimising mix of measures consists of an optimal mix of measures and an optimal localisation of the measures, so that a specified goal for reductions of loads to one or more sea regions is obtained to the least total abatement cost. The minimisation problem is constrained by the exogeniously set potential of each policy measure. These limits on the potential emission reductions of the measures can be explained by the fact that each measure, e.g. wetlands, has a limited feasibility range within each drainage basin. Secondly, the implementation range should reflect that the estimates of costs and emission reductions shall remain coherent with the assumptions of prices, technology, etc.

The cost minimization problem is a non-dynamic (static) problem described by:

$$\min_{x_{i,å}} \sum_{i=1}^{n} \sum_{k=1}^{m} TAC_i^k \left(x_{i,p} \right)$$

$$st. \qquad \text{a)}$$

$$\sum_{k=1}^{m} g_j^k \left(x_{i,p} \right) = T_j$$

$$h_i^k \left(x_{i,p} \right) \leq h_i^k \max$$

Where

$x_{i,\,p}$	is the reduction of nutrient load,
p	is (nitrogen, phosphorus) in each drainage basin,
g_j	is a function describing the share of reduced emissions emitted from drainage basin i reaching sea region j, and
h_i	is a function describing the extend of policy measure k implemented when reducing the emissions in drainage basin i.
T_j	is the target load reduction for sea region j, and
h_i^k max	is the maximum emissions reduction of policy measure k implemented in drainage basin i.

In the model the functions g and h are linear in *x*, saying that there is a simple linear relationship between emissions from a drainage basin and the nutrient loads reaching a sea region (function g). Similarly there is a positive linear effect on nutrient emissions in a drainage basin of implementing a specific policy measure (function h). Therefore the first derivative for g and h exists and are continuous (g and h are C^1 functions).

By assumption the gradient for the function h for the drainage basins that constitutes the active constraints and the gradient for the function g in all sea regions are linearly independent.

For a solution to the problem, x_p* - a specific reduction of nutrient emissions - the necessary conditions for optimality are the Kuhn-Tucker conditions. However, the Kuhn-Tucker conditions only provide possible solutions that are potentially optimal, which means that they are not sufficient for a globally optimal solution. To obtain optimal solutions the general microeconomic theory on cost functions are applied. Hence total abatement costs are quasi-convex and all separate cost functions for policy measures are convex. Because g and h are linear functions and the objective function is quasi-convex, the Hessian to the Lagrangian is positive semi-definite and in conclusion the solution is optimal and unique. The necessary Kuhn-Tucker conditions are given in b) – f).

$$\sum_{i=1}^{n}\sum_{k=1}^{m}\frac{\partial TAC_i^k\left(x_{i,p}*\right)}{\partial x_{i,p}}-\sum_{j=1}^{o}\sum_{k=1}^{m}\mu_j\frac{\partial g_j^k\left(x_{i,p}*\right)}{\partial x_{i,p}}-\sum_{i=1}^{n}\sum_{k=1}^{m}\lambda_i\frac{\partial h_i^k\left(x_{i,p}*\right)}{\partial x_{i,p}}=0 \qquad \text{b)}$$

$$\sum_{k=1}^{m} g_j^k\left(x_{i,p}*\right)=T_j \qquad \text{c)}$$

$$h_i^k\left(x_{i,p}*\right)\leq h_i^k \max \qquad \text{d)}$$

$$\lambda_i h_i^k\left(x_{i,p}*\right)=0 \qquad \text{e)}$$

$$\lambda_i\geq 0\left(=0\ if\ h_i^k\left(x_{i,p}*\right)< h_i^k \max\right) \qquad \text{f)}$$

The condition in b) ensures optimality, c) and d) are feasibility conditions, e) is the complementary slackness condition and f) is a non-negativity condition. The vectors λ_i and μ_j are the Lagrangian multipliers; in economic theory often referred to as shadow prices.

2.3 Modelling the cost functions

An essential feature of the model is how the separate costs of each of the measures are modelled. The cost-function for a given measure can be written as C(w,y), where w is the factor prices, and y is the given level of the measure. In the welfare economic assessment the costs constitute the alternative costs following the resource use from implementing the measure, the direct costs of implementing the measure and the indirect costs from secondary environmental effects. We will return to the measurement of these three parts constituting the abatement costs after a short discussion of the properties of the cost-function.

The general properties of the cost-functions for each of the measures are assumed to be:

- Non-negativity C(w,y) > 0 for w > 0 and y > 0. Meaning that costs always will be incurred when implementing a measure.

- No fixed costs C(w,0) = 0. Meaning that the costs associated with a measure are assumed to vary.

- Monotonicity in y: if y' ≥ y, then C(w,y') ≥ C (w,y). This means that if the required level of measures increases in order to fulfil a more restrictive target, then the total costs at the cost-minimising point will be higher, with everything else assumed constant.

- Monotonicity in w: if $w' \geq w$, then $C(w',y) \geq C(w,y)$. This means that at factor price w, x is the cost minimising activity level, while at price w′, x′ is the cost-minimising activity level.

- Homogeneity of degree one in prices: $C(tw,y) = tC(w,y)$. This means that when factor prices are changed, the costs changes with the same order.

- Concavity: C(w,y) is concave in w. This can be explained rather intuitively: If a price of a factor rises, costs will never decrease. If the cost function is linear the costs will increase at the same rate as the price increases. If the cost function is not linear, the costs will go up at a decreasing rate. The reason being that when one factor-price increases and the prices of other factors stay the same, cost-minimisation implies that this factor will be substituted by other inputs or activities.

- Continuity: C(w,y) is continuous in w. Continuity means that C(w,y) changes continuously when w changes; continuity is necessary for the existence of an optimum.

- If C(w,y) is differentiable, then there is a unique vector x, so that $\partial C(w, y)/\partial w_i = x_i$. Uniqueness of optimum exists if the function is strictly concave and the constraint is convex.

For more details about the theoretical aspects see Varian (1992).

For some of the measures with limited impact implemented at small scale it may be reasonable to assume marginal costs to be constant, i.e. that total abatement costs are a linear function of the abatement level. However, this model operates at national scales with measures, which should result in environmentally significant changes in the nutrient loads to the Baltic Sea. Therefore, total costs are likely to be marginally increasing with the scale of implementation of each measure. This means that the extra costs of reducing loads by one extra unit are expected to increase. One example is construction and restoration of wetlands where farmers should be expected to choose the areas yielding the least economic rent first when converting agricultural land into wetlands. But as more land is converted into wetlands higher yielding areas need to be chosen, thus leading to cost increase at the margin.

Non-constant costs can be represented in two ways: constantly increasing (linear form) or increasing at the margin (quadratic or polynomial form); the former is the usual assumption in theoretical environmental economic analysis. Using a polynomial form for the total abatement costs (TAC) curve, the relationship between TAC and marginal abatement costs (MAC) can be described as shown below.

$$TAC = k + ax + bx^2 + cx^3 \quad \Leftrightarrow \quad TAC' = MAC = a + 2bx + 3cx^2$$

In Figure 2 the three types of marginal cost curves are illustrated.

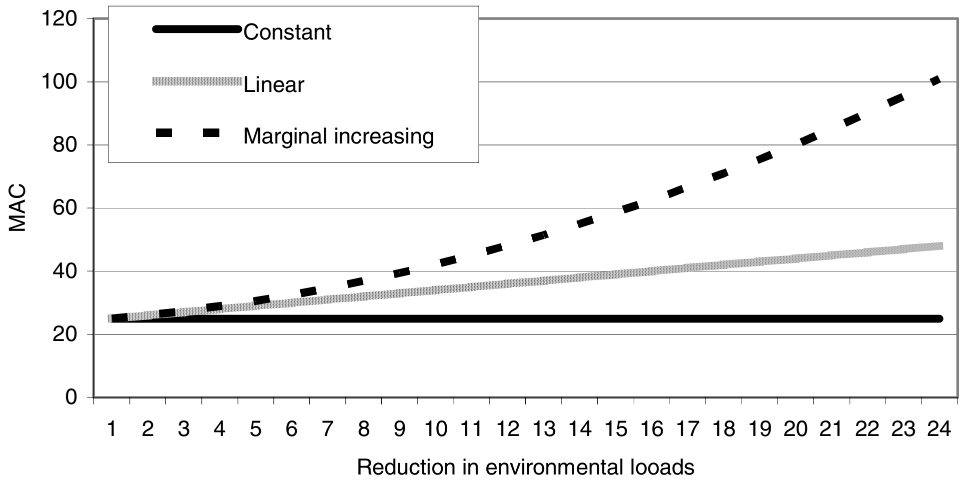

Figure 2 Examples of marginal abatement cost curves (arbitrary units).

Choice of coefficients can lead to all three types of marginal cost curves. For the first curve a=25 and the coefficients b and c are zero. For the second curve a=25, b=0,5 and c=0. For the third curve a=25, b=1 and c=1/30. Thus, using the non-linear form in the initial modelling gives flexibility to adjust the cost functions depending on the data available.

The various cost functions hold different properties with respect to solving the cost minimisation problem stated in Section 2.2. Using the linear cost function leads to a non-continuous abatement cost curve as the cost efficient combination measures with respect to reducing loads to sea region j will be stepwise (MAC are constant). For the non-linear cost functions (resulting in linear or non-linear MAC) the abatement cost curves may be continuous, however, this will depend on the parameters. Using cost functions resulting in non-constant MAC has the theoretically appealing feature that "corner" solutions are less frequent. This has the implication that the model result is less dependent of the model restrictions that define the potential emission reductions of each measure.

However, estimating non-linear cost functions demands extensive economic data of the distribution of the economic rent from e.g. agricultural holdings in each country represented in the model. For some measures such cost curves can be estimated roughly based on either assumptions of a step-wise implementation of cleaning technologies or by generalising national estimates of relationships between input and production. It has for example been possible to find national specific yield-functions for nitrogen application. In other words, for some measures no national specific data are available. Based on these considerations the cost-minimisation model is initially formulated as a combination of a linear and non-linear programming model. For those measures where data are available for estimation of non-linear cost functions or for which point estimates of costs can be used, those cost functions are build into the model. For measures where establishment of non-linear TAC functions is not feasible linear TAC functions are applied. However, in the programming of the model implementation of non-linear cost functions are made possible if relevant cost data should be accessible during future revisions.

Choosing this approach to specify the model leads to the following minimum data requirements for the cost-minimisation model:

- costs functions for each measure in each country (data specifying the TAC-functions)
- potential emission reductions of policy measures in each country (data specifying the load function)
- impact coefficients expressing the share of one unit emission in country *i* ending up as load in sea region *j*

In the current model version the cost-functions and load functions are specified at drainage basin level. This enables analyses of cost minimising fulfilment of separate targets for each of the sea-basins receiving loads of nutrients from a set of drainage basins.

3 Cost-functions for each measure

In this chapter the assessment of the total abatement costs of each of the measures, which represent the policy options in the model, are described. The analysis involves a definition of each measure including its maximum level of implementation, and description of the cost functions and the data and estimations leading to the parameters. The model data, e.g. land use, are presented in Appendix 3. In the current prototype model version the maximum level of implementation of each measure are set arbitrarily. However, as for the parameters in the cost and load functions it is possible to change these default values based on local expert knowledge.

The definitions of the measures are co-ordinated with the work package developing the BASIN-SIM model securing consistency between the economic and environmental estimates, which represent the databases of the cost-minimisation model. Whenever possible existing cost-functions for each country are used, and for other measures published cost data are used to derive cost functions that quantify the costs of implementing the measures at different scales in each country.

3.1 Description of measures

The measures incorporated in the revised model comprise:

1. Wetland restoration
2. Reduced fertiliser use
3. Introduction of catch crops in agriculture
4. Livestock reduction in agriculture
5. Improved treatment of sewage
6. NO_X reduction
7. "Blank" measures

3.1.1 Wetlands

This involves conversion of agricultural land into wetlands. The anticipated maximum feasible number of hectares in each drainage basin (and country) is estimated based on historical land use statistics for each country. The maximum area converted into wetlands is assumed to correspond to 5 % of the agricultural area.

3.1.2 Reduced nitrogen fertiliser use

This measure applies to agricultural production and is implemented by a uniform reduction in fertiliser use on the agricultural area in rotation. The present average fertiliser application per hectare is estimated by FAO statistics. The maximum reduction per hectare in each country is set at 25 % of the initial fertiliser use.

3.1.3 Catch crops on agricultural land

This measure also applies to agriculture and involves under sowing of catch crops (e.g. grass) when sowing the ordinary crop (e.g. barley or wheat). The present acreage with catch crops is estimated by FAO statistics of agricultural land-use, and the feasible acreage with catch crops and the percentage reduction in N-leaching in each country are set at 1/3 of the agricultural area in rotation.

3.1.4 Reduced livestock density

This measure applies to agricultural livestock production. The present livestock production is estimated based on FAO statistics, and the maximum reduction in livestock production in each country is set at 80 %, estimated based on FAO-statistics of agricultural livestock.

3.1.5 Improved sewage treatment

The baseline for this measure is the number of person equivalents (PE) not connected to municipal waste water plants. The costs and load effects are estimated based on assumptions of various improvements in terms of investments in sewage treatment technology. The maximum constraints of the measure are set as 20% of the number of PE not connected to municipal waste water plants.

3.1.6 Reduced NO_X emissions from fossil fuels

This measure involves installation of de-NO_X units at large power plants. The maximum scale of the measure is set at a reduction of 1,000 tonnes for each country.

3.1.7 Blank measures

A number of "blank" measures are implemented in the model in order to enable developments and updates of the model by inclusion of more measures. Thus, if desired and if the necessary data is available it will be simple to add new measures to the model at a country level basis.

3.2 Components of the cost measurements

Costs of each of the policy measures are represented by the change in welfare economic rent to society caused by implementing the measure. The reduction in economic rent includes three components. The first is the opportunity costs, which represent the loss in economic rent from changing the initial resource use, e.g. the change of land use from agriculture to wetlands.[2] The second component is the costs of establishing the new activity, e.g. the construction costs of wetlands comprising investment costs, operational costs and maintenance costs. This also includes any income from the new activity. The third is the possible secondary benefits. An example is the reduction in climate gas emissions resulting from establishment of wetlands.

[2] A description of the agricultural economic reference which are the basis of modelling opportunity costs of crop and livestock production are given in appendix 2.

Calculation of the total abatement costs (loss of *economic rent*) of implementing measure k in drainage basin i (TAC_i^k) is described in the formula below.

$$TAC_i^k = OC_i^k + IC_i^k + MC_i^k + OPC_i^k - (SEB_i^k + I_i^k)$$

Where

OC_i^k	is the opportunity costs
IC_i^k	is the investment costs
MC_i^k	is the maintenance costs
OPC_i^k	is the operational costs
SEB_i^k	is the value of the secondary environmental effects
I_i^k	is the income or costs saved

All calculations are presented as annual values. Costs only occurring in one period, e.g. investment costs are transformed into average annual costs using the formula below.

$$\overline{IC_i^k} = TIC_i^k \frac{(1+r_i)^n r_i}{(1+r_i)^n - 1}$$

Where

$\overline{IC_i^k}$	is the average annual investment costs of implementing measure k in drainage basin i
TIC_i^k	is total investment costs in period 0 of implementing measure k in drainage basin i
r_i	is the discount rate in drainage basin (country) i
n	is the depreciation period of the investment

The discount rate is set as default at 3 % p.a. The depreciation period depend of the type of the investment, but is set as default at 20 years.

In the model drainage basins are the geographical unit for implementing the single measures. This is done to enable integration of the results from the drainage basin model into the cost minimisation model. In the parameterization of the model, data has not been available to reflect differences in cost functions between the drainage basins within one country.

3.3 Wetlands[3]

The costs of this measure occur when agricultural land is withdrawn from production and converted into wetlands. Therefore the first cost component is the agricultural opportunity costs expressed as the value of the yearly economic rent from the current land use. Additional to this, administrative costs may be expected as well as construction costs from use of machinery and labour if drainage needs to be dismantled or excavation is needed. Finally, establishment of wetlands will lead to secondary benefits in terms of reduced ammonia and climate gas emissions and benefits related to use and non-use values. These are addressed later in the report.

In order to reflect the non-constant MAC in the cost modelling empirical evidence for variations in economic rent among farmers is needed. Such information can either be obtained by farm scale modelling or estimated based on individual farm account statistics. An example of modelling of economic rents at farm level has been done in Schou (2003a) for a study area of 1,000 hectares located in Jutland, Denmark. These data have been used for estimations in order to indicate the functional form of the opportunity cost curve for converting agricultural land into wetlands.

In the following the estimation of functions for agricultural opportunity costs from wetland restoration is explained. The design and the OLS[4] estimates of the cost function are shown below.

$$\begin{aligned} Wetlands: \quad & TAC(x) = ax^2 + bx \\ & TAC(ha) = 0.06075 \cdot ha^2 + 214.781 \cdot ha, \\ & R^2 = 0.97 \end{aligned}$$

The estimation includes quadratics, which offers a simple way to capture the increasing marginal cost of restoring wetlands. The marginal cost or simply the slope of the graph (Figure 3) is given by the 1st derivative and the estimation results are shown in Figure 3 and Table 1.

$$\frac{dTAC^{wetlands}}{dha} = 2ax + b = 0.1215 * x + 214.781$$

[3] A description of the agricultural economic reference in each country can be found in appendix 2. The data are used for assessing the economic rent from crop production an livestock production and, thus, the opportunity costs of these activities.

[4] OLS is short for 'Ordinary Least Squares', which is the common procedure for estimation in linear models.

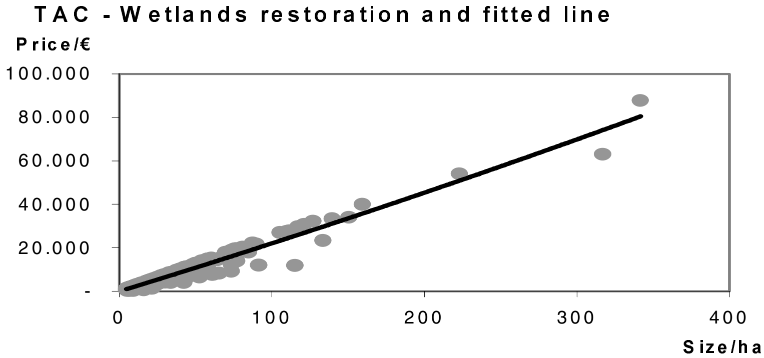

Figure 3 Estimated agricultural opportunity costs from wetland restoration for wetland restoration (OC).

Table 1 Agricultural opportunity costs, parameter estimates and test-statistics.

	Estimate	Standard error	t-value
A	214.781	4.523	47.490
B	0.061	0.022	2.770
N	250.000		
R^2	0.970		

A number of assumptions concerning: 1) linearity in parameters, 2) random sampling, 3) zero conditional mean, 4) no perfect collinearity, and 4) homoskedasticity, must be met in order to have BLUE-estimators. BLUE is an acronym for 'Best Linear Unbiased Estimator', which means that a linearly estimated parameter is unbiased and has the smallest variance among alternatives. The model of agricultural opportunity costs violates two of the assumptions. It regards the assumptions of no collinearity between the independent variables and that of homoskedasticity. The latter causes difficulties when testing the significance of the estimates. The first difficulty is due to the relative small estimate of 'a' compared to the relative large estimate of 'b' and it causes bias in the estimates. Although this implies violation of assumption 3) it can be discussed whether it is enough to discard the estimations of the parameters. The fit of the estimates are very high and the model is in accordance with the intuition. In addition it is worth noting that a linear relationship between income and property size imposes the same amount of bias in the estimator.

The assumption of homoskedasticity is tested with a Breusch-Pagan test. This test requires a second regression of the squared residuals obtained from the first regression, on the independent variables. The test-statistics are shown in Table 2.

Table 2 F-statistics.

	F-value	p-value
Agricultural opportunity costs	75.67	<0.0001

The heteroskedasticity test, based on the F-distribution, is shown in Table 3 and the p-value is computed using $F_{k,\ n-k-1}$. The F-value is as high as 75.67 and it is not possible to reject the hypothesis of heteroskedasticity.

In Table 3 the heteroskedasticity robust standard errors, the robust T-tests plus the robust LM-test of the wetlands model are shown. It is clear from the table that testing the model with heteroskedasticity robust standard errors make evidence against the model specification. The errors are now larger, which make the t-statistics smaller and the p-values larger.

Table 3 Heteroskedasticity robust standard errors in the wetlands restoration model.

	Robust standard errors	t- value	p-value
A	9.162611	23.44	<0.0001
B	0.073238	0.83	0.408
F-value	891.44		

The square term may be excluded or at least a parameter value in the critical area is observed 41 % of the time. By accepting H_0 or in other words by including the square term in the model, it is exposed to a type I error; that is rejecting the hypothesis even if it is true. Overall it is not possible to exclude both independent variables because of the F-value of multiple exclusion restrictions.

Obviously the estimation of wetlands restoration causes the problem of heteroskedasticity and furthermore robust testing shows that the square term might be excluded. It is important to keep in mind that only the testing of the estimators is affected; although it is not a sign of bias or inconsistency of the estimator, OLS is no longer BLUE. The quadratic model specification for wetlands is chosen anyway because of its high R^2-value, and because the quadratic form captures the intuition of increasing returns to scale. Furthermore, without having actually shown it the fact is that of various model specifications the quadratic form is the better in an attempt to satisfy the assumptions as good as possible. Another model design implies other and stronger assumption violations and among the tested models the quadratic function exhibits the best fit with the available data. In other words, the quadratic design is the best compromise among the alternatives.

The estimated functions are meant to fit each country. This is done by a simple calculus. The known country specific factors are: mean income on landed property for crop production and livestock, plus total size of cultivated land and total amount of livestock. Therefore the interval and the mean value of the country specific functions are known. The functions are then calculated by finding in the estimated model, e.g. 5 % of the total area and the corresponding percentage of income. It is easy then to impose the same percentage correspondences on the specific countries. The scaled functions are shown in Table 4.

A study by Söderqvist (2002) where costs of wetland reestablishment are estimated based on Swedish experiences report that total costs are distributed with 16 % on loss of agricultural production (approximated by the compensations paid), 24 % on administrative costs and 60 % on construction costs. Thus, the lump sum construction costs are estimated to approximately 92,500 SEK per hectare converted (1997-prices; own calculation based on Söderquist, *op cit.*). An important result from the Swedish study is the identification of a large administrative effort related to establishment of wetlands, which both relates to the need for spatial planning prior to the wetland establishment and the need for identifying landowners and negotiating compensations.

Table 4 The scaled OC functions for crop production.

	OC function parameters	
	A	B
Denmark	$1{,}595 \cdot 10^5$	442
Estonia	$1{,}388 \cdot 10^5$	128
Finland	$1{,}060 \cdot 10^5$	243
Germany	$0{,}253 \cdot 10^5$	446
Latvia	$2{,}863 \cdot 10^5$	73
Lithuania	$0{,}127 \cdot 10^5$	46
Poland	$0{,}088 \cdot 10^5$	167
Russian fed.	$0{,}007 \cdot 10^5$	167
Sweden	$2{,}848 \cdot 10^5$	926

A recent Danish analysis performed as preparation of the third Danish Water Action Plan (Jacobsen et al., 2004) estimates the lump sum costs to the administration of establishing wetlands to about 66 € per hectare (2001-prices). Construction lump sum costs are estimated on ad hoc basis to 250 – 625 € per hectare. Compared with the estimated compensations of 6 –7,500 € per hectare in Jacobsen et al., *op cit.* construction costs and administrative costs in the Danish study only account for approximately 6 and 1 % of the total costs, respectively.

These studies indicate that the construction costs of establishing wetlands may be highly site dependent. For example if wetlands can be constructed by simply abandoning drainage pipes or by stopping maintaining drainage canals construction costs may be low. Another issue leading to country variations in construction costs is differences in labour costs and the availability of machinery. It seems that the construction costs in relation to the wetlands in the Swedish study may be high compared with the possibilities in for example the large estuaries in Poland or reestablishment of wetlands in drained river valleys.

The estimates of administrative and construction costs are therefore done on a country basis so that the results of Söderqvist (*op cit.*) are used for Sweden and the results in Jacobsen (*op cit.*) are used for the other countries. In order to reflect differences in labour costs etc. administrative and construction costs are estimated as a fixed fraction of the loss in economic rent from agricultural production based on the percentage cost distributions stated above. The results are shown in Table 5.

Table 5 Estimated loss of economic rent from agriculture, construction costs and administration costs of establishing wetlands, annual values in 2001-prices.

Country	Construction costs €/ha	Administration costs €/ha
Denmark	28	5
Estonia	8	1
Finland	15	3
Germany	28	5
Latvia	5	1
Lithuania	3	0.5
Poland	11	2
Russian fed.	11	2
Sweden	583	233

Source: Own calculations.

Denoting the number of hectares agricultural land converted into wetlands as x the TAC function can be specified as:

$$TAC(x) = ax^2 + bx + cx + dx$$

$TAC(x)$ is the total costs of reducing the number of hectares with crop production in rotation with x hectares. a and b are constants specifying the opportunity costs of crop production, cf. Table 4, c and d are constants specifying the construction costs (c) and administrative costs (d) of establishing wetlands, cf. Table 5.

3.4 Reduced livestock production

This measure applies to agricultural livestock production. The method for calculating the opportunity costs is similar to that applied to wetlands, only economic rent from livestock production is used. The opportunity costs of reducing livestock production were also tested for being marginally increasing following the same arguments as those for establishment of wetlands. A TAC function has also been estimated for this measure using the Danish data set. The TAC function is estimated for both pigs and cattle, and in both cases the estimations resulted in a linear specification of the TAC function:

$$TAC(x) = ax$$

Where

$TAC(x)$ is the total costs of reducing the number of livestock with x heads

a is a constant specifying the opportunity costs of livestock production

The parameters of the country specific cost functions are shown in Table 6.

Table 6 Average unit costs from reduced livestock hold.

	Pigs €/head	Cattle €/head
Denmark	59	295
Estonia	68	162
Finland	59	282
Germany	53	196
Latvia	57	130
Lithuania	22	47
Poland	38	111
Russian fed.	38	111
Sweden	47	194

It should be noted that reductions in livestock production and, thus, the production of manure may result in derived reductions in the economic rent from crop production either because of reduced N input or extra costs to N fertiliser use. This is not reflected in the cost estimates for reduced livestock production. In order to reflect this, the cost estimates should be added a cost component reflecting that extra fertiliser application is needed in order to compensate the lost nutrient input from manure. However, this requires specific knowledge on the utilisation of nutrients in the manure applied in each country, and this is currently not available.

3.5 Reduced nitrogen fertiliser use

Because of the ongoing attention on nitrogen emissions from agriculture a large number of studies of the costs of reducing nitrogen application have been published. The economic modelling of nitrogen application is done using the principle in the formula below.

$$\pi(n) = p^{crop} y^{crop}(n) - p^n n - C$$

Where

$\pi(n)$	is the economic rent per hectare
n	is the nitrogen input per hectare
p^{crop} and p^n	is the price on crops (output) and nitrogen input
$y^{crop}(n)$	is the yield response function
C	is fixed costs per hectare

The maximum profit resulting from input of nitrogen can then be deducted by differentiating the equation with respect to *n*. Then the optimal nitrogen inputs are found where:

$$\frac{\partial \pi(n)}{\partial n} = 0 \Leftrightarrow p^{crop} \frac{\partial f^{crop}(n)}{\partial n} - p^n = 0$$

(Cf. e.g. Varian, 1992).

In the following the notions of n_1 and n_2 are used. n_1 and n_2 are different levels of applications of nitrogen fertilisers and in the current model-use n_1 will be identical with the nitrogen input in the reference scenario. $n_1 \geq n_2$. By subtracting the profit from the application of n_2 from the profit of the initial application of fertiliser the opportunity cost from lost yield is calculated. The costs of reducing nitrogen input is dependent on both the current level of economic rent from crop production (and thus, yields and factor productivity) and the relative modelled marginal costs. TAC per hectare is then:

$$\Delta\pi(n_1, n_2) = (p^{crop} y^{crop}(n_1) - p^n n_1) - (p^{crop} y^{crop}(n_2) - p^n n_2)$$

Where $\Delta\pi_i(n_1, n_2)$ is the loss in economic rent per hectare agricultural land in rotation in drainage basin i

Note first that all other costs than those to N fertilisers are assumed constant and, thus, are netted out in the equation. Note secondly that the first part of the equation (π_1) is a constant depending of the initial nitrogen fertiliser application. This relationship is illustrated in Figure 4.

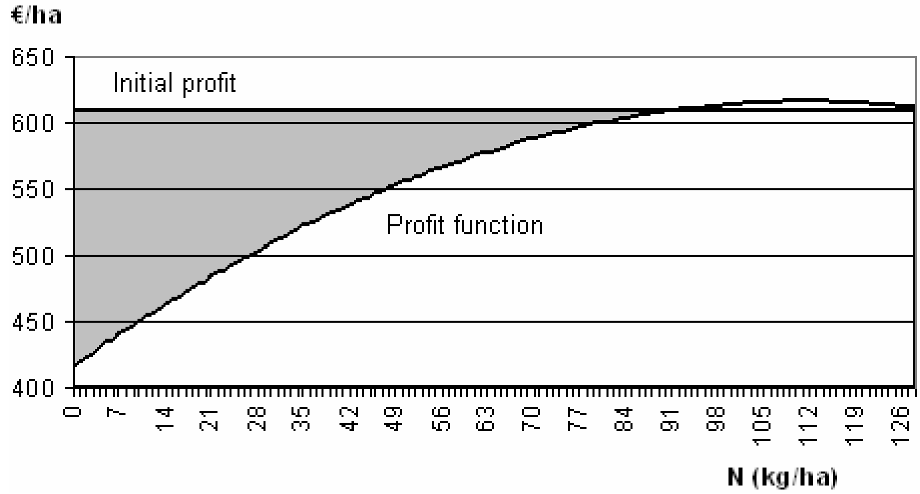

Figure 4 Profit function and initial profit from nitrogen fertilizer application.

The intuition behind this figure is the following: in the initial situation no fertiliser is reduced and the application of fertiliser is found where the two curves cross. In Figure 4 the application of nitrogen is 90 kg/ha. Note that the input of nitrogen is not at an optimal level and that the particular country could gain from an increase in fertiliser use. Compared to the optimal situation the marginal cost at the initial point is larger in Figure 4 making it even more expensive to reduce the application of nitrogen. The shaded area in the figure is the opportunity cost from reducing nitrogen application and the total area corresponds to a full reduction. This straightforward relationship can be captured directly by letting the initial profit line be a horizontal axis representing the reduction of fertiliser application per hectare. Thereby Figure 4 is literally reversed and the opportunity cost function is shown in Figure 5.

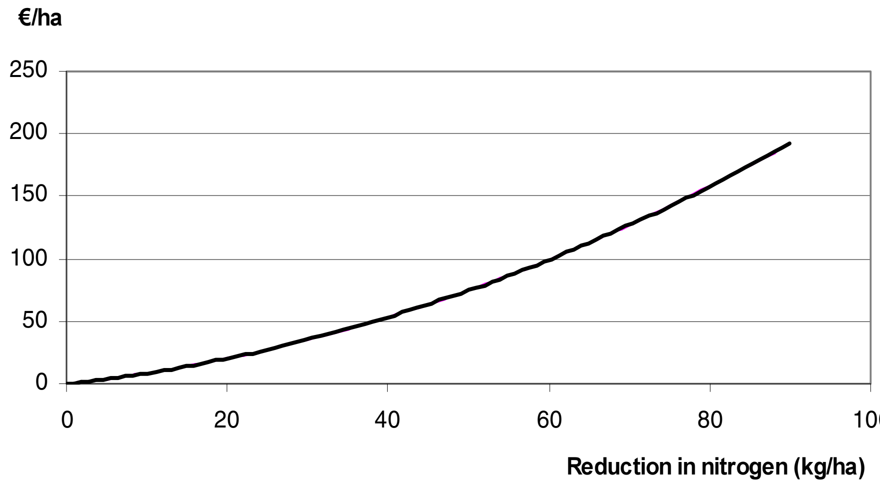

Figure 5 Opportunity cost from reducing the application of fertiliser.

The opportunity cost function is assumed to be a second order function of the form.

$$OC_{i,fertilizer} = a \cdot \Delta n + b \cdot \Delta n^2$$

Where $OC_{i,fertilizer}$	is the opportunity cost from reducing the application of nitrogen fertiliser
a and b	are cost function coefficients
Δn	is the reduction of fertiliser in kg/ha

3.5.1 Yield functions

Basically two types of models can be used for assessing the marginal abatement costs, a programming model or an econometric model. The programming models are commonly used in agronomy as it is based on yield-response functions. Yield response functions are derived from field trials showing the relationship between yields and nitrogen application.

The second methodology for establishing a quantitative relationship between profits and nitrogen application is using econometric estimations on observed data, e.g. farm account statistics. This technique is used in for example Jensen (1996) and has the advantage that it reflects actual recorded farm behaviour. The drawback of the methodology with respect to this analysis is that econometric studies and the relevant data are not found in all countries and further, methodologies and data sources vary between studies making them incommensurable.

Therefore we choose to use a yield-response based programming model for assessing the marginal costs of reduced fertiliser use. It is seen that the production intensity varies significantly between the different countries. Thus, the marginal costs of reducing fertiliser use are also expected to vary between countries, as especially the farmers in the Baltic countries operate at a different point of the yield response curve than farmers in Germany and Denmark.

In various studies (Mortensen 2000) the notion of a second order polynomial representing the yield response curve is suggested. The yield response curve is given as

$$y(n) = k + a \cdot n + b \cdot n^2$$

Where

$y(n)$ is the yield response function, hkg/ha

n is nitrogen input, kg N/ha

Based on a literature survey yield response functions have been identified for most of the countries in the model. Table 7 shows the results in terms of the parameters of the second order polynomial representing the yield response curve. For those countries where no yield response curves were found, the parameters from the neighbouring countries have been used; se Table 7 notes below.

Table 7 Yield response functions for spring barley. The response curve is a second order polynomial.

Country	k	A	b
Denmark[1]	34.7	0.345	-0.00133
Estonia[2]	31.3	0.167	-0.00083
Finland[3]	10.1	0.529	-0.00173
Germany[4]	33.3	0.520	-0.00270
Latvia[5]	31.3	0.167	-0.00083
Lithuania[6]	31.3	0.167	-0.00083
Poland[7]	31.3	0.167	-0.00083
Russian fed.[8]	31.3	0.167	-0.00083
Sweden[9]	27.1	0.224	-0.00065

1. Source: Mortensen, J.R., 2000 + personal communication.
2. Source: See 6.
3. Source: Bäckman, S., Vermeulen, S. & Taavitsainen, V-M., 1997.
4. Source: Köhn,W. et al., 2000
5. Source: See 6.
6. Source: Lazauskas S, Vaisvila Z. & Matusevicius K. (1995).
7. Source: See 6.
8. Source: See 6.
9. Source: von Blottnitz, H. et al., 2004 + personal communication

Using Denmark as an example the second order response curve is illustrated in Figure 6.

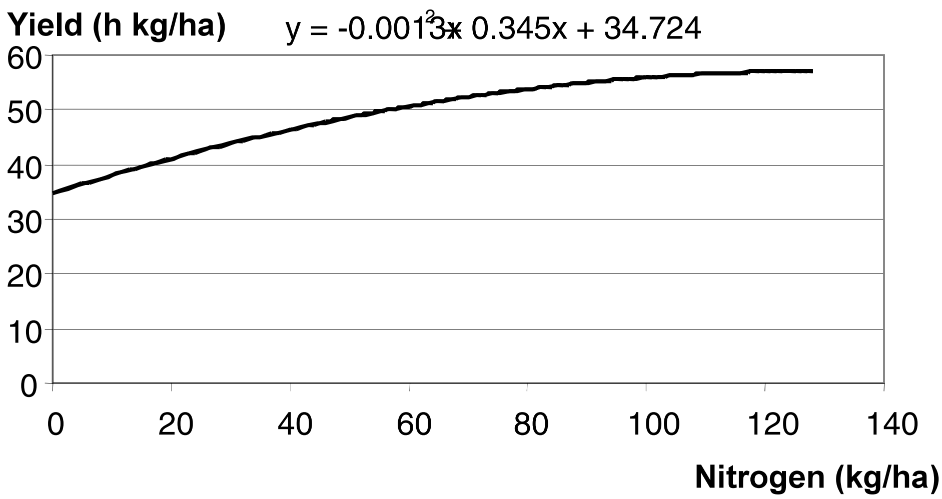

Figure 6 Yield response curve for Danish spring barley.

The figure shows the Danish nitrogen/yield function for spring barley:

$$y(n) = 34.72 + 0.345n - 0.0013n^2$$

The marginal yield response function is:

$$y'(n) = 0.345 - 0,0027n$$

The marginal revenue of fertiliser use is shown in Figure 7. As seen the optimum (marginal economic rent = 0) is found at 111 kg N per hectare, and the marginal costs of reducing N application are increasing.

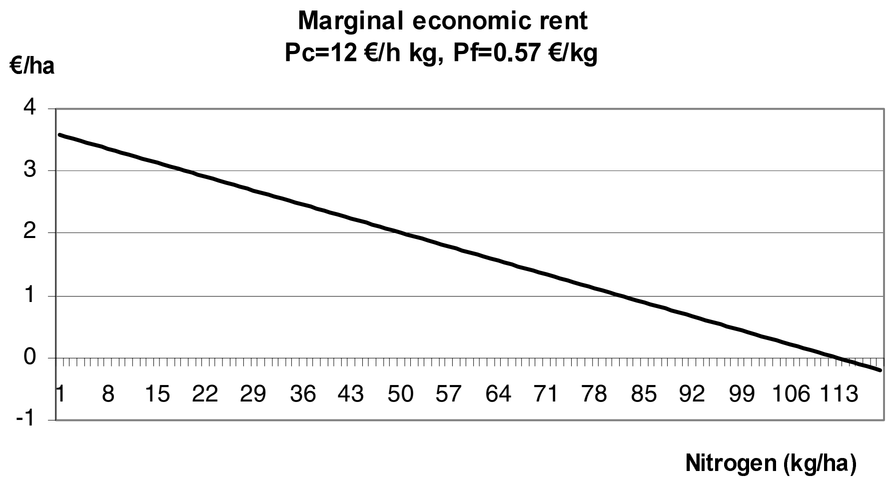

Figure 7 Marginal revenue of increased N application.

In Table 8 the average fertiliser use per hectare arable land and the average number of livestock (pigs and cattle) are shown for each country based on FAO agricultural statistics.

It is seen that the intensity of fertiliser use measured per hectare varies significantly between the countries. This is due to differences in the production intensity and to variations in the sources of total nitrogen input in agriculture, *in casu* the substitution between nutrient input from fertilisers and manure. Common for all countries the total nitrogen input is higher than total N fertiliser input. The difference occurs as manure also contributes to the total nitrogen input. Thus, the total effective nitrogen input is calculated as:

$$n_{tot} = n_f + un_m$$

Where

n_{tot} is total effective nitrogen application

n_f is total effective nitrogen applied with fertilisers

n_m is total nitrogen applied with manure

u is the utilisation of nitrogen in manure measured relative to nitrogen applied with fertilisers.

Table 8 Average agricultural fertiliser use (N) and livestock hold per hectare.

Country	Cultivated area	Agricultural fertiliser use	Number of livestock per hectare	
	1000 ha	Kg N/ha	Pigs	Cattle
Denmark	2,214	90	5,8	0,8
Estonia	3,787	42	0,5	0,4
Finland	1,985	75	0,6	0,5
Germany	1,574	152	2,2	1,2
Latvia	7,329	20	0,2	0,2
Lithuania	5,901	34	0,4	0,3
Poland	21,515	62	1,4	0,4
Russian fed.	6,385	9	1,4	2,1
Sweden	3,757	74	0,7	0,6

Source: Own calculations based on FAOSTAT (2004).

A special case is Estonia, Latvia, Lithuania and the Russian federation where the use of fertiliser has declined significantly since the liberalisation of their economies in the mid-1990s. However, in recent years fertiliser use show a tendency to increase. For these countries it may seem problematic to apply the measure due to the low initial input level and for the same reason the marginal costs of the measure are likely to be high as the current input is way below the agronomic-economic optimum. Using 2001 as baseline this issue should be reflected in the marginal reduction costs. This stresses the need for applying a long term EU-enlargement scenario where the cost-effective scenarios are developed based on the expectations of intensified agricultural production in Estonia, Latvia, Lithuania (and Poland) resulting from their inclusion in the CAP (EU Common Agricultural Policy).

The countries of Estonia, Latvia, Lithuania and the Russian federation has also had significant reductions in their livestock production following the liberalisation of their economies in the mid 1990's, and the same considerations are relevant as mentioned above.

The costs of reducing nitrogen input are dependent of both the current level of economic rent from crop production (and thus, yields and factor productivity) and the relative modelled marginal costs. Generally the lowest relative unit costs of this measure are found for the countries with a high nitrogen input and the highest costs for the countries with a low nitrogen input. The two exceptions are Lithuania, for which the low economic rent in crop production strikes through, and for Denmark, where the current regulation of the total nitrogen input to 90% of the optimal level leads to relatively high costs of further reductions in the nitrogen input.

All of the results for reduced nitrogen application have been presented per hectare, as the yield response functions are developed to analyse effects of changes in cropping intensity on a given area. When analysing the aggregated costs of the measure on a country or drainage basin scale the TAC will be a function of both the change in intensity and the area for which this change occur. Therefore the change in economic rent resulting from the change in cropping intensity needs to be multiplied by the total area on which the intensity changes. In this analysis this is assumed to be the total agricultural area in rotation in each drainage basin. The costs of reducing nitrogen fertiliser application from n_2 to n_1 kg N per hectare ($TAC_i(n_1, n_2)$) can now be calculated as:

$$TACi(n_1, n_2) = \Delta\pi_i(n_1, n_2)A_i^r = (a \cdot \Delta n + b \cdot \Delta n^2)A_i^r = OC_{i,fertilizer}A_i^r$$

Where

A_i^r is the hectares of agricultural land in rotation in drainage basin i

As A_i^r is held constant in the model analysis the TAC can be calculated by use of the formula above.

3.6 Catch crops on agricultural land

This measure also applies to agriculture and involves under sowing of catch crops (e.g. grass) when sowing the ordinary crop (e.g. barley or wheat). The catch crops have a positive effect on yields due to the extra nutrients withheld in the soil of which some are accessible to the crops in the following season. On the negative side extra costs are expected because of extra costs to seeds and possibly increased weed problems. Danish estimates show that catch crops reduce the economic rent from cash crops by about 10 percent, and this estimate is used at a country basis in this analysis. Thus, the costs per hectare can be calculated directly from the estimated economic rent in crop production (Table 10).

The highest costs are seen for Denmark and Germany and the lowest for Latvia and Lithuania. No secondary effects are expected from this measure.

Table 9 Estimated costs of catch crops on arable land.

Country	Current economic rent	Reduction in economic rent
	€/ha	€/ha
Denmark	431	43,1
Estonia	125	12,5
Finland	237	23,7
Germany	435	43,5
Latvia	72	7,2
Lithuania	44	4,4
Poland	163	16,3
Russian fed.	-	16,3
Sweden	155	15,5

Source: Own calculations.

3.7 Reduced NO_X emissions from fossil fuels

The precise interpretation and implementation of this measure needs to be discussed further with respect to the potential emission reductions from possible measures.

Cost estimates can be derived from Illerup et al. (2002) where a number of initiatives for reducing NO_X emissions from cars and power plants are analysed. The measures analysed and the results are shown in Table 11.

Table 10 Estimates of annual total socio-economic costs for different NO_X reducing measures.

Technology	Reduction costs, € per tonne NO_X
Offshore wind turbine farms	73
Installation of de-NO_X units at large power plants	17
EGR-filter installation in heavy duty vehicles	96
Electrical vehicles	1,688

Source: Illerup et al. (2002)

As seen the lowest reduction costs are found for installation of de-NO_X units at large power plants followed by substituting coal fired power plants by offshore wind turbines and EGR-filter installation in heavy duty vehicles.

In the following installation of de-NO_X units at large power plants is chosen as the abatement technology representing the NO_X – measure as this measure also is the one with the largest emissions reduction potential. Thus for a power plant producing 350 MW the emissions reduction amounts to 3,230 tonnes NO_X per year (Illerup et al., 2002).

3.8 Improved sewage treatment

There are several matters that should be taken into consideration when analyzing the costs of establishing or upgrading treatment plants, e.g. the dimension of the treatment plant and ambition for environmental treatment. In order to show the variability in the costs a number of examples are developed. The examples are based on information from the report Krüger (2001) and Winther et al. (2004). However, the estimates are considered transferable to other countries.

In the following examples of different treatments plants with different dimensions and their associated costs are given in Table 12. The estimated costs are annualized yearly costs in 2004-prices and the terminology refers to: M = Mechanic; B = Biological; N = Nitrification; D = Denitrification; and K = Chemical.

Table 11 Estimates of annual total costs for different sewage treatment technologies and sizes of treatment plants, €.

Technology	Investment costs	O and M [2)]	Total annual costs[1)]	TAC per PE
2,000 PE[3)]				
M	325,000	9,750	31,595	16
M+K	475,000	22,500	54,428	27
M+B+N	625,000	33,750	75,760	38
M+B+N+K	875,000	37,500	96,314	48
M+B+N+K+D	1,000,000	46,250	113,466	57
30,000 PE				
M	2,437,000	82,500	246,339	8
M+K	3,375,000	226,854	448,104	15
M+B+N	3,937,000	243,750	527,163	18
M+B+N+K	7,125,000	478,914	872,664	29
M+B+N+K+D	7,500,000	450,000	954,120	32
100,000 PE				
M	5,875,000	206,250	601,144	6
M+K	8,125,000	625,000	1,171,130	12
M+B+N	10,000,000	725,000	1,397,160	14
M+B+N+K	15,000,000	1,125,000	2,133,240	21
M+B+N+K+D	18,750,000	1,312,000	2,572,800	26

1) The investment costs are annualized using 3% p.a. and a depreciation period of 20 years.

2) Operation and maintenance costs

3) One PE (person equivalent) corresponds to 0,2 m^3 per day or 72 m^3 per year.

As seen in the table above the price per PE is increasing as the type of plant gets more complex and advanced. Furthermore, it is seen that the costs per unit are decreasing as the plant size increases for every type of plant.

The technologies applied are assumed to be M+B+N. In the further analysis it is assumed that all investments are done in sewage treatment plants dimensioned to process the largest multiplier of the amount of untreated waste water. Thus, for example for Poland where the amount of untreated waste water corresponds to 750,000 PE per year treatment plants are assumed to be dimensioned to 100,000 PE, where as for Denmark, where the amount of untreated waste water corresponds to 44,000 PE per year treatment plants are assumed to be dimensioned to 30,000 PE.

Based on information on the number of PE not connected to treatment plants (HELCOM, 2004), an assumption of a average yearly production of N and P per PE of 4,38 kg N/PE and 1,095 kg P/PE (Winther et al., 2004), expected rates of removal for N and P of implementing the M+B+N technology of 80 percent for N and 60 percent for P (own estimate based on Winther et al. *op cit.*), and the costs in table 12, the abatement costs are estimated.

4 Examples of secondary environmental effect

4.1 Definition

Regulating nitrogen emissions from agriculture also influences other environmental pressures such as emissions of ammonia and climate gasses. In addition regulations result in changes in land use, which directly influence the supply of goods related to biodiversity and landscape. From a socio-economic point of view these secondary benefits should be reflected in the cost estimates.

As these secondary benefits are closely related to the fulfilment of countries' obligations, e.g. under the recently implemented Kyoto protocol and the EU Habitat Directive, there is also an increased administrative attention to include them in policy analyses. When preparing the third Danish Aquatic Action Plan in 2003-04 an attempt was therefore made to quantify the secondary environmental effects in terms of air emissions and include these in the economic analysis using the shadow price approach. Secondary benefits resulting from changes in biodiversity and landscape were, however, not considered.

In this section we demonstrate how secondary environmental effects can be included in the cost measures. Due to uncertainties in data and methodologies the assessments are only made for the measures wetlands and reduced livestock hold. The estimates should be seen as examples on how secondary benefits can be included in cost measurements, but is not exhaustive as some of the other measures lead to secondary benefits that are not analysed here.

4.2 Wetlands

The establishment of wetlands leads to a number of other (secondary) environmental effects besides reducing nitrogen and phosphorous losses. These encompass changes in air emissions, changes in biodiversity and recreational goods. With respect to biodiversity and recreational goods the benefits are expected to be highly site specific and country dependant. Further, estimates of the benefits are only available for a small number of goods and countries, making the use of benefit transfer not feasible. Therefore, in this analysis we only include secondary benefits from reduced ammonia and climate gas emissions, as the scale other types of benefits are site specific to an extend which makes general benefit transfer problematic; see Schou & Birr-Pedersen (2005) for further discussion.

When agricultural land is drained the organic layers are decomposed leading to emissions of CO_2. Similarly conversion of drained land into wetlands is expected to lead to accumulation of CO_2 in the soil. Establishment of wetlands may also lead to increased air emissions in terms of CH_4 (methane) and N_2O (laughing gas). The effect on methane occurs due to a change in the balance between methane formation and methane oxidation, and laughing gas is a by-product of nitrification.

The total effects on climate gasses have been estimated based on a literature survey as part of the preparation of the third Danish Aquatic Action Plan (Petersen, 2004). The effects were given as *min*, *max* and *mean* in order to reflect the large variations to be expected between locations and, thus, the uncertainty related to general estimates (Table 12).

Table 12 Estimated effects on climate gas emissions from establishment of wetlands, kg CO_2-equivalents/ha per year.

Emission	Mean estimate	Min estimate	Max estimate
Carbon	-14,670	-33,000	-11,000
Methane	8,560	1,680	19,600
Laughing gas	-2,780	-13,150	0
Total effect	**-8,880**	**-44,470**	**8,600**

Note. Negative values represent a reduction in emissions.

The estimated mean effect on climate gas emissions is an annual reduction of 8,900 kg CO_2-equivalents per hectare. Variations in soil types, hydrology and management are estimated to lead to variations ranging from a reduction of 44,500 kg CO_2-equivalents per hectare to an increase in emissions of 8,600 kg CO_2-equivalents per hectare.

In order to include the secondary effects in the cost estimates it is necessary to attach values to the physical effects. These should reflect the marginal benefit of reducing the environmental impact resulting from each type of emission or from providing one extra unit of the environmental goods.

With respect to air emissions an extensive work has been done in the ExternE project in deriving estimates of the marginal damages resulting from energy related air emissions (ExternE, 2005). In the ExternE analysis two different models are used for analysing the damages resulting from scenarios of global warming. The overall marginal damages calculated by the two models are in good agreement whereas variations are found with respect to the different damage components (ExternE, 2003, p 65). Also the damage estimates were subject to a sensitivity analysis and based on this a range of marginal damage estimates from 3,8 to 139 € per tonnes CO_2-equivalents in 1995-prices were deducted. Within this interval the suggested range for the aggregate marginal damages of climate gas emissions were 18 to 46 € per tonnes CO_2-equivalents. No mean estimate is given, which reflects the significant uncertainty related to the climate scenarios and the resulting marginal damages.

Therefore it can be discussed if the inclusion of the effects on climate gas emissions is priced meaningfully by use of the very uncertain estimates of marginal damages. As a number of countries have taken initiatives to reduce climate gas emissions an alternative way of pricing the effects is the shadow price method. Using the shadow price approach the value of derived environmental emissions is estimated by the marginal costs of reducing the emissions. In the optimal economic world marginal reduction costs and marginal damages should correspond, but in a more realistic second best policy setting the shadow price reflects society's (or the politicians) willingness to pay for reducing the uncertain damages resulting from global warming. This is the drawback of the method seen from an economic theoretical point of view.

The approach can only be applied for including secondary benefits. Further, it requires an explicit target for reducing the emissions, and the existence of a cut-off price or estimates of the marginal willingness to pay for reducing the emissions.

Estimates of the future compliance costs (or "cut off prices") for the European Union have been carried out by the Commissions given a number of policy constraints (European Commission, 2003). The scenarios range from no possibilities of applying the measures of Joint Implementation (JI) and Clean Development Mechanism (CDM) in the Kyoto Protocol to allowing for the use of these measures to the full extend. The resulting price estimates range from 11 - 26 € per tonnes CO_2-equivalent.

For the EU countries no restrictions were put on the use of JI and CDM, and therefore the cut off price of 11 € per tonnes CO_2-equivalent is used as shadow price for reduction of climate gasses. Combining this price with the mean reduction in climate gas emissions from establishment of wetlands (Table 12) of 8,881 CO_2-equivalents per hectare the resulting secondary benefit is estimated to 98 € per hectare.

Combining this benefit from climate gas reductions with the opportunity costs, the net costs of establishing wetlands can be derived for the single countries. It is found that for most countries the net costs of establishing wetlands are negative. But for Latvia and Lithuania the measure represents a net benefit as the secondary environmental benefits exceed the costs of establishing wetlands. The argument is that the reduced climate gas emissions can be traded with other countries that have high costs of complying with the Kyoto-commitments, and thus, represent a welfare economic benefit (option value) to the two countries.

4.3 Reduced livestock production

Reduction of husbandry production leads to a number of other environmental effects including reductions of ammonia and methane emissions and these secondary benefits are included in the net unit costs using the shadow price method. The measure has recently been analysed in Schou et al. (2004) and here the effects on ammonia emissions and climate gas emissions are shown in Table 13.

Table 13 Effects on ammonia emissions and climate gas emissions from reduced livestock hold.

Livestock type	Ammonia	Climate gas reduction
	KG N/LU	*Kg/LU*
Pigs	31.5	1,300
Cattle	17.4	2,800

Note. One Livestock Unit (LU) corresponds to a production of 100 kg N in manure. In the analysis it is assumed, that N in fertilisers substitutes the reduction in N input from manure.

Source: Schou et al., 2004.

In order to include these secondary effects in the cost estimates an economic value needs to be attached to the physical effects. With respect to climate gas reductions the same shadow price is used as the one used for wetlands, i.e. 11 € per tonnes CO_2-equivalent. Also, the shadow price approach can be used for reduced ammonia emissions, but here it does not seem reasonable to use a uniform price as the problem of eutrophication from ammonia deposition varies across countries. In a case study by Schou (2003b) a shadow price of approximately 1 € per kg NH_4 estimated based on the Danish Ammonia Action Plan, but obviously this value is not valid for all of the 7 countries. Therefore the effect on ammonia emissions is not included in the cost estimates.

5 Example: Reducing aggregate N loads by 160,000 tonnes

The starting point of the cost minimisation problem is setting the target for reduction in loads in one or more sea regions. To keep the example simple only reduction of nitrogen (N) is included in this scenario, but the model is capable of including simultaneous reduction of phosphorous loads in the environmental target. The targets used in the example are an aggregate reduction of 160,000 tonnes N distributed to Bothnian Bay 20,000 tonnes N; Bothnian Sea 25,000 tonnes N; Baltic Proper 30,000 tonnes N; Gulf of Finland 30,000 tonnes N; Gulf of Riga 25,000 tonnes N; Danish Straits 10,000 tonnes N; Kattegat 20,000 tonnes N.

Table 16 shows the results in terms of costs distributed between countries and measures given exogenous target reductions in each sea region; a spreadsheet with the full results of the model run are available on request. With respect to the reduction in emissions and loads, two points should be stressed. First, because of the flow of nutrients between sea regions the aggregate modelled reduction is larger than the aggregate target (323,000 vs. 160,000 tonnes N). Secondly, because of the interconnection of the sea regions and synergy effects related to increased metabolism of nutrients when loads are reduced, the actual reduction in the loads are higher than the reductions in emissions from sources (323,000 vs. 184,100 tonnes N) even though retention during transport of the nutrients in the drainage basins are included in the load-response modelling (see Savchuk, 2004).

Table 14 Example of scenario run: Costs distributed by measures and countries, 1000 €.

	Wetland	NFertilizer	LandUse	Cattle	Pigs	Sewage	NOx	Total
DE	0	14.303	0	0	0	0	35	14.337
DK	0	2.957	0	0	0	0	52	3.009
EE	16.424	28.282	16.366	4.095	0	0	52	65.220
FI	11.059	72.395	14.089	120.087	30.812	5.164	52	253.658
LT	0	31.013	8.568	0	0	0	17	39.599
LV	14.691	34.326	17.405	33.977	0	0	35	100.433
PL	0	113.390	115.642	0	0	0	52	229.085
RU	0	14.090	26.376	0	0	0	35	40.501
SE	7.198	3.163	12.563	10.786	2.978	4.368	74	41.131
Total	**49.373**	**313.919**	**211.010**	**168.945**	**33.790**	**9.532**	**405**	**786.974**

The intuition behind the results in the table is that each country in succession utilizes the measure with the smallest marginal cost until the targets are reached. Generally NO_X from power plants reduction is the measure resulting in the least costs. However, due to the low potential of reducing nitrogen loads of this measure, they only contribute marginally to the reduction. The least costs within agriculture are found for reduced fertiliser use, followed by wetlands and land use changes (catch crops). The reduction of nitrogen from livestock has the largest marginal cost and therefore cattle- and pigs reductions are employed as the last measures in the model. It is also seen that different measures are selected in the different countries; this reflects the difference in marginal costs. For example reduced fertiliser is not chosen in countries with a low initial fertiliser use, because this implies high marginal costs. Comparing the costs with the environmental effects it is seen that the per unit reduction cost of aggregate N loads are 2.4 € per kg N. In order to get an impression of the validity of the estimates, the reduction costs in terms of € per unit of reduced N emission (N leaching) can be compared to the results from the economic evaluation third Danish Aquatic Action Plan (Jacobsen et al., 2004). This analysis showed per unit reduction costs for N emissions for Denmark of 3.2 € per kg N (or 24 DKK per kg N) for the measure reduced fertiliser use (this can not be deducted from the table). Results of Jacobsen *op cit*. show marginal reduction costs of reducing fertilizer use by an extra 10% in addition to the already implemented 10% cut (20% all in all) of 28 DKK per kg N. Thus, compared to those analyses performed to support the third Danish Aquatic Action Plan, the level of reduction costs seem reasonable.

It should be noted, that the current model version works under the assumption that both costs and load effects occur at the same time. This may not be the case in the real world, and therefore an additional version of the cost-minimisation model has been programmed enabling the setting of time-lags between the implementation of a measure and the occurrence of the load effects. The effects of time-lags on the cost minimising abatement strategies are reported in a separate paper (Møller, Neye and Schou, 2006).

When presenting the results to policy makers it should be stressed that although the result is presented as an aggregate cost estimate for the countries in the Baltic Sea region this should not be interpreted as an indication of which countries that eventually should bear the costs. This is important to note when passing the results to policy makers, as the model prescribes how the effort should be mixed in order to reach the least cost solution but <u>not</u> how this solution is reached in a financial and political economic context.

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Appendix 1: The load response modelling

This appendix describes the principles of the load modelling in the cost minimisation model, and the parameters used at present. The parameters are derived from a literature survey of studies dealing with aggregate or general effects on N and P emissions resulting from changes in agricultural production and retention of N and P during transport in streams and lakes.

Steps in the load modelling

The aim of the load response functions is to express the quantitative relationship between the scale of implementation of measure for reducing nutrient emissions in a given drainage basin and the change in nutrient loads to a given sea region. First a number of central terms when describing the load response relationships are explained in the table below.

Table. A1.1 Central terms [denotation of the function]

Emission [e]	Outlets of a pollutant, e.g. nitrate, from the source
Retention [rt]	Expresses the share of emissions degraded or withheld during transport to the recipient
Load [l]	The amount of a pollutant, e.g. nitrate, reaching the recipient
Impact coefficient [b]	Coefficient (together with the retention) describing the relationship between emissions and loads
Effects [ef]	Changes in the state of a recipient resulting from changes in loads
Measure	Term describing an initiative aiming at reducing (environmental) effects

Denoting the scale of implementation of measure *k* in drainage basin *i* x_i^k the first functional relationship to be quantified is the change in emissions: $e_i^k = f_i^k(x_i^k)$. Note that the emissions function will be specific for each measure (*k*) and may be specific with respect to drainage basins too (e.g. depending of soil types and climate).

Knowing the change in emissions resulting from a specified implementation of measure *k*, the change in loads from drainage basin *i* to sea region *j* can be denoted: $l_{ij}^k = (1 - rt_{ij})b_{ij}e_i^k$.

Because of transport of nutrients between the sea regions the impact coefficient is an important input to the minimisation problem.

Knowing the change in loads to sea region *j* the final effects, e.g. in transparency, are quantified by: $ef_j = f(l_{ij}^k)$. This last step is not part of the economic model, but is added as a satellite-model.

Definitions of measures and parameters in the load functions

The measures incorporated in the revised model are:

1. Wetland restoration
2. Reduced fertiliser use
3. Introduction of catch crops in agriculture
4. Livestock reduction in agriculture
5. Improved treatment of sewage
6. NO_X reduction

First the measures are described quantitatively and then the parameters of the emissions functions $e_i^k = f_i^k(x_i^k)$ are presented. Note that the function for a given measure is assumed to be the same for all drainage basins, i.e. $f_1^k = f_2^k = f_n^k$. Last, the retention coefficients are presented for the combination of countries and sea regions.

Wetlands

This involves conversion of agricultural land into wetlands. The feasible number of hectares in each drainage basin (and country) is estimated based on historical land use statistics.

The measure is only relevant in third and fourth order streams in relation to agricultural production. This is reflected by estimating the maximum (potential) hectares to be concerted into wetlands as a fraction of the agricultural area. As a starting point conversion of wetlands are assumed to take place on maximum five percent of the agricultural area.

For this measure x is the number of hectares agricultural land converted into wetlands.

For each hectare converted into wetland 100 kg/ha of nitrogen is reduced and 5 kg/ha for phosphorus (Ministeriet for Fødevarer, Landbrug og Fiskeri, 2003a & b).

Reduced nitrogen fertiliser use

This measure applies to agricultural production and is implemented by a uniform reduction in fertiliser use on all of the agricultural area in rotation.

Emission effects of reduced fertiliser use are calculated as a fixed fraction of the change in nitrogen input, but as the fraction is dependant of nitrogen input the coefficients vary between countries. The fractions used are from Finish Environmental Institute report no. 524, 2002: Evaluation of the implementation of the 1988 ministerial declaration regarding nutrient loads reduction in the Baltic Sea area.

For this measure x is the reduction in nitrogen application measured in kg per hectare of agricultural area in rotation.

The maximum reduction in each country is set at 25 percent, estimated based on FAO-statistics of agricultural land use and fertiliser use.

The reduction of nitrogen by fertilizer reduction is measured by the difference between the initial level of fertilizer use pr hectare and the new level multiplied by the size of arable land. The reduction coefficient is 33 percent.

NLoadReductionFertilizer = 0.33 * (IniNFertilizerUse - NFertilizerUse) * Arable(ha)

Catch crops on agricultural land

This measure also applies to agriculture and involves under sowing of catch crops (e.g. grass) when sowing the ordinary crop (e.g. barley or wheat).

For this measure x is the number of hectares agricultural land with catch crops.

The feasible hectares with catch crops are set as 1/3 of the agricultural area in rotation, estimated based on FAO-statistics of agricultural land use.

For each hectare catch crops result in a nitrogen emissions reduction of 3.5 kg/ha. The coefficient is assumed to be the same for all countries as it mainly involve reduction of leaching after harvest and, thus, reduces the leaching from organic nitrogen mobilised in the period between harvest and sowing.

NLoadReductionLandUse = 3,5 * LandUseChange

Reduced livestock hold

Cattle

This measure applies to agricultural livestock production. The measure only relates to nutrient leaching resulting from application of manure on the field. In the current model an average of 17% of total N production and 3% of total P production is assumed to be emitted to water bodies. The emission per animal can therefore be calculated based on the nutrient production per animal. Claesson and Steinneck (1991) show an interval between 75 and 115 kg N/year and 13 and 18 kg P/year for dairy cows with a yearly milk production of 5,000 and 9,000 DKK per year. For non-dairy cows the levels are approx. 38 kg N and 4 kg P.

In this analysis the emission coefficients are estimated based on the figures of Claesson and Steinneck (1991) and the average milk production per cow according to FAOSTAT in order to reflect differences in production intensity. For other cows the emissions are assumed to be at the levels stated by Claesson and Steinneck (1991). As the cost functions are estimated for an "average" cow the emissions are calculated for the same unit by assuming a distribution of the cattle stock of 80% dairy cows and 20% other cows.

The data and coefficients used are shown in the table below.

Table A1.2 N emissions per "average" cow. Assuming 80% dairy cows and 20% other cows. All figures are in kg per animal per year.

Country	Production		Average nutrient production	Emission coefficient
	Dairy cows	Other cows		
Denmark	96	32	84	14
Estonia	74	32	67	11
Finland	94	32	83	14
Germany	85	32	75	13
Latvia	64	32	58	10
Lithuania	62	32	57	10
Poland	63	32	58	10
Russian fed.	47	32	45	8
Sweden	102	32	89	15

Source: FAOSTAT; Claesson and Steinneck (1991); and own calculations.

Pigs

For pig sows with piglets the yearly output is 26 kg N and 10 kg P, for slaughter pigs the output is 9 kg N and 2 kg P. The distribution of the pig stock is assumed to be 30% slaughter pigs and 70% other pigs. This results in an average nutrient production per pig of 21 kg N and 8 kg N. Using the emissions ratios of 17 % for N and 3 % for P this leads to emissions coefficients of 3,6 kg N and 0,2 kg P per pig.

Table A1.3 P emissions per "average" cow. Assuming 80% dairy cows and 20% other cows. All figures are in kg per animal per year.

Country	Production		Average nutrient production	Average emission coefficient
	Dairy cows	Other cows		
Denmark	16	5	14	0,4
Estonia	13	5	12	0,4
Finland	16	5	14	0,4
Germany	15	5	13	0,4
Latvia	12	5	11	0,3
Lithuania	12	5	10	0,3
Poland	12	5	11	0,3
Russian fed.	10	5	9	0,3
Sweden	17	5	15	0,4

Source: FAOSTAT; Claesson and Steinneck (1991); and own calculations.

For this measure x is the number of heads with pigs and cattle. Pigs and cattle are modelled separately.

The maximum reduction in livestock hold in each country is set at 80 percent, estimated based on FAO-statistics of agricultural livestock.

Improved sewage treatment

This measure is as a starting point based on the number of PE (person equivalents) producing untreated municipal waste water.

For this measure x is the extra number of PE (person equivalents) whose waste water will change from no treatment to undergo M+B+N-treatment.

The maximum amount being treated is set to 20 percent of the PE not connected to waste water treatment in the reference. The technologies applied are assumed to be M+B+N, which are resulting in a removal of 80 percent of the N and 60 percent of P. The N and P content in the waste water are estimated based on emission coefficients (e_s^N, e_s^P) of 4,38 kg N per PE and 1,095 kg P per PE (Winther et al., 2004). Thus the N and P removal and, thus, changed emissions $(\Delta N, \Delta P)$ can be modelled as:

$$\Delta N = 0.8 e_s^N x \quad \text{and} \quad \Delta P = 0.6 e_s^P x$$

Reduced NO_X emissions from fossil fuels

This measure involves installation of de-NO_X units at large power plants.

For this measure x is the amount of NO_X abated.

The maximum scale of the measure is set arbitrarily 1,000 ton per year.

Retention and dilution coefficients

For those measures where the nutrients are transported through streams before reaching the sea retention will occur. The retention of nitrogen before entering the sea regions varies from one drainage basin to another. The amount of nitrogen in a sea region from a drainage basin is calculated using a drainage basin specific coefficient.

$$\Delta L = \Delta e_j^i (1 - \frac{r_i}{100}),$$

where ΔL is the change in loads,

Δe_j^i is the change in emissions from drainage basin i,

r_i is the retention in drainage basin i measured in percent.

The river retention coefficients are all taken from the PLC-4 report of HELCOM as reported in the table below.

Table A2.4 Retention coefficients.

Parameter NRetentionLand(l) 'Retention of nitrogen in drainage basin (percent)		
	P	N
DE_BP	60	34
DE_DS	60	34
DK_BP	1,8	10
DK_DS	1,8	10
DK_KT	1,8	10
EE_BP	36	23
EE_GF	36	23
EE_GR	36	23
FI_BB	26	29
FI_BS	26	29
FI_GF	26	29
LT_BP	2,3	0,9
LV_BP	40	45
LV_GR	40	45
PLO_BP	38	44
PLO_BP	38	44
PLO_BP	38	44
RU_BP	60	60
RU_GF	60	60
SE_BB	28	24
SE_BP	28	24
SE_BS	28	24
SE_DS	28	24
SE_KT	28	24

Source: HELCOM (2004)

Note that the issue of large cities (e.g. St. Petersborg) is not reflected, as some of the waste water from here is emitted directly into the Baltic Sea. Retention will be low for these sources.

For NOx the amount for emissions that eventually will deposit on the sea regions are modelled using a set of country specific dilutions coefficients. The calculation routine is similar to that used for retention.

The emission reductions are equal to the coefficients from the model in NEST1 shown in table A1.5.

Table A1.5 Dilution coefficients for NO_X.

Country	Percent
Denmark	14.9
Estonia, Baltic Proper	4
Estonia, Gulf of Finland	9.56
Estonia, Gulf of Riga	6
Finland	3.43
Germany	5.51
Latvia, Baltic Proper	8
Estonia, Gulf of Riga	8.36
Lithuania	9
Poland	5.73
Russian fed.	2.07
Sweden	11

Source: NEST 1

Appendix 2: Agricultural economic reference

Table A.2.1 shows the value of agricultural output and economic rent for the countries included in the cost-minimisation model except for the Russian federation for which data have not been retrieved. The data include both crop and livestock production. The calculation of economic rent corresponds to the method shown for calculating the unit costs, if the value of the secondary environmental effects is excluded.

Measured per hectare the average economic rent in agriculture varies from 1,000 €/ha in Denmark to 110 €/ha in Latvia indicating significant differences in production intensity as well as outputs.

Table A2.1 Agricultural output and economic rent 2001.

Country	Output	Economic rent	Output	Economic rent
	M €	*M €*	*€ per ha*[1]	*€ per ha*[1]
Denmark	8,348	2,288	3,640	1,000
Estonia	475	149	700	220
Finland	4,288	881	1,960	400
Germany	41,454	9,236	3,510	780
Latvia	587	208	320	110
Lithuania	1,067	189	3,640	650
Poland	13,421	3,610	0,960	260
Russian fed.	-	-	-	-
Sweden	4,710	828	1,750	310
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SUM / Average	74,350	17,389	1,544	361

1) Per hectare arable land

Source: Own calculations based on Eurostat (2003) and FAO (2004).

However, the figures shown in Table A2.1 cannot be used directly as they include the economic outcome from both crop and livestock production, and the measure is assumed only to reduce the production potential in crop production. The estimates of economic rent are not divided into livestock and crop production by Eurostat, but this is done with respect to total production value. However, this cannot directly be used for splitting economic rent into crop and livestock production, if the cost structure is different. A review of Danish farm account statistics from 1991 to 2002 shows that economic rent corresponds to 6 percent of the total production value in the crop production and 8 percent of the total production value in the pig production but with large year-to-year variations.

Therefore, as a rough estimate, economic rent in the agricultural sector is divided into livestock and crop production based on the share of each sector to the total value of production. In Table A2.2 the estimates are shown at a country basis.

Table A2.2 Estimated economic rent in livestock and crop production 2001, M €.

Country	Value of production			Economic rent	
	Livestock	Crops	Share of livestock	Livestock	Crops
Denmark	4,707	3,573	57%	1,300	987
Estonia	172	226	43%	64	85
Finland	1,340	1,932	41%	361	520
Germany	17,967	22,569	44%	4,094	5,142
Latvia	183	318	37%	76	132
Lithuania	343	751	31%	59	129
Poland	5,359	9,222	37%	1,327	2,283
Russian fed.	-	-	-	-	-
Sweden					
----------	1,970	2,015	49%	409	418
SUM / Average	32,041	40,606	44%	7,669	9,720

Source: Own calculations based on Eurostat (2003).

Also, economic rent is divided between pigs and cattle based on the share of the total production value resulting from each livestock type. Results are shown in Table A2.3.

Table A2.3 Estimated economic rent in livestock production 2001, M €.

Country	Value of production			Economic rent	
	Pigs	Cattle	Share of pigs	Pigs	Cattle
Denmark	2,848	1,859	61%	787	514
Estonia	62	110	36%	23	41
Finland	282	1,058	21%	76	285
Germany	6,147	11,820	34%	1,401	2,693
Latvia	62	121	34%	26	50
Lithuania	132	211	38%	23	36
Poland	2,917	2,442	54%	722	605
Russian fed.	-	-	-	-	-
Sweden	431	1,539	22%	90	320

SUM / Average	12,881	19,160	40%	3,083	4,586

Source: Own calculations based on Eurostat (2003).

The total estimated economic rent from the two types of livestock production is transferred into unit costs by dividing loss in economic rent by the number of animals. The economic rent per hectare from crop production is calculated by dividing the total economic rent from crop production by the agricultural area. Results are shown in Table A2.4.

Table A2.4 Economic rent per hectare and per head of livestock, 2001-prices.

Country	Pig production	Cattle production	Crop production[1)]
	€/head	*€/head*	*€/ha*
Denmark	59	295	431
Estonia	68	162	125
Finland	59	282	237
Germany	53	196	435
Latvia	57	130	72
Lithuania	22	47	44
Poland	38	111	163
Russian fed.	-	-	-
Sweden	47	194	155
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Average	38	89	202

1) Per hectare arable land.

Source: Own calculations based on Eurostat (2003) and FAO.

Appendix 3: Data in the model

Data structure

In the model the Baltic Sea is divided into seven sea region (Table C1 in this appendix) surrounded by nine countries (Table C2 in this appendix). The sea regions are sub-divided into a total of 24 drainage basins (Table C3 in this appendix). The classification of drainage basins and the geographical linkage to sea regions and countries correspond more or less to the classification used in PLC-4 (HELCOM, 2004). In the current model, six different measures are available for reducing nutrient load to the Baltic Sea (Table C4 in this appendix); the model, however, has been prepared for the inclusion of two more measures as two blank measures are included in the programming ('blank1' and 'blank2'). In the remainder of this appendix variables referring to these blank measures will be disregarded as they carry no information relevant for the model (i.e. they are basically set to zero).

List of input data

In the following the data needed to run the model is listed. The 'content' column contains a brief verbal description of the data input, the 'unit' column specifies the unit in which the data is entered while the 'parameter' column states the parameter name attached to the data in GAMS. Finally, the 'reference' column specifies where the data used in the present model can be found.

Table A3.0 Model input data.

Content	Unit	Parameter	Reference
Target load reduction of nitrogen in each sea region	Ton	NTargetLoadReductionSea (S)	To be specified prior to each run of the model.
Target load reduction of phosphorus in each sea region	Ton	PTargetLoadReductionSea (S)	To be specified prior to each run of the model.
Area of drainage basin	Ha	DrainageBasinArea	Table A3.5 in this appendix
Arable land in catchment area (at drainage basin level)	Ha	LandCover (ARABLE)	Table A3.5 in this appendix
Initial holdings of cattle in catchment area (at drainage basin level)	100 heads	Cattle	Table A3.5 in this appendix
Initial holdings of pigs in catchment area (at drainage basin level)	100 heads	Pigs	Table A3.5 in this appendix
Content of nitrogen from cattle in drainage basin	ton N/100 heads	NLoadRedCoeffCattle	Table A3.7 in this appendix
Content of nitrogen from pigs in drainage basin	ton N/100 heads	NLoadRedCoeffPigs	Table A3.7 in this appendix
Content of phosphorus from cattle in drainage basin	ton P/100 heads	PLoadRedCoeffCattle	Table A3.7 in this appendix
Content of phosphorus from pigs in drainage basin	ton P/100 heads	PLoadRedCoeffPigs	Table A3.7 in this appendix
Base loads of nitrogen in sea regions	Ton	NBaseLoadSea	Table A3.6 in this appendix
Base loads of phosphorus in sea regions	Ton	PBaseLoadSea	Table A3.6 in this appendix
Initial amounts of untreated waste water at country level	m3	WasteWater	Table A3.10 in this appendix
Nitrogen load reduction by sewage treatment	percent	NLoadRedCoeffSewage	Table A3.14 in this appendix
Phosphorus load reduction by sewage treatment	percent	PLoadRedCoeffSewage	Table A3.14 in this appendix
Average content of nitrogen in waste water	ton/m3	NLoadCoeffSewage	Table A3.14 in this appendix
Average content of phosphorus in waste water	ton/m3	PLoadCoeffSewage	Table A3.14 in this appendix
Dilution of nitrogen by installing de-NOx units	Percent	NLoadRedNOx	Table A3.16 in this appendix
Retention of nitrogen at drainage basin level	percent	NRetentionLand	Table A3.13 in this appendix
Retention of phosphorus at drainage basin level	percent	PRetentionLand	Table A3.13 in this appendix
Load transport of nitrogen between sea regions	-	NLoadFlow	Table A3.11 in this appendix
Load transport of phosphorus between sea regions	-	PLoadFlow	Table A3.12 in this appendix
Nitrogen load reduction by wetland restoration	ton/ha	NLoadRedCoeffWetland	Table A3.15 in this appendix
Phosphorus load reduction by wetland restoration	ton/ha	PLoadRedCoeffWetland	Table A3.15 in this appendix
Nitrogen load reduction coefficient for fertilizer reduction	percent	NLoadRedCoeffFertilizer	Table A3.15 in this appendix
Nitrogen load reduction coefficient for land use change	ton/ha	NLoadRedCoeffLandUse	Table A3.15 in this appendix
Measures control – specify which measures that are available for use at drainage basin level	0/1 variable	MeasuresControl	Table A3.17 in this appendix
Restriction for reduction of fertilizer	percent	MAXFertilizerReduction	Table C9 in this appendix

Average application of nitrogen fertilizer	kg/ha	IniNFertilizerUse	Table A3.8 in this appendix
Price of nitrogen fertilizer	€/kg	NFertilizerPrice	Table A3.8 in this appendix
Average price of crops	€/hkg	CropPrice	Table A3.8 in this appendix
Cost calculation factors for reduction of fertilizer use	-	-	See the specific cap-tor in the report
Secondary benefits of reduced fertilizer use	-	-	Set to zero in the current model
Restriction on wetland restoration	percent	MAXWetlandRestoration	Table A3.9 in this appendix
Cost calculation factors for wetland restoration	-	-	See Tables 1 and 6 in the report
Secondary benefits of wetland restoration	-	-	Set to zero in the current model
Restriction on land use change – i.e. use of catch crops	percent	MAXLandUseChange	Table A3.9 in this appendix
Cost calculation factors land use changes	-	-	See Table 10 in the report
Secondary benefits of land use changes	-	-	Set to zero in the current model
Restriction on sewage treatment	percent	MAXSewageTreatment	Table A3.9 in this appendix
Cost calculation factors for sewage treatment	-	-	See Table 13 in the report
Secondary benefits of sewage treatment	-	-	Set to zero in the current model
Restriction on livestock reductions	percent	MAXCattleReduction / MAXPigsReduction	Table A3.9 in this appendix
Cost calculation factors for livestock reductions	-	-	See Table 6 in the report
Secondary benefits of livestock reductions	-	-	Set to zero in the current model
Restriction on NO_X reduction	ton	MAXNOxReduction	Table A3.9 in this appendix
Cost calculation factors for NO_X reduction	-	-	See table 11 in the report
Secondary benefits of NO_X reduction	-	-	Set to zero in the current model

List of tables

Table A3.1 Sea regions in the model.

Sea region	Abbreviation
Bothnian Bay	BB
Bothnian Sea (incl. the Archipelago Sea)	BS
Baltic Proper	BP
Gulf of Finland	GF
Gulf of Riga	GR
Danish Straits (Western Baltic and The Sound)	DS
Kattegat	KT

Table A3.2 Countries included in the model.

Country	Abbreviation
Germany	DE
Denmark	DK
Estonia	EE
Finland	FI
Lithuania	LT
Latvia	LV
Poland	PL
Russia	RU
Sweden	SE

Table A3.3 Drainage basins.

Drainage Basin	Abbreviation
'Germany – Baltic Proper'	DE_BP
'Germany – Danish Straits'	DE_DS
'Denmark – Baltic Proper'	DK_BP
'Denmark – Danish Straits'	DK_DS
'Denmark – Kattegat'	DK_KT
'Estonia – Baltic Proper'	EE_BP
'Estonia – Gulf of Finland'	EE_GF
'Estonia – Gulf of Riga'	EE_GR
'Finland – Bothnian Bay'	FI_BB
'Finland – Bothnian Sea'	FI_BS
'Finland – Gulf of Finland'	FI_GF
'Lithuania – Baltic Proper'	LT_BP
'Latvia – Baltic Proper'	LV_BP
'Latvia – Gulf of Riga'	LV_GR
'Poland Coast – Baltic Proper'	PLC_BP
'Poland Oder – Baltic Proper'	PLO_BP
'Poland Vistula – Baltic Proper'	PLV_BP
'Russia – Baltic Proper'	RU_BP
'Russia – Gulf of Finland'	RU_GF
'Sweden – Bothnian Bay'	SE_BB
'Sweden – Baltic Proper'	SE_BP
'Sweden – Bothnian Sea'	SE_BS
'Sweden – Danish Straits'	SE_DS
'Sweden – Kattegat'	SE_KT

Table A3.4 Measures for reducing nutrient loads to the Baltic Sea.

Measure	'Abbreviation'
Restoration of wetlands	'Wetland'
Reduced fertilizer use	'Fertilizer'
Sowing of catch crops	'Land Use'
Reduced livestock – i.e. cattle and/or pigs – production	'Livestock'
Improved sewage treatment – i.e. treatment of currently untreated wastewater	'Sewage'
Reduced NO_X emissions from fossil fuels	'NO_X'

Table A3.5 Baseline information at drainage basin level on aspects related to agricultural production.

Drainage Basin	Area of Drainage basin (ha)	Arable land in catchment area (ha)	Holding of cattle in catchment area (in 100 heads)	Holding of pigs in catchment area (in 100 heads)
DE_BP	1.194.626,4	849.038,2	79.014	140.163
DE_DS	977.330,9	726.007,2	67.565	119.852
DK_BP	158.232,7	124.374,4	1.046	6.677
DK_DS	1.616.247,7	1293.071,9	10.877	69.422
DK_KT	959.372,6	803.082,4	6.756	43.116
EE_BP	607.090,6	214.460,3	144	154
EE_GF	6.548.995,9	3.283.813,7	2.212	2.365
EE_GR	1.133.668,1	469.263,6	316	338
FI_BB	13.429.516,5	907.529,9	5.323	6.528
FI_BS	4.665.868,2	536.894,9	3.149	3.862
FI_GF	5.256.098,1	357.041,4	2.094	2.568
LT_BP	9.668.453,9	5.901.063,7	8.978	9.361
LV_BP	1.711.925,6	1.000.097,7	517	553
LV_GR	12.245.271,4	6.325.031,0	3.267	3.496
PLC_BP	2.557.861,4	1537556,7	4.350	12.245
PLO_BP	11.758.900,9	7551183,1	21.366	60.139
PLV_BP	19.289.917,6	12410135,0	35.114	98.836
RU_BP	1.999.657,6	1507573,7	65.918	42.965
RU_GF	31.010.206,9	4903513,4	214.405	139.749
SE_BB	12.885.781,3	155395,9	695	792
SE_BP	8.312.104,4	1734091,4	7.759	8.839
SE_BS	18.018.565,2	566713,2	2.536	2.889
SE_DS	289.727,4	246605,5	1.103	1.257
SE_KT	7.164.749,8	1060018,9	4.743	5.403

Table A3.6 Base loads of nitrogen and phosphorus in sea regions.

Sea region	Base load of nitrogen (ton)	Base load of phosphorus (ton)
BB	69.893	3.451
BS	82.666	3.670
BP	293.236	16.046
GF	113.561	6.029
GR	70.076	2.209
DS	41.739	1.415
KT	73.696	1.814

Table A3.7 Nitrogen and phosphorus load reduction coefficients for cattle and pig reductions in drainage basins; specifies the content of nitrogen/phosphorus from cattle/pigs in each drainage basin (ton N/P per 100 heads).

Drainage Basin	Nitrogen load reduction coefficient for cattle	Nitrogen load reduction coefficient for pigs	Phosphorus load reduction coefficient for cattle	Phosphorus load reduction coefficient for pigs
DE_BP	1,3	0,36	0,04	0,02
DE_DS	1,3	0,36	0,04	0,02
DK_BP	1,4	0,36	0,04	0,02
DK_DS	1,4	0,36	0,04	0,02
DK_KT	1,4	0,36	0,04	0,02
EE_BP	1,1	0,36	0,04	0,02
EE_GF	1,1	0,36	0,04	0,02
EE_GR	1,1	0,36	0,04	0,02
FI_BB	1,4	0,36	0,04	0,02
FI_BS	1,4	0,36	0,04	0,02
FI_GF	1,4	0,36	0,04	0,02
LT_BP	1,0	0,36	0,03	0,02
LV_BP	1,0	0,36	0,03	0,02
LV_GR	1,0	0,36	0,03	0,02
PLC_BP	1,0	0,36	0,03	0,02
PLO_BP	1,0	0,36	0,03	0,02
PLV_BP	1,0	0,36	0,03	0,02
RU_BP	0,8	0,36	0,03	0,02
RU_GF	0,8	0,36	0,03	0,02
SE_BB	1,5	0,36	0,04	0,02
SE_BP	1,5	0,36	0,04	0,02
SE_BS	1,5	0,36	0,04	0,02
SE_DS	1,5	0,36	0,04	0,02
SE_KT	1,5	0,36	0,04	0,02

Table A3.8 Baseline information regarding fertilizer use.

Country	Initial fertilizer use – average application of nitrogen fertilizer (kg/ha)	Price of nitrogen fertilizer (€/kg)	Average price of crops (€/hkg)
Germany	133	0,57	12
Denmark	90	0,57	12
Estonia	42	0,57	12
Finland	75	0,57	12
Lithuania	34	0,57	12
Latvia	20	0,57	12
Poland	62	0,57	12
Russia	9	0,57	12
Sweden	74	0,57	12

Table A3.9 Restrictions on the use of the measures – maximum limits for the application of the measures (in the current model identical limits are set for all drainage basins, but the model allows specific limits to be set for each drainage basin).

Measure	Restriction/Limit
'Wetland'	Maximum extent of wetland restoration is set to 25% of the area of arable land
'Fertilizer'	Maximum level of reductions set to 25% of initial use
'Land Use'	The extent of under sowing with catch crops is restricted to be available for 33% of the area of arable land
'Livestock'	For cattle the maximum level of reductions is set to 80% of initial holdings.
	For pigs the maximum level of reductions is set to 80% of initial holdings.
'Sewage'	The limit for improved sewage treatment is set to 20% of the initial amount of untreated waste water
'NO_X'	The maximum reduction obtainable through application of the NO_X measure is set to 1000 ton for each drainage basin

Table A3.10 Number of PE not connected to urban wastewater treatment.

Drainage Basin	PE
DE_BP	1,560,000
DE_DS	1,740,000
DK_BP	82,400
DK_DS	3,100,000
DK_KT	1,500,000
EE_BP	10,000
EE_GF	1,265,000
EE_GR	295,000
FI_BB	9,825,700
FI_BS	1,387,970
FI_GF	2,536,330
LT_BP	3,404,400
LV_BP	276,000
LV_GR	1,978,000
PLC_BP	4,560,000
PLO_BP	12,920,000
PLV_BP	20,520,000
RU_BP	878,000
RU_GF	8,000,000
SE_BB	390,000
SE_BP	4,100,000
SE_BS	1,123,000
SE_DS	625,000
SE_KT	2,136,000

NOTE: For now the data are the total number people in the drainage basins. These will be substituted with the number of PE not connected to urban sewage treatment as soon data are delivered from the drainage basin group.

Table A3.11 Load transport coefficient of nitrogen between sea regions.

	BB	BS	BP	GF	GR	DS	KT
BB	0,05	0,95	0	0	0	0	0
BS	0,11	0,14	0,75	0	0	0	0
BP	0	0,26	0,26	0,11	0,03	0,34	0
GF	0	0	0,75	0,25	0	0	0
GR	0	0	0,66	0	0,34	0	0
DS	0	0	0,40	0	0	0,03	0,57
KT	0	0	0	0	0	0,40	0,6

Table A3.12 Load transport coefficient of phosphorus between sea regions.

	BB	BS	BP	GF	GR	DS	KT
BB	0,62	0,38	0	0	0	0	0
BS	0,11	0,24	0,65	0	0	0	0
BP	0,26	0	1	0,28	0,04	0,42	0
GF	0	0	0,81	0,19	0	0	0
GR	0	0	0,90	0	0,10	0	0
DS	0	0	0,41	0	0	0,04	0,55
KT	0	0	0	0	0	0,33	0,67

Table A3.13 Retention of nitrogen and phosphorus in drainage basins (percent).

Drainage Basin	Retention of nitrogen	Retention of phosphorus
DE_BP	34	60
DE_DS	34	60
DK_BP	10	1,8
DK_DS	10	1,8
DK_KT	10	1,8
EE_BP	23	36
EE_GF	23	36
EE_GR	23	36
FI_BB	29	26
FI_BS	29	26
FI_GF	29	26
LT_BP	0,9	2,3
LV_BP	45	40
LV_GR	45	40
PLC_BP	44	38
PLO_BP	44	38
PLV_BP	44	38
RU_BP	60	60
RU_GF	60	60
SE_BB	24	28
SE_BP	24	28
SE_BS	24	28
SE_DS	24	28
SE_KT	24	28

Table A3.14 Loads and load reductions for sewage treatment.

Drainage Basin	**Average content in waste water (ton/PE)**		**Load reduction by sewage treatment (%)**	
	Nitrogen	Phosphorus	Nitrogen	Phosphorus
DE_BP	0,00438	0,001095	80	60
DE_DS	0,00438	0,001095	80	60
DK_BP	0,00438	0,001095	80	60
DK_DS	0,00438	0,001095	80	60
DK_KT	0,00438	0,001095	80	60
EE_BP	0,00438	0,001095	80	60
EE_GF	0,00438	0,001095	80	60
EE_GR	0,00438	0,001095	80	60
FI_BB	0,00438	0,001095	80	60
FI_BS	0,00438	0,001095	80	60
FI_GF	0,00438	0,001095	80	60
LT_BP	0,00438	0,001095	80	60
LV_BP	0,00438	0,001095	80	60
LV_GR	0,00438	0,001095	80	60
PLC_BP	0,00438	0,001095	80	60
PLO_BP	0,00438	0,001095	80	60
PLV_BP	0,00438	0,001095	80	60
RU_BP	0,00438	0,001095	80	60
RU_GF	0,00438	0,001095	80	60
SE_BB	0,00438	0,001095	80	60
SE_BP	0,00438	0,001095	80	60
SE_BS	0,00438	0,001095	80	60
SE_DS	0,00438	0,001095	80	60
SE_KT	0,00438	0,001095	80	60

Table A3.15 Load reduction coefficients for the measures wetlands, fertilizer and land use.

Drainage Basin	Nitrogen load reduction by wetland restoration ton/ha	Phosphorus load reduction by wetland restoration ton /ha	Nitrogen load reduction coefficient for fertilizer pct	Nitrogen load reduction coefficient for land use change ton/ha
DE_BP	0,1	0,005	33	0,0035
DE_DS	0,1	0,005	33	0,0035
DK_BP	0,1	0,005	33	0,0035
DK_DS	0,1	0,005	33	0,0035
DK_KT	0,1	0,005	33	0,0035
EE_BP	0,1	0,005	10	0,0035
EE_GF	0,1	0,005	10	0,0035
EE_GR	0,1	0,005	10	0,0035
FI_BB	0,1	0,005	10	0,0035
FI_BS	0,1	0,005	10	0,0035
FI_GF	0,1	0,005	10	0,0035
LT_BP	0,1	0,005	10	0,0035
LV_BP	0,1	0,005	10	0,0035
LV_GR	0,1	0,005	10	0,0035
PLC_BP	0,1	0,005	10	0,0035
PLO_BP	0,1	0,005	10	0,0035
PLV_BP	0,1	0,005	10	0,0035
RU_BP	0,1	0,005	10	0,0035
RU_GF	0,1	0,005	10	0,0035
SE_BB	0,1	0,005	10	0,0035
SE_BP	0,1	0,005	10	0,0035
SE_BS	0,1	0,005	10	0,0035
SE_DS	0,1	0,005	33	0,0035
SE_KT	0,1	0,005	33	0,0035

Table A3.16 Load reduction coefficients for NO_X measure.

DE_BP	5.51%
DE_DS	5.51%
DK_BP	14.9%
DK_DS	14.9%
DK_KT	14.9%
EE_BP	4%
EE_GF	9.56%
EE_GR	6%
FI_BB	3.43%
FI_BS	3.43%
FI_GF	3.43%
LT_BP	9%
LV_BP	8%
LV_GR	8.36%
PLC_BP	5.73%
PLO_BP	5.73%
PLV_BP	5.73%
RU_BP	2.07%
RU_GF	2.07%
SE_BB	11%
SE_BP	11%
SE_BS	11%
SE_DS	11%
SE_KT	11%

Table A3.17 Measure control.

	M_WETLAND	M_LANDUSE	M_LIVESTOCK	M_SEWAGE	M_NOX
DE_BP	1	1	1	1	1
DE_DS	1	1	1	1	1
DK_BP	1	1	1	1	1
DK_DS	1	1	1	1	1
DK_KT	1	1	1	1	1
EE_BP	1	1	1	1	1
EE_GF	1	1	1	1	1
EE_GR	1	1	1	1	1
FI_BB	1	1	1	1	1
FI_BS	1	1	1	1	1
FI_GF	1	1	1	1	1
LT_BP	1	1	1	1	1
LV_BP	1	1	1	1	1
LV_GR	1	1	1	1	1
PLC_BP	1	1	1	1	1
PLO_BP	1	1	1	1	1
PLV_BP	1	1	1	1	1
RU_BP	1	1	1	1	1
RU_GF	1	1	1	1	1
SE_BB	1	1	1	1	1
SE_BP	1	1	1	1	1
SE_BS	1	1	1	1	1
SE_DS	1	1	1	1	1
SE_KT	1	1	1	1	1

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